Chapter 7: Peatlands and greenhouse gases

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Summary points:

1. **By affecting atmospheric burdens of CO_2, CH_4 and N_2O in different ways natural peatlands play a complex role with respect to climate.**
2. **Since the last ice age peatlands have played an important role in global GHG balances. By storing enormous amount of atmospheric CO_2 they have had an increasing cooling effect, in the same way as in former geological eras, when they formed coal, lignite and other fossil fuels.**
3. **GHG fluxes in peatlands have a spatial (zonal, ecosystem, site and intersite) and temporal (interannual, seasonal, diurnal) variability which needs to be considered in assessment and management.**
4. **Small changes in ecohydrology can lead to big changes in GHG emissions through influence on peatland biogeochemistry.**
5. **In assessing the role of peatlands in global warming the different time frame and radiative forcing of continuos and simultaneous CH_4 emission and CO_2 sequestration should be carefully evaluated to avoid not fully applicable global warming potentials.**
6. **Anthropogenic related disturbances (especially drainage and fires) have led to massive increases in net emissions of GHG from peatlands, which are now comparable to global industrial emissions.**
7. **Peatland drainage leads to increase CO_2 emissions, a rise of N_2O release in nutrient rich peatlands but may not significantly reduce CH_4 efflux.**

7.1 GHG related to peatlands

Peatlands are responsible for all three main greenhouse gases – carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O).
All three of the main greenhouse gases (GHG) – carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are related to peatlands, which act as a valuable or even key source and sink. Together with other GHGs, including the most important water vapour, they absorb infrared radiation emitted from the Earth and thus decrease the Earth's radiation; its energy transfer to space. Peatlands remove CO_2 from the atmosphere via photosynthesis, and the carbon not held in biomass or stored in accumulating peat, is returned back to the atmosphere as CO_2. Peatlands' anaerobic conditions are highly favorable for the production of methane (Clymo 1983) and nitrous oxide (Hemond 1983).

The physical properties of GHG molecules that affect their ability to absorb infrared radiation vary widely between compounds and determine the efficiency of a particular gas's radiative forcing. Different gases also have different lifetimes and interact in different ways with the environment. This makes comparisons between them more complicated. Additionally to direct radiative forcing impacts, some gases, such as methane and nitrous oxide have indirect impacts through the alterations they cause in atmospheric chemistry.

Table 7.1: The atmospheric lifetimes and the IPCC (1996) accepted global warming potentials over different time horizons of GHG associated with peatlands

Species	Atmospheric lifetime (years)	Global warming potential (mass basis), time horizon, years: 20	100	500
CO_2	variable	1	1	1
CH_4	12±3	56	21	6.5
N_2O	120	280	310	170

The global warming potential methodology is not directly applicable in helping to understand peatlands' net role in radiative forcing and climate change.
The competing impacts of different GHG emissions/uptake on the radiative forcing of the climate are usually compared and integrated in the widely adopted global warming potential (GWP) methodology (e.g., Ramaswamy et al., 2001; Albritton et al., 1995; Shine et al., 1990; Lashof and Ahuja, 1990), which can be used to relate the radiative forcing, over a specified time horizon, of a pulse emission of CH_4 or N_2O with a pulse emission of CO_2. An emission of CH_4 or

N_2O can then be converted into a CO_2-equivalent emission by multiplying by the GWP value. Using this approach, the emissions from any source are treated as perturbations to an otherwise constant atmosphere, although it may be argued that the assumption about a non-changing background is very unrealistic [e.g. Smith and Wigley, 2000; Lashof, 2000]. This is especially true with regard to peatlands, which both emit and uptake GHGs over their millennia history.

The global warming potential methodology is limited to assess peatland's fluxes of GHG by treating a single year's fluxes as isolated pulse emissions.

The intended application of the GWP methodology is to assess the relative climate impacts of anthropogenic emissions of greenhouse gases (Albritton et al., 1995). The role of the GWP methodology in the Kyoto Protocol process is to provide a mechanism for 'trading' among gases in a multi-gas 'basket' approach (Fuglestvedt et al., 2003). The ease and transparency of the GWP methodology (two features that are important for it usefulness as a policy tool), along with its IPCC imprimatur, have led to its widespread application. In the field of biogeochemistry, it has become common to apply the GWP methodology to compare climate impacts of ecosystem-atmosphere fluxes of greenhouse gases, treating a single year's fluxes as isolated pulse emissions.

Using the standard GWP methodology assumptions of a constant value for lifetime/adjustment time and for radiative efficiency (i.e. under the assumption of small perturbations), the GWP value for methane is a decreasing function of the time horizon chosen, due to the overall slower atmospheric adjustment of CO_2 compared to CH_4. For any pulse emission of CH_4 or N_2O, there is a set of CO_2–equivalent pulse emissions; these values are tabulated for 20-year, 100-year, and 500-year time horizons (Ramaswamy et al., 2001). The 100-year time horizon has been adopted in the Kyoto Protocol (UNFCCC/CP/1997/7/Add.1/Decision 2/CP.3) (e.g. Lashof, 2000). However, the choice of time horizon is often dictated by the specific impact under consideration (Rodhe, 1990; Albritton et al., 1995). Some components of the climate system, like troposphere temperature, may respond quickly, and so a short time horizon might be more appropriate, while others such as ice sheet dynamics, may respond more slowly, and might be better assessed by using a long time horizon (Albritton et al., 1995).

7.2 Net peatland impact on GHG radiative forcing of climate

Peatlands could historically play an important role in the control of atmospheric GHG, especially carbon dioxide and methane, levels.

The concentrations of carbon dioxide and methane were not stable in the past. Variations are largely due to changes in the global hydrological cycle, with the development and expansion of peatlands contributing either as a mediator or as a positive feedback for the atmospheric change (Prinn, 1994, Chappellaz et al. 1997). This can be assumed for the whole Quaternary period and for its Holocene epoch in particular. Although the exact contributions of wetlands to these more recent GHG variations are still under investigation, the observed change in methane values during the latter half of the Holocene (i.e. within the last ca. 5000 years) has been attributed to increasingly arid conditions in the tropics and accelerated peat development in the north (Blunier et al. 1995). It appears that regional changes in precipitation and temperature patterns over the past few millennia may have been sufficiently important to trigger some significant readjustments to the land carbon reservoir (Korhola et al. 1996, Ciais 1999). The increased concentrations of methane in the atmosphere in the early Holocene were related to the expansion of tropical peatlands (Chappellaz et al., 1990, 1993, Blunier et al., 1995, Severinghaus and Brook, 1999), and later, supported by interpolar methane gradient data, to the widespread peatland establishment in West Siberia Lowland (Smith et al., 2004).

Past variations of atmospheric mixing ratios of the two most important GHGs, CO_2 and CH_4 can be reconstructed by analyzing the composition of bubbles trapped in polar ice sheets. Throughout the Quaternary period (the last 2 million years), mixing ratios of these gases have varied in accordance with changing temperature; the main pattern being lower mixing ratios during glaciation (400-450 ppmv CH_4) and higher mixing ratios during interglacial (ice-free) periods (>600 ppmv CH_4) (Lorius et al. 1990, Raynaud et al. 1992, Brook et al. 1996). In addition, large oscillations in particular in methane mixing ratios have been noted during the warm interglacial periods (Chappellaz et al. 1993). Ice-core records of atmospheric methane concentration showed dramatic peaks in the early Holocene (Kennett et al., 2003).

Peatlands play a dual role in greenhouse gas radiative forcing of climate, affecting the atmospheric burdens of carbon dioxide, methane and nitrous oxide.
The role of peatlands as global GHG sinks and sources has often been mentioned, but both positive (e.g. Armentano and Menges, 1986, Gorham 1991, Oechel et al. 1993) and negative feedbacks (Hobbie 1996, Laine et al. 1996, Myneni et al. 1997) of GHG emissions following utilization and/or global warming have been suggested. Peatlands could have both cooling and warming impacts on the climate system through their influence on atmospheric burdens of CO_2 and CH_4. Carbon sequestration in peat lowers the atmospheric CO_2 burden, and thus causes a negative radiative forcing of climate (i.e. cooling); methane emissions from peatlands increase the atmospheric CH_4 burden, and thus cause a positive radiative forcing (warming). A positive input could also be made by N_2O. To evaluate the net result of a peatland's competing impacts on climate radiative forcing (cooling and warming), the effects of both CO_2 removal as well as CH_4 and N_2O emissions have to be quantified on a comparable basis (see Frolking et al., 0000).

Peatlands could be assessed as sources or sinks of GHGs, depending on the different time horizon under consideration – 20, 100 or 500 years.
There are several examples of using GWP methodology to assess the climate impact of wetlands based on the annual exchange of CO_2 and CH_4 at the wetland surface (see Frolking et al., 0000). Whiting and Chanton (2001) classified seven wetland sites (sub-tropical to boreal) where they had concurrently measured CO_2 and CH_4 fluxes. Without considering a time horizon, all wetlands were to be a net greenhouse gas sink. For a 20-year time horizon, all seven sites were classified as net greenhouse gas sources; for a 500-year time horizon, all seven sites were classified as net greenhouse gas sinks; and for a 100-year time horizon, the boreal sites were classified as sources, and the temperate and sub-tropical sites as sinks. Similar GWP results were found for Canadian peatlands (Roulet, 2000), for natural and managed peatlands in Finland, if to exclude emissions from storage and combustion of harvested peat (Crill et al., 2000), and for a site in western Siberia (Friborg et al., 2003). The meaning of the time horizon here is the impact horizon for a single, annual pulse emission; it is not an assessment of a continuous greenhouse gas source/sink lasting for 20, 100, or 500 years.

The net radiative forcing impact of a peatland could change its direction depending how long it exists.
The model of methane and carbon dioxide pools in the atmosphere shows that the net radiative forcing impact of a northern peatland could change from warming to cooling over the history of its life. Frolking et al., 0000 used modelled CH_4 and CO_2 pools in the atmosphere to quantify the dynamics, over years to millennia, of the net radiative forcing impact of a peatland that continuously emits CH_4 and sequesters C. Taking the observed ratios of CH_4 emission to C-sequestration (roughly 0.1–2 mol mol-1), the radiative forcing impact of a northern peatland begins, at peatland formation, as a net warming that peaks after about 50 years. It remains a diminishing net warming for the next several hundred to several thousand years, depending on the rate of C sequestration, and thereafter is or will be an ever-increasing net cooling impact. In all cases, taking various changes in CH_4 and/or CO_2 emissions, the impact of a change in CH_4 emissions dominates the radiative forcing impact in the first few decades, and then the impact of the change in CO_2 emissions slowly exerts its influence.

Specific conclusions about peatland impact on radiative forcing that do not emerge from a GWP analysis could be made if to assess the impact of sustained greenhouse gas emissions on radiative forcing.
The GWP methodology puts time-integrated radiative impacts of CH_4 and CO_2 pulses into common units (CO_2-equivalent emissions), providing a mechanism for evaluating trade-offs between the climate impacts of different gases. It does not assess the impact of sustained or variable greenhouse gas emissions on radiative forcing and the climate system at any given time. The analysis which assessed sustained GHG emission made by Frolking et al. (2007) led to several conclusions about impact of northern peatlands: – Relatively constant methane emissions maintain the atmospheric methane perturbation burden and the associated perturbation to radiative forcing at relatively constant levels. – Relatively constant C-sequestration causes an increasingly negative (or cooling) perturbation to radiative forcing. – The current radiative forcing impact of a peatland is determined primarily by a trade-off between the total C sequestered since the peatland's formation and the recent (decades) methane fluxes. For many northern peatlands that would be characterized as net greenhouse gas emitters by a 20-year or 100-year GWP analysis, the current radiative forcing perturbation due to past and present methane emissions and C sequestration is negative (i.e., cooling). This is a direct consequence of their persistence as a C sink over millennia. If peatland CH_4 and CO_2 fluxes change,

the atmosphere and radiative forcing will respond rapidly to changes in CH_4 fluxes, and more slowly to changes in CO_2 fluxes. If the methane flux stabilizes at a new value, the atmospheric burden and radiative forcing due to methane will also stabilize within a few decades.

Considering as a stable sink of atmospheric CO_2 over millennia the overall current climate impact of peatlands is a likely to be a net cooling.
Two factors influence a GHG's accumulated radiative forcing for a chosen time horizon: its radiative efficiency per molecule or unit mass, and its lifetime in the atmosphere (Albritton et al., 1995). CH_4 has a higher radiative efficiency per unit mass than CO_2, and for equal mass pulse emissions, CH_4 will initially generate a stronger instantaneous radiative forcing than CO_2. However, because CH_4 has a shorter atmospheric lifetime than CO_2, for all the time after about 65 years following equal mass pulse emissions, the remaining CO_2 in the atmosphere will generate a stronger instantaneous radiative forcing than the remaining CH_4. After 4000 years of constant fluxes, only 0.3% of the total emitted methane is still in the atmosphere, while ~20% of the CO_2 sequestered as peat has not been restored to the atmosphere from the other components of the carbon cycle (Frolking et al., 0000). At the same time most northern peatlands are at least several thousand years old (e.g. Smith et al., 2004; Clymo et al., 1998; Kuhry et al., 1993; Gorham, 1991). The area-weighted mean age of ~2600 peatlands in Finland, where primary peat formation started relatively late after glacier melt and sea land uplift, was estimated to be about 4200 years (Turunen et al. , 2002), and the basal age of peatlands of adjacent southern regions are dated by 8-10 thousand years and more (e.g. Klimanov and Sirin, 1997).

The net peatland impact on GHG radiative forcing of the climate depends on the peatlands' natural characteristics.
In the case of a peatland that emits CH_4 and takes up CO_2 (sequesters C), its overall instantaneous impact on the atmosphere must eventually be dominated by C-sequestration and will be a net cooling. However, peatlands have a wide range of GHG emission rates depending on their origin: some of them are accumulating carbon in peat and do not emit methane, others with high methane emission rate are characterized by low peat growth, and lots are in between. As applied to Finnish peatlands with an estimated area-weighted mean age of 4200 years (Turunen et al., 2002) the current radiative forcing impact of a peatland of so years old that has been an approximately constant source of methane and sink for carbon will be a net cooling if the mole-ratio of CH_4 emission to C sequestration is less than 0.75, and a net warming if the ratio is greater than 0.75 (Frolking et al. 0000). At the same time the existing data show that peatlands with flux ratios both less than and greater than 0.75 could be found (Laine et al., 1996; Alm et al., 1997; Minkkinen et al., 1999, 2002).

The methods applied to assess the net role of peatlands and their utilization in understanding global warming should carefully evaluate the competition between the quick, strong warming from CH_4 emissions and the slow cooling from CO_2 uptake.
Peatland greenhouse gas fluxes will inevitably involve competition between quick, strong warming from CH_4 emissions and slow cooling from CO_2 uptake. The methods used to evaluate this competition can obscure or highlight the dynamics. These dynamics are important for our understanding of past changes, and for the assessment of possible future paths for emissions and uptake from peatlands. If the methane fluxes from northern peatlands (or another source) changes significantly and rapidly, the atmospheric methane burden and associated radiative forcing will respond in decades, possibly stabilizing at a new level.

7.3 Ecological and environmental control of GHG emission from peatlands

7.3.1 General

Patterns and controls of GHG emissions from peatlands may vary depending upon the spatial and temporal scale being examined. The factors affecting these emissions are thought to be hierarchically related according to their respective scales of importance.
The emissions of GHGs related to peatlands are influenced by a wide range of biological, physical and chemical processes which are interrelated in a hierarchical fashion. Given that these processes are tightly coupled, relationships between GHG emissions and controlling factors should be found when comparisons are made at the appropriate scales (Klinger et al. 1994). Some of these processes may affect the magnitude of GHG fluxes, while others may affect the spatial and temporal distribution. Peatland ecology could strongly control regional GHG emissions

by influencing net primary productivity, species composition, community structure, peat characteristics, and landscape hydrology. Site–to-site variations in mean GHG fluxes could be closely related to the mean water-table level, and for a specific site for a period without water-table changes, GHG fluxes will tent to follow soil temperature fluctuations.

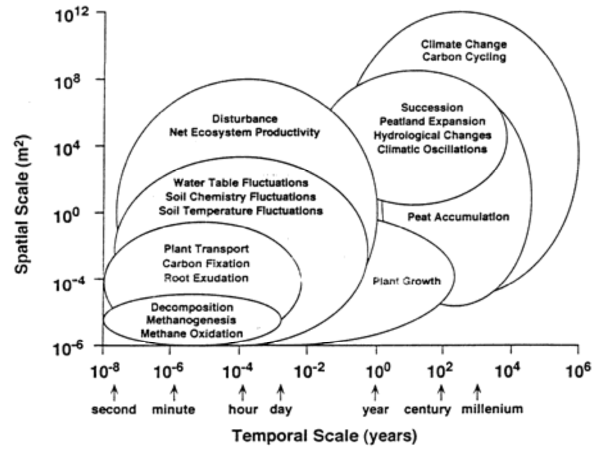

Figure 7.1: The hierarchical relations of processes proposed to affect methane emissions and the spatial and temporal scales at which these processes predominate (Klinger et al., 1994)

Peatland hydrology (sources and quality of water, water flow direction and rate, depth to the water table, etc.) is the single most important condition influencing peatland ecology and biogeochemistry, as well as the level of CO_2, CH_4 and N_2O flux.

The quantity and quality (chemistry) of water coming to the peatland via precipitation, groundwater discharge, upland inflow, flooding, or other sources is the most important condition influencing peatland ecology, development, functions and processes. Water chemistry has a large influence on the plants that can occur in a peatland, and therefore on the character of peat that accumulates. The hydrological origin of a peatland defines the key ecological factors – depth to the water table and its fluctuations in time, and the direction and rate of water movement in surface and deep peat layers. Water exchange in peat is the main driving factor of mass and energy exchange in peat deposits. There is a strong link between temperature and water regime in peat deposits. Water delivers various dissolved substances and suspended particles both upward and downward. This could support GHG production and movement (e.g. Sirin et al., 1998a). GHGs can even be released from a peatland horizontally with the lateral outflow (Sirin et al. 1998b), while a significant amount of organic matter may leave the peatland dissolved in water (Kortelainen & Saukkonen 1994, Sallantaus 1992), later supporting GHG production in adjacent aquatic systems like streams, ponds and drainage ditches.

Atmospheric CO_2 fixed to mire plant biomass through photosynthesis, is the primary source of carbon GHGs (CO_2 and CH_4) emission from peatlands. Thus, net primary productivity of peatland vegetation is potentially controlling CO_2 and CH_4 fluxes.

Part of the carbon photosynthesized by plants is returned to the atmosphere as CO_2 in the maintenance and growth respiration of above- and below-ground parts of plants and their associated heterotrophic microbial communities. In the aerobic surface peat layer about 80–95 % of the litter is decomposed by aerobic bacteria and released as CO_2 (Reader and Stewart 1972, Clymo 1984, Bartsch and Moore 1985). The remaining C is transformed into plant structures, and finally deposited as peat. In the underlying water-saturated anaerobic peat layers, a large portion of the available organic carbon is used to form CH_4 as the end product, later released to the atmosphere as itself, or being oxidized by methanotrophic bacteria in the upper aerobic peat layers, and diffusing upwards to CO_2 (e.g. Sundh et al. 1994). Thus, being a primary source of organic carbon in peatlands net primary productivity (total C fixed by plants minus carbon used in respiration) is a potentially controlling factor for CO_2 uptake and release and CH_4 release.

The water table determines the oxic-anoxic ratio of the peat profile and thus the preference for aerobic or anaerobic biogeochemical processes influencing the rate and ratio between GHG emitted from a peatland.

The position of the water table determines the oxic-anoxic ratio of the peat profile that defines the preference for aerobic or anaerobic biogeochemical processes. It is the key abiotic factor that influences the type of microbial respiration – aerobic versus anaerobic, and finally the rate of peat decomposition. The aerobic metabolism is more efficient and tends to favour the rapid production of CO_2. The water table influences the ratio between CO_2 and CH_4, two main GHGs emitted from peatlands. Up to 90% of the CH_4 produced in the anaerobic peat zone may be oxidized above the water table in peat (e.g., Fechner and Hemond, 1992; King et al., 1990), but in wet sites or during wet periods, it could directly release to the atmosphere. The aerobic conditions in peat could stimulate nitrification of peat matrix nitrogen and potentially support further N_2O production.

Temperature could widely influence of GHG flux in peatlands. CO_2 and CH_4 production, CH_4 oxidation are temperature dependent. Surface temperature affects GHG release from a peatland to the atmosphere.

Temperature directly affects decomposition through its influence on microbial activity. CO_2 and CH_4 production and CH_4 oxidation are all temperature dependent. Rising temperatures could stimulate both processes to a different extent. Temperature affects the solubility of GHGs in peat, and thus determines the mechanism of their upward movement. Higher surface peat temperatures lead to higher CO_2 respiration and CH_4 emission supporting diffusion between soil and atmosphere. Soil temperature is the best predictive environmental variable for the seasonal dynamics of in situ CO_2 and CH_4 emission. Temperature convection in northern peatlands could support water exchange in deep peat layers with potential effects on GHG production and release (Sirin, 2003).

Substrate that include peat matrix, dissolved substances and suspended particles in peat pore water, serve as the energy source for GHG production but only in the form of suitable material. Substrate quality could support or inhibit GHG production in peatlands.

The substrate composition of peat is highly variable and this is dependent on the original peat-forming plant communities. Generally, peat substrate should include peat matrix, dissolved substances and suspended particles in peat pore water. This material serves as the energy source for the microorganisms, so differences in the substrate quality result in differences in peat decomposition rates, and finally in the resulting GHG production. Differences in CO_2, CH_4 and N_2O production are not directly due to differences in peat nutrient levels, and are mainly determined by the availability of suitable material. When peat becomes depleted of high-quality substrate GHG production falls.

Net ecosystem production is a master variable, integrating many factors that control CH_4 emission in vegetated peatlands.

Peatlands with high net ecosystem productivity could increase CH_4 fluxes because of enhanced transport via plants (Chanton and Dacey, 1991) and/or enhanced methanogenesis through root senescence, decay, or exudation (Whiting and Chanton, 1992). The idea that CH_4 emissions are directly related to net primary production via substrate availability and plant transport were shown in field studies in the Florida Everglades (Whiting et al, 1991), subarctic Canada (Whiting and Chanton, 1992), and arctic Alaska (Morrisey and Livingston, 1992). From simultaneous measurements of CO_2 and CH_4 exchange in wetlands extending from subarctic peatlands to subtropical marshes, Whiting and Chanton (1993) found a positive correlation between CH_4 emission and net ecosystem production, and suggested that net ecosystem production is a master variable, integrating many factors that control CH_4 emission in vegetated wetlands. According to this study, about 3 per cent of the daily net ecosystem production is emitted back to the atmosphere as CH_4. Net primary production may also be indirectly related to the ability of certain plants to transport CH_4 through their stems and leaves by providing metabolic energy for transpiration or for oxygen transport to the roots (Klinger et al., 1994). The relationship derived between primary production and methane emission provides an important tool for refining global scale source estimates (Whiting and Chanton, 1993).

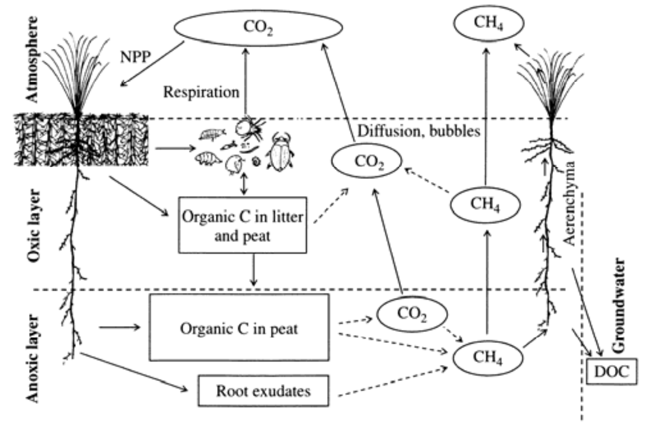

Figure 7.2: Simplified description of carbon flow and peat formation in a peatland with an oxic upper part and an anoxic layer beneath. Encircled symbols represent gases, and dashed arrows show microbial processes. NPP is the net primary production (i.e. the difference between photosynthesis and the plants' respiration). DOC is dissolved organic carbon leaching out from both the oxic and anoxic layers of the peatland via the groundwater. (After Rydin and Jeglum, 2006)

7.3.2 Carbon dioxide

CO_2 emission from a peatland includes autotrophic respiration, regulated by photosynthesis and temperature, and heterotrophic respiration controlled largely by soil temperature.

CO_2 emission from a peatland is formed by autotrophic and heterotrophic respiration, which comprise a significant part (in northern peatlands around 1/3 – Bubier et al, 1998) of the CO_2 uptake in photosynthesis during intensive growth. The rate of autotrophic respiration is regulated by photosynthesis and temperature while heterotrophic respiration is controlled largely by soil temperature (Chapman et al, 1998). Total CO_2 release from peat includes aerobic and anaerobic decomposition, root respiration and respiration of soil fauna. Root-associated respiration follows the phenology of the vegetation and may account for 10-45% of the total soil CO_2 release (Silvola et al., 1996), mainly originating from the turnover of fine root litter and root exudates.

7.3.3 Methane

The flux of CH_4 from a peatland is a function of the rates of CH_4 production and consumption in the profile and the transport mechanisms to the atmosphere, such as diffusion, ebullition, or plant-mediated movement.

The methane flux from a peatland is dependent not only on the production rate of methane through anaerobic degradation of organic matter, but also on the transport pathways and consumption of methane in oxic parts of the system. Methane formation is restricted to the anoxic part of the profile below the actual water-table position. Methane in the saturated part of the profile occurs either as a dissolved species or as bubbles, and can be transported to the atmosphere by: 1) molecular diffusion through the soil pore space and, if present, the snow pack or water standing above the peatland surface, 2) transport by ebullition to the atmospheric interface at the water table, and 3) transport through aerenchymous tissues in plants from the rooting zone directly to the atmosphere. A variable fraction of the produced methane is oxidised by methanotrophic bacteria in the oxic zone above the water table, and either in the rhizosphere or inside vascular plants.

CH_4 production depends on the amount of high-quality organic material that reaches the anoxic zone. This means that plant primary productivity and depth to water table are the two most important factors controlling this process.

CH_4-producing organisms grow anaerobically beneath the water table, and their main zone of activity shifts up and down with the water table. Their most common substrates are H_2 + CO_2 (to form CH_4+H_2O) and acetate (to form CH_4 + CO_2). The primary control on CH_4 production is the amount of high-quality organic material (fresh litter and root exudation) that reaches the anoxic zone. This means that plant primary productivity, vegetation type and depth to water table are the most important factors, influencing the process. A minerotrophic *Carex rostrata peatland site* had higher rates of CH_4 production than nutrient-poor Sphagnum-dominated sites, probably owing to the higher supply of easily degradable litter and root exudates (Bergman *et al.* 1998). Within a bog, *Sphagnum majus* peat produces 1.5 times more CH_4 than *S. fuscum* peat. Where the water table is closer to the surface, the upper anoxic zone will contain greater amounts of fresh, resource-rich material. Secondary controls on methane production are alternative electron acceptors, which could inhibit this process, temperature, and pH. Occurrence of competing electron acceptors like nitrate (NO_3^-), ferric iron (Fe^{3+}), or sulfate (SO_4^{2-}) will lead to decreased CH_4 production.

Depth to water table is the key factor controlling the balance between CH_4 production and consumption and finally the ratio of CH_4 and CO_2 emitted from a peatland.

The most active zone of CH_4 production follows the depth to water, and is located just below it. The methane production potential in peatlands has been found to peak approximately 10 cm below the average water table (Sundh *et at.* 1994). A large fraction of the methane transported by diffusion through the oxic part of the peat profile is oxidised by methanogenic bacteria. This portion reached up to 80% in a swamp forest (Happel and Chanton, 1993) and 90% in a *Sphagnum* bog (Fechner and Hemond, 1992). Oxidation capacity is also suggested to be maximal around the mean water table (Sundh et al., 1994). The position of the actual water table determines both processes of CH_4 production and consumption, multiplying its effect and defining the rate of CH4 emission. On a spatial level, the highest emissions of CH_4 from will be from peatland surfaces close to the water table. Wet periods with high water levels also provide better conditions for methane release to the atmosphere.

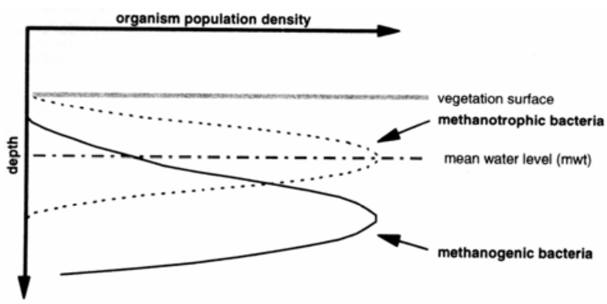

Figure 7.3: A generalized depth profile describing distribution of the methanogenic and methanotrophic communities in relation to the mean water table in a peatland (after Granberg et al., 1997).

Temperature can influence methane production and consumption as both processes are temperature dependant.
Methane production increases exponentially with increasing soil temperature (Svensson, 1984; Westermann, 1993; Bergman et al., 1997). The optimum temperature observed in the laboratory for methane production is 25-30°C (Dunfield et al., 1993) and production occurs down to 2°C provided there is no substrate limitation (Bergman et al., 1997). Like production, the consumption rate is also determined by soil temperature. Consumption of methane occurs at temperatures between 0 and 35°C, with optimum consumption at 20-25°C (Dunfield et al., 1993). The decrease in CH_4-oxidation at low temperatures might be due to the decrease in substrate availability as an effect of reduced numbers of methanogens (Saarnio et al., 1997).

Specific peatland vegetation could provide a direct route for methane release to the atmosphere, bypassing oxidation, with a final effect on high emission rates.
A substantial amount of the CH_4 emitted from peatlands passes through vascular plants (e.g. Sebacher et al., 1985; Schimel, 1995, Thomas *et al* 1996), bypassing the attacks of the methanotrophs (e.g. Saarnio *et al.* 1997; Nilsson *et al.* 2001b). In an adaptive response to submerged soils, vascular plants have developed porous tissues (aerenchyma), which facilitate O_2 transport to the roots (Armstrong, 1991) and on the other hand allow dissolved or trapped gases in the sediment to pass to the atmosphere. Transport through the vascular plants is usually the dominant emission pathway when peatland is covered by appropriate vegetation (e.g. Chanton and Dacey, 1991, Chanton and Whiting, 1995, Thomas et al. 1996). Such peatland species as *Phragmites sp., Typha spp., Scheuzeria palustris* and others could significantly enhance CH_4 release to the atmosphere (e.g. Grosse et al., 1996, Shannon et al., 1996). Molecular diffusion is the most important transport mechanism in most sedges (one of the dominating vascular plant groups in peatlands). Peatland sites with a high cover of aerenchymatous plants are characterized by high emission rates. Transport via plants could make the CH_4 cycle relatively independent of the long-term C cycle. Peatlands in their fen stages act as sedge-mediated CH_4 pumps, converting atmospheric CO_2 to CH_4 (Korhola et al. 1996).

7.3.4 Nitrous oxide

Environmental conditions of undisturbed peatlands are rather favorable for N_2O production, but in most cases it is severely limited by low nitrate concentrations.
Emission from soils of nitrous oxide and the other nitrogen oxide gases (NO and NO_2) results from the cycling of N in the soil. Where N cycling is rapid, through nitrification and denitrification, and the microbial cycle "leaky", significant emissions of these gases are likely to occur (Davidson 1991, Moore 1994). The environmental conditions within an undisturbed peatland (i.e. anaerobic and relatively acidic, Hemond 1983), are considered highly favourable for N_2O production through denitrification, although generally low nitrate concentrations in peatlands (Clymo, 1984) severely limit the extent to which the process can proceed.

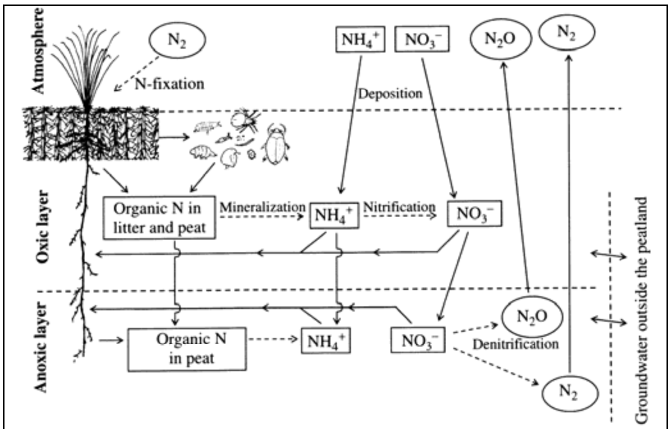

Figure 7.4: Simplified scheme of the nitrogen cycle in peatlands. Encircled symbols represent gases, dashed arrows – microbial processes. The largest pool in the peatland is the organic nitrogen in peat which is unavailable to the plants. In the oxic zone, plant uptake includes the ecologically important assistance of mycorrhizal mutualism. Some microbial nitrogen fixation occurs in the anoxic zone, based on N_2 gas transported down through plant aerenchyma. The bidirectional arrows to the right indicate the exchange with groundwater outside the peatland, that is, leaching and inflow of inorganic components and nitrogen in dissolved organic matter. (after Rydin and Jeglum 2006).

Limited by low nitrate concentrations, N_2O production in peatlands can be raised by the external sources or by higher nitrification of the organic-N within the peat matrix.
The lack of suitable for N_2O production in peatlands can be compensated by external sources, or by stimulating mineralisation (and nitrification) of some of the organic-N within the peat matrix (Williams & Wheatley 1988). The aerobic processes of nitrification are a potential nitrous oxide source (Bremmer & Blackmer 1978). Water samples from a gradient from bog to extreme rich fen (Vitt *et al.* 1995) showed decreasing NH_4^+ and

weakly decreasing NO_3~. This may mean that the vegetation in the rich fen is consuming the supply of mineralized N faster than in the bogs. In cases where organic-N is available from peat matrix, the extension of aerobic layer in both depth and time could support N_2O production.

7.4 GHG flux rate in natural peatlands

Peatlands emit large amounts of CO_2 to the atmosphere even though, in general, they are its net sink. (This is clearly shown by the occurrence of peat accumulation). However, different peatlands in different geographical conditions and over specific time periods (season, year or longer) could act both as a sink and a source of CO_2.
Soil respiration measurements, which include the C released by decomposition of organic matter as well as the respiration of plant roots and heterotrophic organisms, give average figures for annual CO_2 emissions between 50 and 400 g C m^2 a^{-1}, depending on the climate and peatland site type (Raich and Schlesinger 1992, Moore 1996, Silvola et al. 1996a). Micro-topographical differences within sites (e.g. Moore 1989, Waddington and Roulet 1996) and differing climatic conditions between years (Silvola et al. 1996a) further increase the variation in CO_2 fluxes from peatlands. Root respiration may account for 10-40% of soil respiration in peatlands, the major part of which is probably derived from decomposing root exudates, not from the maintenance respiration of roots (Silvola et al. 1996b). The variation of CO_2 flux in peatlands could be very high.

Peatlands are one of the main sources of atmospheric methane
Peatlands together with other wetlands are the major biogenic source of atmospheric methane, contributing about 20% of the total input of 552 Tg CH_4 yr^1 (IPCC, 1995). Many peatlands emit methane, and northern peatlands only are currently contributing ~3 to 5% of total global methane emissions (Prather et al., 2001; Milakoff Fletcher et al., 2004). Wetlands in high latitudes have been identified as one of the main sources of CH_4 emission to the atmosphere (Cicerone and Oremland, 1988; Fung et al., 1991). Estimates of the contribution from high latitude peatlands, at 50°–70°N, have converged around 17-35 Tg CH_4 yr^1 (Aselmann and Crutzen, 1989; Fung et al., 1991; Taylor et al., 1991, Bartlett and Harris, 1993; Reeburg et al., 1994; Roulet et al., 1994). A survey published in 1991 estimated that Swedish wetlands emit 2.2 Tg CH_4 yr^1, which would account for about 70% of the total CH_4 emission from Sweden (Svensson et al., 1991). Annual CH_4 emissions from boreal peatlands have varied between 0 and 70 g CH_4 m^{-2} a^{-1} (Crill et al. 1992), with mean fluxes for Finnish undisturbed bogs and fens being 8 and 19 g CH_4 m^{-2} a^{-1} respectively (Nykanen et al. 1998). Usually these comprise less than 10% of the annual net C flux from peat to the atmosphere (Aim et al. 1997).

Based on the differences in C stores between mineral subsoils under young peatlands (<500 years) and adjacent upland soils, the C input into the mineral subsoil has been estimated at 10-20 g C m^{-2} a^{-1} (Turunen et al. 1999a).

Natural peatlands make relatively small contributions to N_2O emissions to the atmosphere
Even though relatively few data are available on the flux of N_2O from undisturbed peatland, studies show very similar low emissions. For example, in the Hudson Bay lowland the annual flux was between -2.1 to 18.5 mg N_2O m^{-2} (Schiller and Hastie 1994), Goodroad and Keeney (1984) observed annual fluxes of 0.02 to 0.08 g N_20 m^{-2} from undrained marshes, Urban et al. (1988) reported fluxes of < 0.2 to 0.6 µg N_2O m^{-2} h^{-1} from Minnesota and western Ontario ombrotrophic peatlands, while Yavitt and Fahey (1993) measured fluxes of 14 µg N_2O m^{-2} h^{-1} from a peaty soil in a New England forest. Even if nitrogen concentrations in peats are fairly high (generally 0.8-2.5% of dry matter) N_2O emissions have been found to be negligible (Martikainen et al. 1993). N_2O emissions from all Finnish undrained peatlands are estimated to be only 0.2 Gg (Crill et al. 2000). Laboratory estimations of the annual flux of nitrous oxide based upon a daily flux of 0.11 mg N_2O m^2 d^1 assumed to be constant year round, and an area of intact sub-arctic/boreal peatland of 3.3×10^{12} m^2 (Gorham 1991), yields an annual flux of 0.133 Tg yr^{-1} (Freeman et al. 1993). This represents a relatively small contribution to global N_2O emissions of 12.4 Tg N_2O yr^{-1} (Davidson 1991).

Uncertainties of estimates on GHG emissions from peatlands are very large because of spatial variability depending on their natural origin, geographical location and weather conditions.
Temporal and spatial variations of GHG emissions from peatlands can vary by up to 100-1000 x depending on the natural origin of the site, geographical location and weather conditions. Spatial differences in GHG fluxes in peatlands can be seen on geographical, typological and site levels. Neighboring peatlands or even micro-sites could simultaneously be a source and a sink of GHGs. On

a raised bog in summer, hollows were actively emitting CH_4 while closely adjacent elevated sites with pine dwarf-shrub vegetation were absorbing CH_4 (Chistotin et al., 2006). Thus, it is extremely hard to make general spatial estimates of GHG flux from peatlands, considering their ecosystem, site and micro-site diversity. Use of a combination of in situ measurements with remote sensing technologies is the only way to extrapolate GHG flux observation data for spatially relevant territories.

For GHG flux measurements in peatlands different methods, most of which are based on chamber or micrometeorological techniques, can be applied. They provide data at scales ranging from <1 m^2 (chambers) to several hundred m^2 and km^2 (micro-meteorological towers and aircraft platforms). CH_4 and N_2O flux, and short-term net CO_2 exchange can be measured using dynamic or static chambers (e.g. Silvola et al. 1985, Crill et al. 1988), but on tree-covered peatlands micrometeorological methods using towers offer the only direct way of possibly measuring the net CO_2 exchange (e.g. Fowler et al. 1995) of the whole peatland ecosystem. Chamber techniques allow for micro-scale studies, i.e. within hummock, hollow and lawn communities, but a sufficient number of chambers need to be employed at each peatland site if reasonably precise spatial estimates are to be obtained. Chamber measurements give gas exchange values in the specific conditions during the chamber closure only. These instantaneous flux rates, sensitive to peat moisture, temperature and other conditions, changing in seconds or hours, must be expanded in time. For this, different model approaches were developed, ranging from simple regression-based to physiologically based models (e.g. Silvola and Hanski, 1979, Granberg et al. 1997, Alm et al. 1997, 1999c, Saarnio et al. 1997, Kettunen 2000, Tuittila et al. 2003). Micrometeorological flux measurement techniques (gradient, eddy covariance and other methods), allow continuous measurements at the level of the whole mire site (Hypponen and Walden 1996, Aim & al. 1999c, Aurela et al. 2004). However, rather high costs, the need for a long monitoring period, the need for large homogenous peatland areas and even methodological concerns, restrict the usability of this method. For specific conditions, for example, during wintertime, special approaches like snow gradient methods can be employed (Sommerfed et al. 1993, Aim et al. 1999b, Saarnio et al. 2003). Generally a combination of different methods provide better estimates of GHG flux data in peatlands.


Figure 7.5: GHG flux measurements. Chamber measurements in tropical peatlands (left), micrometeorological measurements in tundra peatlands (right).

Local uncertainties such as hollows, pools, streams could play a valuable and sometimes key role in GHG emissions from peatlands
Local uncertainties could provide specific conditions for GHG production and release in peatlands. Thus, their contribution to the GHG flux from a peatland could be relatively much higher their surrounding area. Hollows, pools and streams support release of GHGs partly delivered by lateral flow from adjacent peat layers and partly produced in situ. For example, ditches in drained peatlands are a cause of increased CH_4 emissions and this partly offsets reductions in CH_4 emissions due to the drainage of peat soils.

GHG fluxes in peatlands demonstrate high temporal variability. To estimate average GHG emissions, diurnal and seasonal fluctuations are to be considered, and the annual flux rate could change between years with different weather conditions.
High temporal variability in GHG flux is observed in peatlands. Three different time scales can be identified: annual, seasonal and diurnal. Seasonal changes in GHG flux are related to the seasonal

changes of peatland ecosystems and could be very large, varying from absorption to high emission rates. Usually the emission peaks when the above-ground biomass in peatlands is at a maximum. A large proportion of GHGs, especially methane, could release over short time episodes (e.g. for northern peatlands, this takes place during the spring thaw period). The variation on an annual scale may, to a large extent, be determined by differences in inter-annual and seasonal weather conditions. Most GHGs related to peatlands demonstrate diurnal variability, which in turn, is dependent on the weather conditions (mostly relating to temperature changes and precipitation). The most promising approach to estimate annual GHG flux in peatlands is to use correlations between environmental parameters and measured emissions to reconstruct flux patterns over longer periods (Granberg et al., 1997).

Non-growing season emissions could make valuable contributions to GHG fluxes from northern peatlands. A considerable amount of CO_2 and CH_4 could be released through the snow pack during winter. Very high emission rates of CH_4 may take place after the snowmelt in spring.

Winter efflux was found to be an important component of the annual C balance; carbon losses during winter may constitute 10-30% of the growing season net carbon gain in the boreal zone (Zimov & al. 1993, Aim & al. 1999c). Methane release alone, during the long boreal winter, is significant and consists of 5-33% of the annual total (Dise 1992, Aim & al. 1999b). Under Finnish seasonal weather conditions about 80% of the emissions of CO_2 and CH_4 occur during the growing season (Aim et al. 1999), but a considerable part (20%) of the C fixed in the ecosystem is lost during winter. CH_4 stored in deep peat layers is released during warm periods but rebuilt in winter (Sirin et al. 1998). A large fraction of the total emissions could occur via spring episodic events. Significant amount of CH_4 captured below the freeze layer may be emitted during the snowmelt.

7.5 Human influence on GHG flux from peatlands

Human intervention and land-use may create multiple pressures on the GHG flux in peatlands, mainly connected with water level draw-down

There are a wide range of human activities that have a potential effect on GHG flux in peatlands. Peatland utilization for excavation, forestry, agriculture and other purposes is mostly limited by highly saturated conditions and therefore drainage is often needed. Most peatland land-use activities bring about changes in the ecohydrology of sites, together with changes in, or the removal of, vegetation. The water level draw-down is the initial factor that has an influence on GHG formation and release in peatlands. Heterotrophic CO_2 efflux from the peat increases after drainage, and litter and peat decomposition rates increase, as decomposition in aerobic conditions is always much faster than in anaerobic ones. Water-level draw-down will cause a decrease in the CH_4 emissions, as substrate flux to anoxic layers is decreased (slowing down CH_4 production) while consumption of CH_4 in the thicker aerobic layer is enhanced. N_2O fluxes from natural peatland mires are small but drainage has been shown to increase the fluxes at nutrient-rich sites.

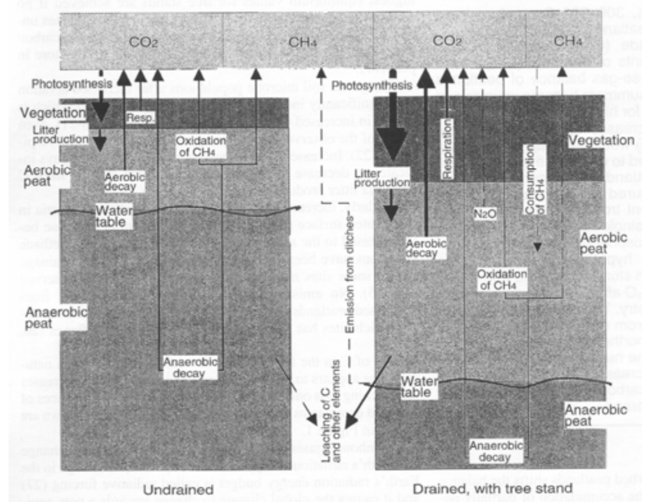

Figure 7.6: Schematic presentation of the GHG balances of undrained and drained peatland sites. The line widths show approximate relative flow rates and dotted lines indicate potential fluxes (Laine et al., 1996)

Arable agriculture always transforms peatlands into sources of GHGs to the atmosphere (first of all CO_2 and very often N_2O). CH_4 emissions from drained peat soils are generally prevented, though they can be rather high from drainage ditches.

Arable agriculture always transforms peatlands into net sources of GHGs to the atmosphere (Armentano and Menges, 1986; Kasimir-Klemedtsson et al. 1997; Maljanen et al. 2004), with the exception of CH_4. Very high annual losses of C, up to ca. 1000 g m^{-2}, have been reported from Northern Europian peat soils (e.g. Maljanen et al. 2001). Similarly, N_2O fluxes may also be high in comparison to other ecosystems, at more than 10 kg N_2O-N ha^{-1} (Maljanen et al. 2003, 2004),.

Kasimir-Klemedtsson et al, (1997) concluded that agricultural practices on organic soils lead to a net increase in radiative forcing due to large fluxes of CO_2 and N_2O, despite decreases in emissions of CH_4. To some degree, the impact on the carbon balance can be controlled by the choice of cropping method, including alternative crops and taking into account the depth of drainage. Ditches in drained peatlands are a cause for increased CH_4 emissions (Chistotin et al., 2006) and this partly offsets reductions in CH_4 emissions due to the drainage of peat soils.

Peatlands used for agriculture are major global emitters of N_2O
N_2O fluxes from natural peatland mires are small and drainage has been shown to increase the fluxes at nutrient-rich sites. Agricultural soils are responsible for most of the global N_2O emissions from soils (Kroeze et al. 1999). Drained organic soils with high N_2O emissions have special importance in the atmospheric N_2O load. As much as 25% (annually 4 Tg) of the N_2O emissions in Finland may originate from organic agricultural soils (Kasimir-Klemedtsson et al. 1997), although these soils cover only 14% of the total agricultural land in the country. Also, when peatlands are drained (or drained and fertilized) mineralization increases and inorganic forms of N increase. When organic N is available from peat matrix, the extension of aerobic layer in both depth and time could support N_2O production and cause further efflux to the atmosphere.

Drainage for forestry causes C losses from soil but decreases CH_4 emissions and on nutrient-rich sites may stimulate N_2O release to the atmosphere
Following drainage, litter and peat decomposition rates increase, since decomposition in aerobic conditions is always much faster than in anaerobic ones. Consequently, heterotrophic CO_2 efflux from the peat increases (Moore and Dalva 1993, Silvola et al. 1996a). However, the effect of increased aeration on increased decomposition rates may be accompanied by decreases in peat pH (Laine et al. 1995a), low peat temperature (Minkkinen et al. 1999) and reductions in litter quality (Laiho and Laine 1996, Laiho et al. 2004), all of which are important determinants of the rate of organic matter decomposition (Berg et al. 1993). For northern peatlands, much information is available about the effects of water level draw-down on carbon fluxes in peatlands drained for forestry (e.g. Glenn et al., 1993; Roulet et al., 1993; Laine et al., 1995a, b; Martikainen et al., 1995; Minkkinen and Laine, 1998, Minkkinen et al. 1999, Silvola et al. 1996, Hargreaves et al. 2003, von Arnold 2005a, 2005b, Minkkinen et al. 2006a). Water level draw-down will cause a decrease in CH_4 emissions as substrate flux to anoxic layers is decreased (slowing down CH_4 production) and consumption of CH_4 in the thicker aerobic layer is enhanced (Glenn et al., 1993; Roulet et al., 1993; Martikainen et al., 1995; Roulet and Moore, 1995, Nykänen et al. 1998). But CH_4 release may still remain in drained peatlands, originating from drainage ditches. Minkkinen et al. (2006b) show that tree stand volume in peatland forests can be used as a scalar in the upscaling of CH_4 emissions. N_2O fluxes from natural mires are small and drainage has been shown to increase the fluxes at nutrient-rich sites (Martikainen et al. 1994, 1995, von Arnold et al. 2004). In general it has been suggested that reduced CH_4 emissions after draw-down of the water level, together with increased C sequestration in trees, may decrease the greenhouse effect of these ecosystems during the first tree stand rotation (Laine et al., 1996, Minkkinen et al. 2002), even though soil CO_2 and N_2O emissions may simultaneously increase.

Forest harvesting in tropical swamp forests supported by drainage may increase the CO_2 release to the atmosphere and cause additional changes in the other GHG flux rates
Forest harvesting in tropical swamp forests can result in changes to the quality and quantity of organic matter inputs from vegetation and, as the work of Brady (1997) has shown, if tree root mats decline, then the net accumulation of peat may also decline. Where selective logging is combined with artificial drainage, decomposition and subsidence of the peat may proceed at annual rates of 3.5–6.0 cm (Brady, 1997). This will be accompanied by an increase in CO_2 release.

Peat harvesting (including land preparation, extraction and abandonment) strongly affects the GHG flux in peatlands. The CO_2 release from dried peat rises, but CH_4 efflux from drainage ditches and under wet conditions can remain.
The production cycle for peat harvesting has three phases: 1) land conversion for peat extraction which includes drainage and removal of natural vegetation; 2) peat extraction when peat is milled, air-dried in summer and collected into stockpiled for later pickup; 3) abandonment when extraction is no longer profitable (Canadian Sphagnum Peat Moss Association 2004, Nilsson, Nilsson 2004). Generally greenhouse gas emission changes have similar features in all three phases. Removal of vegetation and intensive drainage to reach the harvesting moisture content of peat has a fundamental impact on the GHG balances of the harvesting sites. The major greenhouse gas flux in this process

involves CO_2 emissions from the excavated peat area and the stockpiles, as the decay of the drained peat increases. It was expected that methane flux from excavated peatlands would be negligible or almost stop (Nykänen et al., 1995b). But relatively high CH_4 efflux was observed from main and secondary ditches (Minkkinen et al., 1997; Sundth et al. 2000; Chistotin et al., 2006; et al.). CH_4 emissions may take place from the harvested area when the peat is wet. Notable CH_4 flux rates were observed after the snowmelt in spring from the milled peat surface while from the stockpiles, rates reached 227±120 mg C CH_4 $m^{-2}h^{-1}$ (Chistotin et al., 2006).

Fertilisation of drained peatlands may cause increased N_2O emissions and affect the decay rate, with subsequent changes to carbon GHG flux. Nitrogen input in a form of anthropogenic deposition via air pollution and water contamination could also have an influence on GHG flux from pristine as well as drained peatlands.
Nitrogen fertilization on boreal nutrient poor pristine peatlands did not produce increased N_2O emissions (Nykänen et al. 2002), but emissions increased significantly from soils drained for forestry after N addition (Regina et al. 1998). Increases were also observed from cropland peat soils (Maljanen et al. 2003), and addition of N to rewetted peatland has caused very high N_2O emissions (Silvan et al. 2002). The C/N ratio of the organic matter affects the decay rate, and fertilization could cause changes in CO_2 and CH_4 flux rate from peatlands. High nitrogen inputs to peatlands, for instance in the form of anthropogenic deposition, affect many parts of the N cycle and give higher rates of N mineralization, denitrification, and N_2O emission (Verhoeven et al. 1996). The N could originate from air pollution or particularly from water contamination and waste deposition.

Overgrazing of peatlands can cause soil degradation. Fertilisation with manure stimulates peat oxidation and erosion, which in turn increase the release of CO_2 to the atmosphere and have an additional effect on in situ and offsite CH_4 and N_2O emissions
Peatlands are often used for grazing. All highly productive pastures in the Netherlands have a peatland origin. Many abandoned agricultural lands with peat soils in Eastern Europe are put under grass now, as are the previously excavated peatlands in the European Russia. In steppe and mountain regions with a dry climate, peatlands nowadays are the most productive and attractive lands for grazing. This is often because grasslands on mineral soils have already been destroyed by overuse and drought. Overstocking of grasslands on peat soils as observed on the Rourgai Plateau in China, in Mongolia (Minayeva et al. 2004) and in the other regions, led to vegetation changes and soil degradation. Tussock formations, further burrowed by small mammals, greatly increase the soil surface, additionally stimulating peat oxidation and CO_2 emission. Overgrazing may cause peatland erosion, with peat soil loss leading to increased (offsite) CH_4 emissions from adjacent wet depressions, ponds and streams. Fertilisation of peatlands by manure from grazing animals could enhance mineralization, denitrification, and thus the N_2O flux (see also fertilization).

Urban and industrial building construction totally destroys peatlands and disrupts their role in GHG flux, especially their role as a sink of CO_2 from the atmosphere. Roads and pipelines built on peat cause the water table to rise and fall on the up-grade and down-grade sides, resulting in consequent changes to GHG emissions.
Peatlands used as a space for domestic and industrial building construction are usually totally destroyed and as such, lose their role in GHG exchange with the atmosphere. Mainly this loss concerns their role as a sink of atmospheric carbon. The impact on climate change of this use of peatlands differs and depends on the net peatland GHG flux before disturbance. Roads, railways and pipelines network cover many regions where peatlands exist and affect natural flows in peat, as material of road base course and water passing construction could keep its natural rate. As a result very often the water table rises and falls on the up-grade and down-grade sides. This causes consequent changes to GHG emissions. We assume most sensible changes on the upgrade side where over moistening and even period flooding could increase CH_4 emissions additionally supported by organic material originated from plants dead after water level rise.

Peat fires, which nowadays mostly result from human activities, could be one of the largest sources affecting the levels of CO_2 in the atmosphere. Post fire effects reflected in mineralization and fertilization of a peat soil, often a water level rise and changes in vegetation, could lead to additional smaller but more long-term changes in GHG flux.
Peatland fires ignited by lightning strikes were normal phenomena in many areas of the world (see Joosten and Clark, 2000). Today peat fire is most frequently the result of human activities (land clearing of forest land for agriculture, of natural grasslands for cattle breading, careless use of fire for

domestic purposes, while hunting and fishing, etc.) and can be treated as one of the largest sources of CO_2 in the atmosphere, as well as the most significant for GHG flux human-induced disturbance to peatlands. Peat fires are difficult to extinguish and may last for many months despite extensive rains. In cold regions, peat fires could survive even under the snow cover and heat up after the snowmelt. Peat can be burned up to a great depth. For example, during fires in 1997 and 1998 in Kalimantan, some 7500 km^2 of tropical peat-swamp forest was destroyed with a loss of surface peat of between 0.2 and 1.5 metres. Post-fire effects reflected in mineralization and fertilization of a peat soil, often a water level rise and changes in vegetation could lead to additional smaller, but longer-term changes in GHG flux.

Burning peat bogs in Indonesia are releasing massive amounts of carbon dioxide into the atmosphere, in a repeat of the environmental devastation that made headlines around the world five years ago. Tropical peat bogs, such as those beneath the forests of Indonesia, are among the planet's largest stores of carbon. They release much more CO_2 when they burn than burning trees that grow on them do. This week a team of scientists from Britain, Germany and Indonesia It was estimated that when Indonesia's forests burned in 1997, the smouldering peat beneath released as much as 2.6 billion tonnes of carbon into the air. That's equivalent to 40 per cent of global emissions from burning fossil fuels that year, and was the prime cause of the biggest annual increase in atmospheric CO_2 levels since records began more than 40 years ago.

Dam building for different purposes, using a wide range of construction, and resulting a variety of different sized dams could affect existing downstream riparian and fluvial peatlands, with subsequent changes to GHG flux rates

Numerous dams are built on rivers and streams for flood control, water supply, electricity production, shipping improvement and other purposes. Additional to the creation of a headwater reservoir, this alters the runoff regime below the dam. Allowing for differences in river valley geomorphology and climatic conditions, this could affect the water regime and hydro periods of a floodplain on various different spatial and temporal scales. Damming could also cause the timing of the flood period to shift, particularly when natural peak flow is replaced by controlled water passage. All these changes will affect riparian and fluvial peatlands if they exist there. Even though information on the subject is scarce, we can assume different reactions in terms of CO_2, CH_4 and N_2O fluxes in riparian and fluvial peatlands. Sedimentation of particulate organic matter from flood water could enhance the supply of substrate for methanogens (van Huissteden et al. 2005) and damming could suppress this influence.

Inundation of peatlands after creating water reservoirs (especially for generating hydro-electricity) could lead to significant emissions of both CO_2 and CH_4.

Creating water reservoirs for hydro-electricity and other purposes very often affects lowland peatlands in the surroundings due to flooding (totally or periodically) and the raising of ground water levels. Depending on the relief, this could affect different sized areas. Flooding would be expected to lead to an increase in CH_4 emissions. Generally CO_2 emissions may also remain relatively high (or even rise-especially during the first period after inundation when this could be caused by the rapid decomposition of young plant material (Joosten, Clark, 2002)). Emissions due to the flooding of Canadian wetlands were estimated to represent 5% of Canada's anthropogenic emissions (Roulet 2000).

Inundation and rewetting of degraded peatlands could restore their sequestration function for atmospheric CO_2 in the near future. However, to begin with, it could increase CH_4 emissions, keep relatively high CO_2 releases and have different effects on N_2O flux.

The rewetting of degraded peatlands is generally expected to decrease CO_2 and N_2O and increase CH_4 emissions. According to the goal, inundation and rewetting of degraded peatlands could restore their sequestration function of atmospheric CO_2 in the near future (Komulainen et al. 1999, Waddington, Price 2000; Petrone et al. 2001, Waddington et al. 2001, 2003; Tuittila, Laine 2004). But peatland restoration does not necessarily result in lower emission rates- especially at the beginning. In practice, peatland restoration could lead to a variety of GHG flux changes depending on the site and peat type, previous disturbance (excavation, drainage for agriculture or forestry, etc.), restoration method (inundation, damming, filling of drains, etc.), and especially the period of time passing since the measures were taken. Rewetting could increase CH_4 emissions (Marinier et al., 2004, Tuittila et al. 2004) and CO_2 release may remain continuously high (Tuittila et al. 2000, Tuittila, Laine 2004), possibly caused by rapid decomposition of young plant material, though this is probably a transient phenomenon (Joosten, Clark, 2002). Water-level fluctuations of some rewetting plots may cause a drastic increase in N_2O

emissions (Flessa, Klemisch 1997; Komulainen et al., 1999; Joosten, Clark 2002). Rewetting of drained alder forests led to increased emissions of CH_4, but to decreasing N_2O (Augustin et al. 1998).

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Chapter 8: Impacts of future climate change on peatlands

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Summary points:

1. **Climate change scenarios suggest major changes in temperature, precipitation and other phenomena that will have significant impacts on the peatland carbon store, greenhouse gas flux and biodiversity.**
2. **Global temperature rises of 1.1-6.4 ^{o}C will be higher in northern high latitudes where the greatest extent of peatlands occurs.**
3. **High latitudes are likley to experience increased precipitation while mid latitudes and some other regions may have reduced precipitation at certian times of year. All areas may be susceptible to drought due to increased variability in rainfall.**
4. **Increasing temperatures will increase peatland productivity by lengthened growing seasons but this will be moderated by enhanced moisture stress.**
5. **Decay rates in the surface of peatlands will increase as a result of rising temperatures, potentially leading to increased CH_4 and CO_2 release, but moderated by hydrological changes.**
6. **Tree lines in northern peatlands will shift poleward as a result of higher summer temperatures, and hydrological changes may result in increased forest extent on open peatlands. Both factors will contribute to reduced albedo.**
7. **Increased rainfall intensity will likely enhance peatland erosion in susceptible areas. Erosion may also be enhanced in peatlands subject to desiccation, especially where there are other pressures such as overgrazing.**
8. **Fire frequency and intensity may increase on peatlands that are subject to greater extremes of drought, although human activity is expected to remain the primary cause of fire.**
9. **Hydrological changes, combined with temperature rise, will have far-reaching effects on greenhouse gas exchange in peatlands. Drier surfaces will emit less CH_4, more N_2O and more CO_2, with the converse for wetter surfaces.**
10. **Melting permafrost will likely increase CH_4 emissions and lead to increased loss of dissolved organic carbon in river runoff.**
11. **Inundation of coastal peatlands may result in biodiversity and habitat losses, as well as increased erosion, but local impacts will be highly variable depending on land surface uplift.**
12. **The combined effect of changes in climate and resultant local changes in hydrology will have consequences for the overall distribution and ecology of plants and animals that inhabit peatlands or use peatlands as a significant part of their life cycles.**
13. **Human activities will increase peatland vulnerability to climate change in many areas. In particular, drainage, burning and over-grazing will increase losses of carbon from oxidation, fire and erosion.**

8.1 Future climate change scenarios

Current scenarios for climate changes over the 21st century suggest major changes in temperature, precipitation and some other climate phenomena that are likely to result in substantial changes to peatlands.
Any assessment of future impacts of climate change on ecosystems must be based on some assumptions concerning the most likely trajectory for future climate change. Here, we use the summarised results of the IPCC Fourth Assessment Report (IPCC, 2007) as the basis for an evaluation of future changes to global peatland systems, focusing on scenarios for climate change to the end of the 21st century. Based on IPCC data, the magnitude of climate change predicted suggests substantial impacts on the distribution, functioning and biodiversity of peatlands throughout the world. The most recent projections for climate change now have sufficient spatial resolution and adequate confidence levels to begin to suggest likely trajectories for peatlands at the regional scale. Figures 8.1 and 8.2 illustrate the key regional differences in temperature and precipitation predictions.

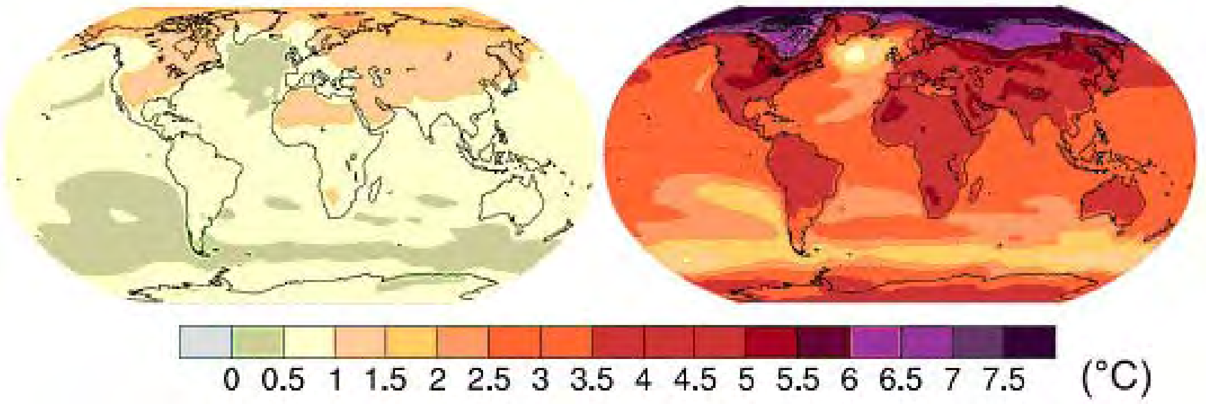

Figure 8.1: Projected changes in temperature in 2020-2029 (left) and 2090-2099 (right) compared to the period 1980-1999, based on the multimodel ensemble for the IPCC A2 emissions scenario. This scenario chosen to illustrate the spatial pattern of temperature change compared to the global average. It is striking that key peatland areas in the northern high latitudes are projected to experience some of the largest temperature changes, much greater than the global average temperature rise.

The likely[1] range for global average mean annual temperature rise by the 2090s is between 1.1 and 6.4°C, as compared with the period 1980-1999. There are significant regional variations in the relative magnitude of change. In general, temperature change will be greatest at higher latitudes over land and less marked at low latitudes over oceans. There may also be seasonal differences in the magnitude of temperature increase for some regions.
The globally averaged surface temperature is projected to increase by 1.1 to 6.4° by the period 2090-2099, as compared with 1980-1999. The range of variability in predictions reflects the a wide range of possible socio-economic scenarios and is based on a suite of different climate models. While these figures indicate global average annual changes, significant regional variability in future temperatures is expected. It is **very likely** that nearly all land areas will warm more rapidly than the global average, particularly in the northern high latitudes in winter. In northern regions of North America and central Asia, estimated land surface temperature increases are >40% above the global average. The only land surfaces where warming is less than the global average, are southern South America in winter and south and southeast Asia. The rate of global temperature rise is very likely to be more than double the rate of natural climate change experienced during the 20th century, and over the next 20 years, an increase of 0.2°C per decade is predicted for almost all emissions scenarios.

There will be changes in the frequency and magnitude of temperature extremes with fewer extremely cold days and more extremely warm days.
It is **virtually certain** that there will be warmer and more frequent hot days and nights over land areas and with fewer cold days and nights over nearly all land areas. It is **very likely** that the frequency of warm spells and heat waves will increase.

There is less confidence over predicted changes in precipitation than for temperature, but the following changes are found in most modelled scenarios. Global average precipitation will be higher but there will be significant regional variability, with some regions seeing an increase in precipitation while others may see decreases. Greater year-to-year variability is very likely over most areas where an increase in mean precipitation is projected.
Globally averaged water vapour, evaporation and precipitation are projected to increase. The magnitude of the global increase in precipitation varies between models and in line with the different socio-economic scenarios but is typically between 2 and 15% (Cubasch et al. 2001). Estimates of change at the regional scale show much greater variation with differences in the sign and magnitude of change, typically of the order of ± 5 to 20%. In addition, seasonal changes in precipitation are crucial to ecosystem impact assessment, especially for peatlands (Figure 8.2). It is **likely** that precipitation will increase in high-latitude regions during the whole year whereas in mid-latitudes projections suggest an increase in winter and decreases in some peatland regions in summer (e.g.

[1] Where they refer to IPCC (2007) report results, the following terms are emboldened and are associated with subjective estimates of likelihood based on expert judgement. Virtually certain >99%, extremely likely >95%, very likely >90, likely >66% chance)

Western Europe, Tierra del Fuego). Winter (DJF) increases in precipitation are also projected for tropical Africa and increases are noted for JJA in southern and eastern Asia. Australia, Central America, and southern Africa show consistent decreases in winter (JJA) rainfall with a **likely** increase in risk of drought. Decreased summer precipitation is **very likely** for the Mediterranean, with a **likely** increase in risk of drought.

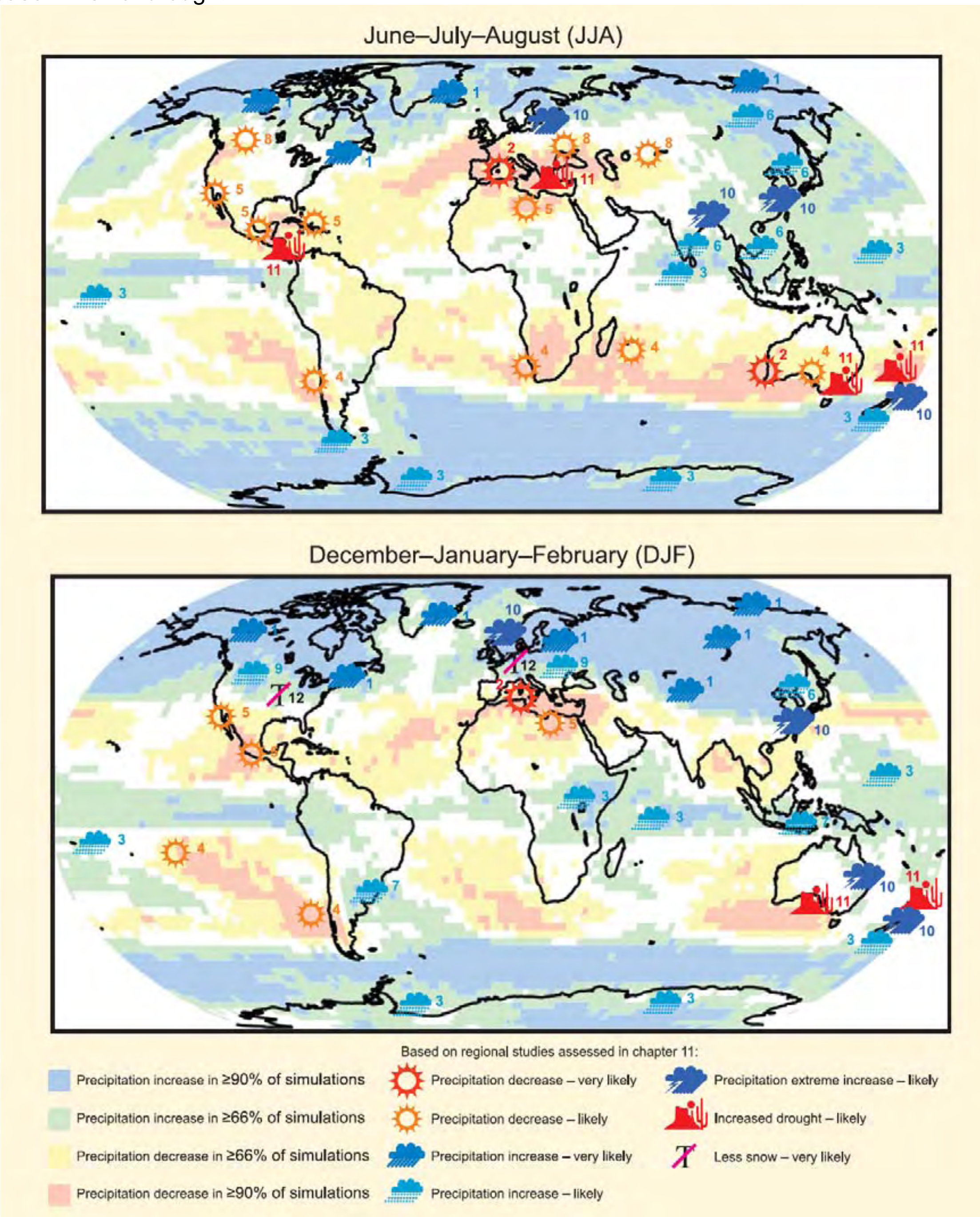

Figure 8.2: Key future changes in mean and extremes of precipitation, snow and drought. Background shading indicates level of consistency between models used (left hand side key) and symbols indicate locations of most significant regional changes. Reproduced from Christensen et al. *(2007).*

Numbers on figure show:

(1) Very likely annual mean increase in most of northern Europe and the Arctic (largest in cold season), Canada, and the North-East USA; and winter (DJF) mean increase in Northern Asia and the Tibetan Plateau.

(2) Very likely annual mean decrease in most of the Mediterranean area, and winter (JJA) decrease in southwestern Australia.

(3) Likely annual mean increase in tropical and East Africa, Northern Pacific, the northern Indian Ocean, the South Pacific (slight, mainly equatorial regions), the west of the South Island of New Zealand, Antarctica and winter (JJA) increase in Tierra del Fuego.
(4) Likely annual mean decrease in and along the southern Andes, summer (DJF) decrease in eastern French Polynesia, winter (JJA) decrease for Southern Africa and in the vicinity of Mauritius, and winter and spring decrease in southern Australia.
(5) Likely annual mean decrease in North Africa, northern Sahara, Central America (and in the vicinity of the Greater Antilles in JJA) and in South-West USA.
(6) Likely summer (JJA) mean increase in Northern Asia, East Asia, South Asia and most of Southeast Asia, and likely winter (DJF) increase in East Asia.
(7) Likely summer (DJF) mean increase in southern Southeast Asia and southeastern South America
(8) Likely summer (JJA) mean decrease in Central Asia, Central Europe and Southern Canada.
(9) Likely winter (DJF) mean increase in central Europe, and southern Canada
(10) Likely increase in extremes of daily precipitation in northern Europe, South Asia, East Asia, Australia and New Zealand.
(11) Likely increase in risk of drought in Australia and eastern New Zealand; the Mediterranean, central Europe (summer drought); in Central America (boreal spring and dry periods of the annual cycle).
(12) Very likely decrease in snow season length and likely to very likely decrease in snow depth in most of Europe and North America.

Importantly for peatlands, a general drying of the mid-continental areas during summer may occur as a result of a combination of increased temperature and evapotranspiration losses without adequate compensatory increases in precipitation, leading to increased associated risk of drought. It is **very likely** that there will be an increase in the frequency of intense precipitation events over many regions and that there will be greater year to year variability over most areas where an increase in mean precipitation is projected.

There may be no change in the amplitude and frequency of El Nino events, but future climate change will increase the risk of El Nino impacts occurring.
Confidence in projections of the future amplitude, frequency and spatial pattern of El Niño is limited by the ability of models to simulate this part of the climate system. Current projections show only small increases or no change in the amplitude of El Niño over the next 100 years. However, even with no change in El Niño, global temperature increases are likely to lead to greater extremes of drying and heavy rainfall, increasing the risk of drought and flood during El Nino events. Furthermore, current trends for surface temperatures to become more ‘El Niño like’ in the tropical Pacific (east warming more than west and an eastward shift in precipitation) are projected to continue (Meehl et al. 2007).

Changes in storms and cyclones are less certain than many other changes, but increased intensity of tropical cyclones may lead to increased and peak precipitation and wind strength in affected regions.
In higher resolution modelling studies, there is a **likely** increase in peak wind intensities and precipitation in future tropical cyclones. However, there may be fewer tropical storms overall. There may be fewer mid-latitude storms, associated with a poleward shift in storm tracks, but increased windspeeds in those regions, especially in the southern hemisphere (Meehl et al. 2007).

The total precipitation and interannual variability of the Asian summer monsoon may be increased. Precipitation may also increase (west African monsoon, Australian monsoon) or decrease (Sahel, Mexico/central America) in other monsoon areas.
The monsoon systems are important factors in seasonal water supply to many regions of the world including for wetlands and peatlands. One of the main projections is for an increase in interannual variability in precipitation during the Asian summer monsoon. Precise changes are dependent on particular emissions scenarios. Furthermore, the size and distribution of aerosol forcing complicates this picture, especially in Asia where there are likely to be major emissions of a number of aerosols. There is thus considerable uncertainty in the degree of change in these regions (Meehl et al. 2007).

There are many other extreme climate phenomena relevant to impacts on peatlands for which projections cannot yet be made.
Climate models are not capable of resolving small-scale phenomena such as thunderstorms, tornadoes, hail and lightning strikes. The latter may be particularly important in determining fire frequency on peatlands.

Northern Hemisphere snow cover and sea-ice extent are projected to decrease further.
The extent and duration of snow cover are expected to decline in association with increased temperatures. However, increased winter precipitation over northern high latitudes may produce greater depths of snow with important implications for the thermal regime at the peatland surface. Models suggest major decreases in sea-ice extent, with some models predicting disappearance of arctic summer sea ice cover by the end of the 21st century under the highest emissions scenarios (Meehl et al. 2007). Reduced snow cover (including that on peatlands) and sea-ice reduces the albedo of land and sea surfaces leading to further enhancement of atmospheric warming.

Atmospheric carbon dioxide concentrations will rise by between 90 and 250% above the pre-industrial natural levels.
In AD 1750 before significant industrial activity and anthropogenic emissions the atmospheric concentration of carbon dioxide was 280 ppm. By the end of the 21st century it is expected that this will have risen to 2-3 times this level unless there is a significant change in emissions. This is higher than at any time in the recent geological past during the evolution of the biota that currently occur in peatlands. The combined effects of raised CO_2 levels on peatland plant and microbial communities are largely unknown.

Global average sea level is projected to rise by between 18 and 59 cm due to thermal expansion of sea water and melting of land based glaciers and ice caps.
These figures are representative of the full range of socio-economic scenarios up until AD 2100. There are considerable uncertainties over the likelihood of acceleration of ice flow in both Greenland and the Antarctic. If these occur, they may increase sea levels by a further c. 20 cm. There will be substantial regional variability in sea-level rise due to local variations in the influence of ocean currents and thermal expansion. Projections for these regional variations are poor although there is some agreement on greater than average rise in the Arctic Ocean and less than average rise in the Southern Ocean. Further local variability will arise from isostatic and tectonic land movements. Short-term extremes in local sea-level may be greater if storms become more severe.

8.2 Impacts of climate change on peatlands

Climate change will have far-reaching consequences for peatlands. There will be a wide range of impacts on biodiversity and carbon cycling. Indirect effects will follow, specifically in terms of feedbacks to the climate system through greenhouse gas exchange, and impacts on other functional values of peatlands.
Preceding chapters have shown that peatlands are fundamentally linked to climate variability and conditions. Climate is a key determinant of their distribution, condition and typology (Chapters 2 and 4). Their biogeography and biodiversity are also linked to climate (chapter 5), and variability in weather and climate is a primary control on many aspects of their functioning in relation to the carbon cycle and greenhouse gas exchange (chapter 7). Peatlands can therefore be affected in many ways by projected future climate changes. Some of the reactions to climate change will result in feedbacks through changes in greenhouse gas fluxes and storage.

8.2.1 Effects of increasing temperatures

Increasing temperatures will increase peatland primary productivity in some regions due to lengthened growing seasons and increased average temperatures during the growing season. In other areas increased temperatures will reduce primary productivity due to enhanced evapotranspiration and increased moisture stress.
The length of growing seasons has already increased in much of the northern hemisphere, as indicated by earlier flowering times of herbaceous plants (Fitter et al. 2002) and earlier emergence of leaves in broadleaf tree species in mid-latitude Europe (Menzel 2000). In future, it is expected that the growing season length will increase by 20-50 days by 2100 as measured by the number of days above 5^{o}C (Figure 8.3). Whilst increased growing seasons will be experienced across many biozones, the productivity response may be greater in northern regions where peatland extent is greatest, because there will be much less drought stress to limit growth.

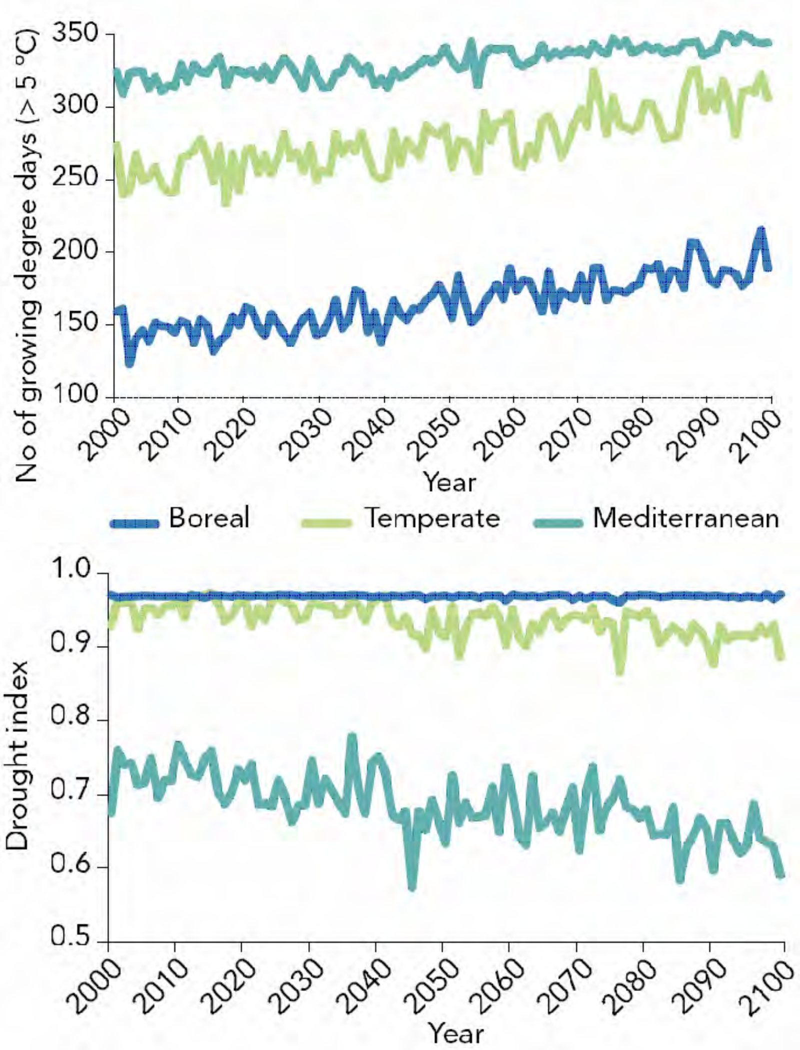

Figure 8.3: Changes in European growing season length (top) and an index of drought stress (bottom) for boreal, temperate and Mediterranean regions. In the lower graph, higher numbers indicate lower drought stress. From European Environment Agency (2004).

Increasing summer temperatures in the arctic and mountain areas will result in the northern tree line migrating to higher latitudes and altitudes in both peatland and non-peatland areas. This will decrease the albedo (reflectivity) of the surface, further enhancing warming of the atmosphere.

Tree lines (the latitudinal or altitudinal limit for tree distribution) is primarily limited by summer temperatures. The threshold for different taxa varies but trees cannot reproduce or grow after germination unless summer temperatures exceed a thermal threshold. Tree lines have varied in the past (Kullman 1999) but recent evidence suggests that northern tree lines are already experiencing northward shifts as a result of recent rises in summer temperatures (ACIA 2005). Forest has a darker surface than the open sedge and moss dominated peatland that it replaces. This leads to increased absorption of solar radiation leading to further enhancement of near-surface atmospheric warming.

Increased temperatures will generally result in enhanced decay rates and loss of carbon. However, the response of decay rates in the surface layers of peatlands to increased temperatures is complex and depends on hydrological and other conditions as well as temperature.

Microbial decay is largely driven by temperature. Increased temperatures result in greater microbial activity and therefore high decay rates. However, the microbial population can only take advantage of raise temperatures if a range of other conditions are also met. A primary constraint is the nature of the substrate. Different plant materials have different intrinsic decay rates. For example, different species of *Sphagnum* decay at different rates, despite their superficially similar appearance (Johnson and Damman 1991). In addition, susceptibility to decay declines over time so that after intense or prolonged decay, the rate of decay slows because only the most recalcitrant material remains. Overlain on effect of substrate is the effect of moisture availability, oxygen content (influencing the dominance of aerobic or anaerobic decay) and the impact of periods of freezing.

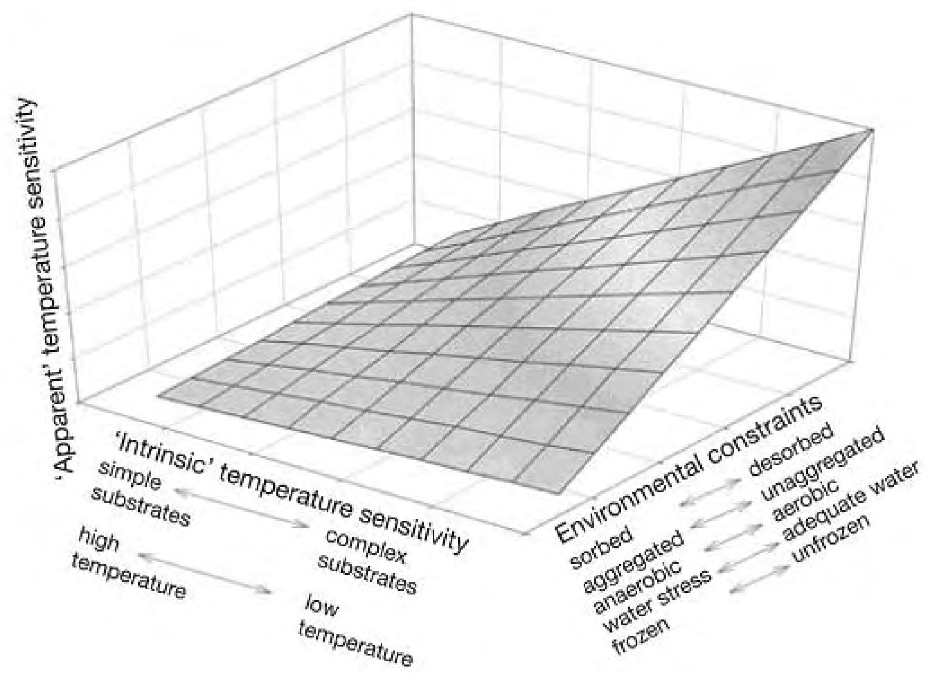

Figure 8.4 Factors affecting the sensitivity of organic matter to decay. The intrinsic temperature sensitivity is affected by the temperature and the substrate. The environmental constraints dampen the sensitivity to decay because of water availability, soil structure, freezing. Although the response is shown as linear, the relationship between these factors and decay is often non-linear. Reproduced from Davidson and Janssens (2006).

Increasing temperatures will generally accelerate the microbial processes responsible for CO_2, CH_4 and N_2O emission from peatlands.

Impacts of increased temperatures on greenhouse gas emissions are difficult to distinguish from the complex effects of simultaneously changing hydrological regimes (see below). Ecosystem respiration has been observed to depend on soil temperature variations but not on water-table level in a continental bog with inherently deep water levels (Lafleur et al. 2005). In high latitudes, the interannual variations in CO_2 balance are mostly due to variations during the snow-free period, but the respiration efflux during the winter time is also a significant component in the annual balance. Snow melt timing appears to be the most important single factor in determining the balance, and consequently a longer growing seasons may give rise to higher rates of C sequestration instead of being a threat (Aurela et al. 2004). Methanogenesis is also highly temperature dependent and where water table remain high, CH_4 emissions will increase (Whalen 2005).

8.2.2 Effects of precipitation changes

Responses to precipitation changes will be highly variable depending on local and regional factors.

Projections of future precipitation show a much more varied response than those for temperature (Section 8.1). There are large regional variations in both the sign and magnitude of change, as well as in seasonality. In regions of increased year-round precipitation, peatlands may become wetter, but only if the increase in precipitation is not counterbalanced by increased evapotranspiration losses from increased temperatures. In northern high latitudes where precipitation is projected to increase year-round, peatland water tables may rise, particularly in spring, when combined with the effects of snowmelt. However, increases in summer precipitation may not compensate for higher temperatures and more very hot days, so there will be periods of increased dryness and drought during the summer. In other regions such as the northern mid-latitudes, a decrease in summer precipitation will cause a fall in water tables during the summer, exacerbated by increased temperatures. Increased winter precipitation in the mid-latitude regions will not provide any compensation for summer drought, as excess moisture will be lost through runoff. In tropical peatland regions, projections suggest decreases (Amazon basin), little change (e.g. tropical west Africa) or increases (e.g. Indonesia) in precipitation. At any particular location, the hydrological impacts will be mediated through topography and mire type. For example, ombrotrophic bogs may be more susceptible to reduced precipitation than fens, where groundwater flow can buffer some of the temporary drought periods.

Increased frequency of heavy rainfall and greater intensity of precipitation may cause enhanced erosion of susceptible peatlands.
Peatlands that are susceptible to erosion through exposure of bare peat surfaces lose large amounts of peat during heavy rainfall events and also through aeolian processes (Evans and Warburton 2007). In temperate and high latitude peatland regions, potential threats are wetter winters and increased drought frequency in summer. In particular the predicted increased intensity of rainfall events may exacerbate erosion of peat in areas already disrupted by the effects of grazing, fire and pollution, and also in regions subject to other future climate-related damage. Increased severity and frequency of droughts would cause desiccation of the peat surface, resulting in structural damage such as cracking and increased susceptibility to erosion during wetter periods (Evans et al. 2006). In extreme cases, very wet conditions have led to mass movement of peat in 'peat-slides' both during intense rainfall events in summer and more prolonged wet conditions during winter (Warburton et al. 2004). More extreme hydrological conditions in future may increase the likelihood of such events.

Where increased flooding of peatlands occurs due to higher frequency and magnitude of heavy precipitation events, there may be increased CH_4 release.
The relationship between water table depth and methane emissions is well known, with higher water tables causing increased methane emissions (Chapter 7). Increased flooding of peatlands in northern high latitudes is expected as a result of thermokarst thawing (see below), but temporary increases in surface wetness may also occur in association with winter and spring increases in precipitation, when temperature and evapotranspiration increases are inadequate to counteract this. The extent of this effect is unknown but it is likely to be minor compared to other changes given relatively low temperatures at these times of year.

8.2.3 Hydrological changes

The hydrological regime is the principal factor controlling ecosystem processes in peatlands. Any changes in water balance should be expected to have far-reaching effects on biogeochemical cycles, productivity and community composition.
Increased evapotranspiration, altered precipitation, and increased frequency of extreme events (e.g. droughts, intense rainfall) are likely to result in generally lower water table depths during the growing season (Roulet et al. 1993, Waddington et al. 1998). However, there may be exceptions to this generalisation in some regions owing to permafrost thawing, and large regional increases in summer precipitation. Impacts of the hydrological changes on ecosystems are mediated through autotrophic (vegetation) and heterotrophic (microbial) communities, and the responses of these components to changing ecohydrology may alter the carbon dynamics of peatlands significantly, as observed during unusually warm and dry summers (Carroll and Crill 1997, Alm et al. 1999).

A combination of increased temperatures and changes in precipitation regime will determine the future hydrological status of peatlands. The direction and magnitude of surface wetness (moisture, water table) changes will vary between regions. Changes in precipitation will be particularly important in determining the local and regional changes in hydrology.
The regional complexity of projected future climate change (see Section 8.1) will lead to a regionally and seasonally variable hydrological response in peatlands. However, some generalisations can be made concerning some likely regional responses. First, the strongest effects are likely to be in areas where peat growth is currently marginal. The southern limit of extensive ombrotrophic peatlands across North America and Europe coincides with the region of reduced summer precipitation and is likely to see major falls in water table throughout the summer. Areas where peatlands are limited to water-collecting sites (e.g. southern Europe, southern Africa) are also likely to contract as these generally lie within regions of reduced future water availability. Where peat growth is currently limited by productivity as a result of low temperatures at high latitudes and altitudes, there may be an expansion of peatlands in topographically suitable locations. This effect is mainly likely to be felt in the high arctic.

Changes in water table levels will affect carbon dioxide and methane exchange of peatlands. The global net effect of these changes on 'global warming potential' of peatlands is not known with any accuracy, but it is likely that the carbon sink function of existing peatlands will be reduced overall. This may be compensated for in part by expansion of some peatland areas.
In general, falling water tables in many peatlands will result in lower CH_4 emissions but higher CO_2 emissions. In areas where peat accumulation is already marginal, this will result in a gradual loss of

the peat by oxidation. Conversely, in some regions (notably thawing permafrost areas), increased flooding will increase CH_4 release but may also increase CO_2 drawdown from increased productivity. The relationship between these two effects in terms of their feedback to radiative forcing is complex, particularly as CH_4 has a much shorter lifetime but a more powerful instantaneous impact as a greenhouse gas. Taking these effects into account, northern peatlands currently appear to have net negative effect on radiative forcing (Frolking et al. 2006). Predictions of future changes are more uncertain.

There is a lot of information on the impacts of water level draw-down based on research in forestry drained peatlands (see Chapter 7). These studies may be cautiously used to represent the climate change impact because the effect of drainage on ecosystem structure and functioning is rather similar to that predicted after drying caused by climate change in the northern latitudes. (Bubier et al. 2003) concluded that a drier climate with lowered water table levels may reduce the CO_2 sink function of peatlands for some growth forms and increase it for others, suggesting that ecosystem carbon and climate models should account for differences in growth form responses to climate change. It also implies that plant functional types respond on short timescales to changes in moisture, and that the transition from sedges to shrubs could occur rapidly in peatlands under a drier and warmer climate. Similar observations were presented by (Strack et al. 2006). Fairly small changes in hydrological regimes have been observed to influence the CH_4 emission rates from peatlands, with dry conditions producing clearly lower emission rates (Strack et al. 2004, Bubier et al. 2005). Exceptionally dry summers have been reported to temporarily convert pristine peatlands from sinks to sources of CO_2 (Alm et al. 1999, Lafleur et al. 2003). Increased frequency of droughts may also affect the CO_2 exchange through changes in vegetation (Laiho et al. 2003) and litter quality (Laiho et al. 2004). Alm et al. (1999) suggested that the ratio between moist and dry summers should be at least 4:1 to retain a positive C balance and 5:1 to retain the average long-term accumulation rate of 25 g C m^{-2} yr^{-1}. If peatlands are generally drier over large areas of the northern mid to high latitudes, the net effect will be to reduce the CO_2 sink function. However, it might be expected that this will be compensated for in part by increased productivity in some northern peatlands and expansion of peatland areas at the northern limit for peat formation.

Lower water tables will increase the rate of release of N_2O.
Increased aerobic decay in the surface layers of peatlands following water table draw-down is accompanied by increased mineralization of organic compounds including nitrogen (N). Although levels of N are often low in ombrotrophic systems they are much higher in minerotrophic peatlands such as floodplain fens. The N is released as nitrous oxide (N_2O), another powerful greenhouse gas (see Chapter 7). Although N_2O emissions from natural peatlands are generally low, peatland drainage is known to increase N_2O release (Minkkinen et al. 2002, Alm et al. 2007). This suggests that N_2O emissions may increase with water table drawdown and increased temperatures in the future. However, such increases may be relatively small on natural peatlands and the largest emissions of N_2O are likely to be from peatlands used for agriculture, where addition of nitrogen fertiisers stimulates further release on N_2O, even when cropping is abandoned (Maljanen et al. 2007).

Hydrological change will result in major changes in microbial, plant and animal communities.
Hydrology is the most important influence on peatland communities and many species occur in restricted hydrological niches (Chapter 5). Changes in hydrology will inevitably result in changes to the distribution of individual species and communities. This will happen at various scales. Small scale microtopographical patterning provides a range of microhabitats for peatland taxa. Changes in the balance between pools, lawns and hummocks on individual sites will lead to a change in the plant and animal communities present. Specialist taxa tend to be more dependent upon the wetter microhabitats in shallow pools and hollows and loss of these habitats may lead to local extinctions of aquatic and semi-aquatic taxa. The ability of these taxa to migrate to more suitable locations may be limited in regions where peatlands have become fragmented.

Hydrological changes will result in expansion of forest cover on drying peatlands and reduction in tree cover on peatlands with increased surface wetness.
Tree cover on peatlands is usually less dense than on surrounding mineral soils. Typically, trees become smaller and less dense towards the wetter, central area of a peatland (see Chapter 2). Tree establishment and growth is limited by high water tables that keep the rooting zone saturated. Thus when water tables fall, tree cover becomes more extensive and existing trees may be able to grow (Choi et al. 2007). Once trees become established on drier areas of peatlands, they may further enhance transpiration, providing a positive feedback and further drying of the peatland surface

beneath the trees and adjacent areas. Increases in tree cover may also add to the decrease in surface albedo arising from the more general changes in tree lines, and reduced snow extent (see above). The situation may be more complex in tropical peatlands; although some trees will increase growth if water tables fall, others may suffer increased mortality in drought years (Nishimua et al. 2007).

Fire frequency may increase on peatlands that are subject to greater extremes of drought.
Natural fires occur throughout many parts of the world, including areas where peatlands are common. Although peatlands are some of the wettest locations in the landscape, in dry summer seasons, surface layers of peat and plant litter provide a highly combustible mixture for fire to start in or to spread to from surrounding upland. More prolonged, more severe or more frequent periods of drought may occur in some peatland regions (see above). Even in some areas where rainfall is projected to increase, there may be more severe short periods of drought if rainfall variability increases. There are many causes of peatland fires, principally related to human activities such as clearance for agriculture, improvement of grazing conditions or accidental causes. Lightning strikes may be a cause of natural fires in some circumstances but the occurrence of natural fires is limited when compared with those started by people. It is unknown if natural fires frequency will increase significantly in the future, because predictions of lightning strike frequency cannot be predicted with any certainty. However, any increase in droughts will make uncontrolled natural or anthropogenic fires more likely and will lead to increased intensity of burn. In particular, lowered water tables may make loss of upper peat layers more likely during peatland fires and the areas burnt may be more extensive.

In areas where peatlands become drier, extreme events such as drought and storm rainfall will increase erosion by water and wind.
Peatlands in some parts of the world such as Central Asia, are likely to become much drier in future. Desiccation of peat surfaces is likely to make them more susceptible to erosion. Exposure of bare peat surfaces and disturbance from activities such as grazing will result in further destabilization of the peat structure. During droughts, the upper peat layers can be eroded by wind and the surface becomes deflated. During intense rainfall peat can also be moved downslope and lost in runoff. There have been relatively few studies on losses of peat from desiccation and wind-blow but measurements of peat loss from exposed peat surfaces in England show that these losses are important and are likely to increase under future climate scenarios (Foulds and Warburton 2007).

In regions strongly affected by drying during ENSO years, the frequency of drought is likely to increase due to the background increase in temperature and changes in precipitation.
Several peatland regions are currently affected by ENSO; during these events peatlands in southeast Asia and to a lesser extent, the Amazon basin, experience much drier than normal conditions. Although ENSO may not increase in frequency or intensity, the frequency and impacts of drought will\increase in ENSO years because of background climate changes against which they occur. This will enhance fire frequency and intensity especially where in drained areas and in areas where people light fires for clearance and agriculture. During dry years, the Amazon Basin already acts as a carbon source (Tian et al. 1998), partly as a result of carbon loss from peat and organic soils. The number of such years is likely to increase in the future.

Changes in river runoff regime may change flood periods and other characteristics of the hydrological regime of riverine and lacustrine peatlands.
The changing balance between precipitation and evapotranspiration will result in alterations to river flows and their seasonal variability. These changes will affect peatlands in floodplains and lake margins through more dynamic flood regimes from increased intensity of rainfall and droughts. Increases in precipitation and earlier snowmelt in the northern high latitudes will lead to earlier and increased runoff of rivers discharging into the Arctic ocean. The total increase in runoff is expected to be in the region of 10-20% depending on the climate scenario used, with much greater percentage increases during winter (Walsh et al. 2005). This will affect the salinity, ice cover and circulation of the Arctic ocean.

8.2.4 Changes in permafrost and snow cover

Melting of permafrost may stimulate CH_4 emissions from wet peatland sites and peat decomposition and CO_2 release from dry peatland sites such as palsas.
Rapid permafrost melting has been reported to increase formation of thermokarst ponds and fen-bog vegetation with rapid peat accumulation through natural successional processes of terrestrialization (Payette et al. 2004). Both increased temperatures and increased snow depth (and therefore

insulation) may be a cause of permafrost degradation. Contrary to current expectations, the melting of permafrost caused by recent climate change did not transform the peatland to a carbon-source ecosystem as rapid terrestrialization exacerbated carbon-sink conditions (Payette et al. 2004). However, the developing fen vegetation may develop into an efficient producer of methane emissions (Wickland et al. 2006), which may counteract the climatic impact of CO_2 sequestration.

Reduction in permafrost extent in peatlands may lead to greater release of dissolved organic carbon in river runoff.
Permafrost prevents or reduces decay of organic matter in northern high latitude peatlands. Where decay occurs, carbon is lost as CO_2 and also in dissolved organic carbon (DOC) in runoff. Much higher DOC concentrations have been found in catchments with permafrost-free peatlands than in those dominated by permafrost. Furthermore, there is a correlation between the extent of peatland cover and DOC concentrations in runoff in permafrost free catchments. On this basis it is predicted that losses of DOC in rivers draining the West Siberian region may increase by 29-46% by the year 2100 (Frey and Smith 2005).

Increasing temperatures are likely to lead to a large reduction in the occurrence of palsa mires.
Palsa mires occupy a zone in the high northern latitudes (Chapter 2). They are tightly constrained climatically suggesting they are particularly sensitive to climate change (Parviainen and Luoto 2007). Recent observations suggest that palsas are already declining in some areas (Zuidhoff 2002). In Fennoscandia, an increase in mean annual temperature of $4^{o}C$ would result in the loss of all palsa mires from this region. Significant reductions in the extent of palsas are likely to occur within the next 30 years and conditions suitable for palsa formation may have disappeared completely by the end of the century. Structural collapse of these relatively dry peatlands will lead to flooding accompanied by increased CH_4 release, and loss of habitat for characteristic bird species (Fronzek et al. 2006).

Increased snow depth during winter may affect the thermal regime of northern peatlands, exacerbating the effect of increased temperatures on permafrost melt.
Snow cover is important in insulating the peat surface during winter. Deeper snow cover leads to less intense freezing and reduced permafrost formation. For example, at the southern margins of palsa distribution, palsas only occur in particular areas where wind action is sufficient to reduce snow cover (Nihlen 2000). Increased winter precipitation may thus increase the insulation of peatlands in northern regions, leading to further reduction in permafrost formation.

8.2.5 Sea level rise

The potential extent and severity of sea-level rise impacts on peatlands has not been assessed in any systematic way. However, many peatland areas occur in coastal areas, often as fens linked to saline wetlands (e.g. saltmarshes) or in low lying floodplains. Assessment of local and regional impacts depends on a good understanding of local land-surface movement as well as global average sea-level change. Only general speculative observations are made here.

Inundation of coastal peatlands may result in biodiversity and habitat losses with conversion of freshwater peatlands to saline marshes. A rise in base level may allow spread of new peatlands inland if land is made available for this.
Gradual sea-level rise will result in a shoreward displacement of the boundary between saltmarsh and freshwater wetland as the height of highest astronomical tide rises. Complete inundation of saltmarsh and coastal peatlands may even result in destabilisation of the peat matrix, release of CH_4. However, due to the rise in base level, areas further inland may be increasingly susceptible to flooding and freshwater peatlands could spread inland if the topography is suitable and land use policy allows for this. Gradual sea-level rise is likely to take place only in areas that are isostatically and tectonically stable. In areas that are undergoing land-uplift at the same or greater rate than sea-level rise, sea-level change will not present a threat. For example in northern Sweden and Finland, land uplift rates of 8-10 mm pa are far in excess of the likely 3-5 mm pa of global sea-level rise. The only impact in such areas will be a slowing of new mire development on uplifted land surfaces.

In low-lying peatland areas, intrusion of saline water into aquifers may give rise of increased salinity and changes in the ecology and functioning of the system
Floodplain fens and some other peatlands close to the coast are partly dependent on groundwater for their water supply. Aquifers close to the coast are often linked to the sea and it is possible that over

time the groundwaters will become more saline. This will change the growing conditions for plants and the geochemical conditions at the surface.

There is an increased risk of erosion of coastal peatlands in areas experiencing sea-level rise.
In some locations where peatlands meet the coast, rising sea-levels may result in incremental erosion of peat, with loss of particulate organic carbon (POC) directly into the ocean. The extent of physical erosion depends upon the wave conditions and rate of sea-level rise. Gradually increasing sea-level in a relatively sheltered wave environment is more likely to result in succession to saltmarsh and marine sediment accumulation rather than loss of peat.

8.2.6 Carbon dioxide fertilisation

Increased atmospheric CO_2 concentrations may lead to enhanced growth of some peatland plants, but any response to elevated CO_2 will be moderated by competitive interactions, and other chemical and hydrological conditions.
It is well known that plant growth can be stimulated by increases in CO_2 concentrations. Peatland plants are likely to respond to increased CO_2 concentration in a similar way but experiments on peatland plants and plant communities have suggested a more complex response. Laboratory studies have shown that *Sphagnum* increases growth rate in response to rising CO_2 (Silvola 1985), but moisture may be a limiting factor and growth rates decline markedly once moisture levels are below optimal levels (Silvola and Aaltonen 1984). Likewise, community responses may be different to individual species effects. For example, competitive interactions between *Sphagnum* and vascular plants resulted in a decrease in moss cover and an increase in vascular plant cover under raised CO_2 (Fenner et al. 2007). Furthermore, in this same experiment, an increase in productivity of the plant community was more than compensated for by increased decay rates as a result of higher decomposition rates in the plant litter. Microbial communities are also affected by changing CO_2 levels (Mitchell et al. 2003), probably controlled indirectly through shifts in plant communities and the nature of plant detritus. The impacts of future elevated CO_2 concentrations on peatland carbon sequestration are therefore rather difficult to assess with any certainty, but it seems unlikely that there will be major gains from CO_2 fertilisation because of other effects on plant communities and growth rates.

8.2.7 Other impacts of climate change on peatlands

Non-linear responses to climate change may lead to ‘surprise’ changes in peatland systems. Some of these changes will be irreversible.
Past climate change has led to sudden changes in peatlands in the past (e.g. the so called Grenzhorizont in Europe, chapter 4). Sometimes peat accumulation is interrupted for several hundred years by events such as fires, floods or long term droughts. Even if climate change proceeds as a gradual process, it is unlikely that the peatland response will be smooth and monotonic. Many of the processes expected to change in response to climate change are likely to have thresholds past which changes are sudden and perhaps irreversible. For example, drying of peat surfaces can lead to cracking and desiccation to the point where rewetting the peat is extremely difficult. Once this initial severe drying has occurred, slow oxidation and peat removal by physical erosion occurs and is largely irreversible until the layer of altered peat has been lost.

There will be longer term (100-1000 years) impacts as a result of climate changes taking place this century, due to altered successional processes, and the slow response of some processes.
Many peatland ecosystems have a high level of ‘self-determination’ due to successional processes. This means that once an initial change has taken place, it may lead to other (different) changes in the future. For example, in the case of permafrost collapse, the initial formation of pools may be succeeded by a phase of pool-infill and renewed peat growth, which leads to a very different peatland system over hundreds to thousands of years.

Climate change will interact with anthropogenic disturbance. Some of the impacts of climate change will exacerbate the impacts of activities such as drainage, grazing, burning and logging. Human disturbance to peatlands often makes them much more vulnerable to climate change impacts.
Climate change is only one of many factors affecting peatlands. In chapter 3, the range of uses of peat and peatlands was described. In most cases, human impacts on peatlands will increase their

vulnerability to climate change. In a number of situations, climate change will simply act to accelerate existing degradation of damaged peatlands. For example, the occurrence of fires in Indonesian peatlands is largely due to drainage, logging and fire setting (Page et al. 2002, chapter 3), but the frequency and severity of fires is increased by changes in the length and severity of droughts. Likewise, over-grazing of vegetation in central Asia has already led to erosion and loss of peat; these impacts are likely to be accelerated by future climate change.

The species that are most vulnerable to climate change are those specifically adapted to peatland conditions, endemics, species at their geographical limits (latitudinal and altitudinal), and those with a disjunct distribution.
Some peatland species have clear relationships with climate variables (see chapter 4). Those species are expected to disappear in areas where significant climate shifts occur. Changes will occur first at the geographical limits of these species. Where taxa have a disjunct distribution, perhaps due to fragmentation of the peatland habitat, these taxa are more likely to become locally extinct rather than migrating in concert with climate changes. Endemic and rare peatland taxa are the most threatened species. The Siberian Crane (*Grus leucogeranus*) is a threatened species dependent upon suitable nesting areas in tundra regions of Yakutia. This and other migratory birds dependent on open breeding areas may suffer reduced populations as the area north of the treeline becomes smaller, with expansion of northern forest areas.

The microclimate of peatlands may present opportunities for ecological adaptation to climate change. Peatlands may provide non-peatland taxa with temporary refugia as they retreat to higher latitudes and altitudes.
Peatlands are often isolated within anthropogenically modified landscapes. They provide some of the last ‘wild’ landscapes in such areas and thus are a refuge for specialist peatland taxa which may be more widespread elsewhere, but also for other wildlife which can survive in a peatland or around its margins. Peatlands may therefore act as ‘stepping stones’ for migration of more adaptable species that could not survive in agricultural or other strongly modified landscapes. In particular, they will be refuges for birds and mammals sensitive to disturbance.

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Chapter 9: Management of peatlands for biodiversity and climate change

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Summary points:

1. **Many stakeholders do not yet understand or appreciate the complex nature and important multiple benefits of peatlands for biodiversity, climate change and community livelihood.**
2. **The current management of peatlands is generally not sustainable and has a major negative impact on biodiversity and climate change.**
3. **A wise use approach is needed to integrate protection and sustainable use to safeguard the peatland benefits from increasing pressure from people and changing climate.**
4. **Policy frameworks often fail to recognize the special eco-hydrological characteristics of peatlands which are so important for their sustainable management.**
5. **Strict protection of intact peatlands is critical for the conservation of biodiversity and will maintain ecosystem function and carbon stores/sequestration.**
6. **Simple changes in peatland management can both improve the sustainability of land use and reduce its impacts on biodiversity and climate change.**
7. **Restoration of peatlands can be a cost-effective way to generate immediate benefits for biodiversity, climate change by reducing peatland subsidence, oxidation and fires.**
8. **New production techniques such as wet agriculture should be developed and promoted to generate production benefits from peatlands without negatively impacting their environmental functions.**
9. **Peatland management should be integrated into land use and socio-economic development planning by taking multi-stakeholder, ecosystem, river basin and landscape approach.**
10. **Peatland issues should be better incorporated into international (CBD, Ramsar, UNFCCC, CCD etc.) and regional policy-making processes.**
11. **Lack of awareness and capacity, poverty, inequity, and perverse incentives are important root causes of peatland degradation and should be addressed in a comprehensive manner.**
12. **In developing countries and countries in transition where poverty may be a root cause of unsustainable peatland resource exploitation, development may be the only way to create opportunities for peatland conservation.**
13. **The huge, but only recently recognized, CO2 emissions from tropical peatland deforestation and degradation, represents one of the single largest but also most concentrated sources of greenhouse gas emissions and provides as such also a major opportunity to reduce current global greenhouse gas emissions.**
14. **The option for local communities to provide services to the emerging carbon market in terms of peat swamp forest conservation and restoration represents a major opportunity for linking climate change mitigation to poverty reduction.**
15. **Evidence has been accumulating that in many cases, natural peatland habitats generate marked economic benefits, which exceed those obtained from habitat conversion.**
16. **There is an urgent need to create an enabling policy environment for innovative mechanisms such as the emerging market in Verified Emission Reductions from peatlands and forests, to provide the necessary basis for long-term commitments from all stakeholders.**

9.1 Protection and rehabilitation of peatlands

9.1.1 Protection of remaining peatlands

Peatlands cover less than 3% of the land surface but store more carbon than the vegetation of all the world's forests combined. They are also critical for maintenance of biodiversity and other functions.
Peatlands are the largest terrestrial carbon store (see Table 9.1) and play a key role in global climate regulation (see Chapter 6 and 7). However, the sustainable maintenance of remaining peatlands could

yield benefits beyond those of CO_2 sequestration. It could help to maintain biodiversity, preserve water retention capacity in catchments, enhance local economic development as well as contribute towards the sustainability of rural livelihoods. To reap these multiple benefits however, a balance needs to be found between the protection and utilisation of peatland areas.

Table 9.1: How peatlands compare with other carbon stores

Storage/area characteristic	Statistic
Area covered by peatlands	400 million ha (Joosten 2002)
Carbon stored by peatlands	550-650 billion tonnes (IPCC, 2001)
Carbon stored by all global plant biomass	694 billion tonnes
Carbon stored in the world's soils (including peat)	1,600 billion tonnes
Carbon in the atmosphere	700 billion tonnes (Gorham 1995)

The protection of remaining peatlands is one of the most important and cost effective management strategies for minimizing CO_2 emissions.
Peatland degradation is becoming one of the most important global sources of CO2 emissions from the Land Use and Land Use Change (LULUCF) sector. Emissions from tropical peatlands in Se Asia alone (covering 0.2% of the world's land area) are estimated to be approximately 2 billion tonnes of carbon dioxide per year (or about 7% of global fossil fuel emissions) (Hooijer et al 2006). Given the high density of carbon in peatlands (see Chapter 6) degradation of peatlands leads to disproportionately high carbon emissions. Since emissions from peatlands are almost always as a result of human induced degradation – protection of peatlands may be a very important management strategy. Peatland protection is also very cost effective compared to other ways for GHG emission mitigation.

Intact peatlands which have not been drained or disturbed should be strictly protected for biodiversity, carbon sequestration & carbon storage.
Intact peatlands with natural vegetation cover and hydrology still have the potential to sequest carbon. Once they are degraded however, they may lose this function. In regions like Europe up to 90% of peatlands have been cleared, drained or otherwise sufficiently degraded to disrupt the relationship between the peat, plants and water (see chapter 2). For example, many of Europe's transition states experienced widespread peatland degradation due to drainage and conversion to agricultural land during 1950s-1980s. Almost all natural peatlands in the Netherlands and Poland have been destroyed/significantly modified /degraded, while Switzerland and Germany have only small areas of relatively intact peatlands remaining.

Given the limited proportion of intact peatlands in many regions (except possibly for parts of Russia, Canada and the USA) it is important that most if not all remaining intact peatlands are formally protected in strictly protected areas such as conservation areas, catchment or floodplain reserves, nature reserves and national parks. In many regions the proportion of the original peatland area which is included in totally protected areas is still low. Even in countries where peatlands are included in protected areas, the proportion of peatland that is intact is still low. For example in Finland – a country with supposedly 30% of its peatlands intact – many of the peatlands in totally protected areas have been degraded by drainage, groundwater extraction and other factors – either within or adjacent to the protected areas boundary (IMCG 2006). Many peatlands form large hydrological units covering an area of hundreds or thousands of ha. In Indonesia for example – some peatland hydrological units cover about 1 million ha. Although part of the hydrological unit may be included in a protected area development or clearance of other portions of the peatland outside the protected area may affect the integrity of the overall peatland. Protection need not mean the complete exclusion of human activities from peatland areas. Instead, it involves reaching acceptable trade-offs between the prevention of degradation and the continuation of wise use.

9.1.2 Fire prevention and control

Fires in peatland are one of the largest global point sources of greenhouse gas emissions (Turetsky et al., 2002).
Predictions of increased drought incidence and severity in many peatland regions due to climate change are likely to lead to an increase in carbon losses due to fire. This may change many peatlands from being net sinks for atmospheric carbon into net sources (Hogg et al., 1992). Reducing the incidence of peatland fires could aid carbon sequestration and storage (Garnett et al., 2000), which could contribute towards meeting emission reduction targets under the Kyoto Protocol – particularly on peatlands that are used for agricultural purposes. Globally the largest scale peatland fires occur in

Southeast Asia and Russia. The fires in Southeast Asia are linked with the large-scale development of agriculture and settlement schemes in the 1980s and 1990s and large scale development of oil palm and pulpwood plantations over the past 10 years. The estimated emissions from fires in Se Asia over the past 10 years are between 14-40 billion tones (Hooijer et al 2006) – see Figure 9.1.

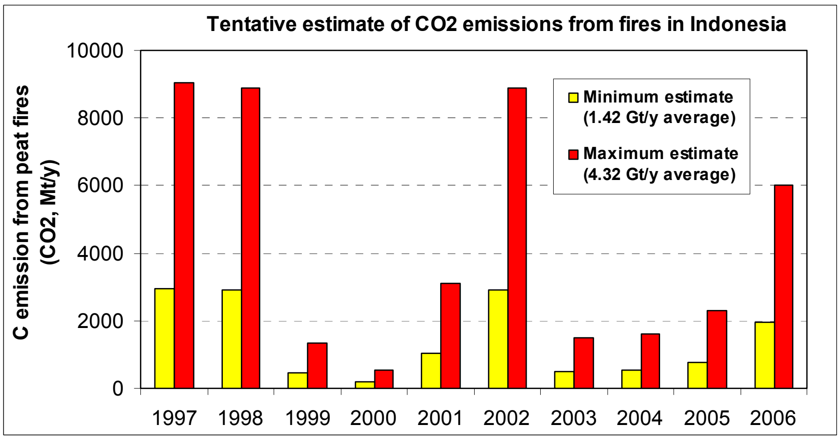

Figure 9.1: Tentative estimates of CO2 emissions from peatland fires in Indonesia 1997-2006 (source Hooijer 2006).

The high intensity of wildfires can destroy both the surface vegetation and litter as well as burning the peat layers. This can lead to a variety of environmental problems.

Peat fires lead to the direct loss of peat and peatland vegetation, as well as massive CO_2 emissions. They may burn deeply into the peat layer in cases where the groundwater table is lowered due to drainage or severe drought. During the period 1997-98 fires in peatlands in Se Asia burnt more than 2 million ha (Taconi 2004) Peat fires have been recorded in Indonesia burning 5 m below the surface, making them impossible to extinguish without bringing the water table back to the surface (Parish 2002). Peat fires in Manitoba in Canada burned for more than 10 years in the 1980s, smouldering under thick snow cover and emerging again in the spring (Gerry Hood pers com. 1997). The fires in Southeast Asia and other regions such as Russia generate large amounts of smoke, which alone can cause several major social, economic and environmental problems. Transboundary Smoke haze – primarily from peatland fires – regularly affecting five countries in SE Asia has been identified as the most serious environmental problems in the region and has stimulated the establishment of the ASEAN Peatland Management Initiative (see Box 9.1).

Figure 9.2: Erosion feature resulting from an historic accidental fire (left) and an eroded area resulting from the same fire that has been reseeded and treated with heather brash (right) on Bleaklow, Peak District National Park, UK (photos: M. Reed)

Fires in upland peat in the UK have been shown to lead to increased soil erosion (and associated water colouration and siltation), as well as long-term loss of vegetation cover (also associated with biodiversity loss). This can then make the soil surface susceptible to wind and water erosion and increase freeze-thaw action. Indeed, the onset of some major erosion episodes in UK peatlands have been linked to historic wild or human-induced fires (Mackay and Tallis, 1996). Figure 9.2 shows erosion from a historic wildfire on Bleaklow, in the Peak District National Park, UK. Increases in pH are likely following wildfires (Allen, 1964; Stevenson et al., 1996), and differences in pH have been noted between different burning regimes on blanket bog. In addition to this, ash has been shown to influence soil microbial populations (MacDonald, 2000). This ash is rich in nutrients, raising soil fertility for the first two years after a burn, benefiting regeneration (Hansen, 1969). However, leaching may be significant particularly after autumn burns, and losses of phosphorus and nitrogen may inhibit regeneration. Kinako and Gimingham (1980) suggested that it may take 75 years for the phosphorus losses from one burn to be replaced. Wildfires also have a range of economic impacts, including the cost of labour or equipment costs, as well as lost revenues from former land uses. For example, smoke primarily from peat fires in Indonesia enveloped 5 countries in Southeast Asia in smoke for up to six months in 1997/98 and led to estimated economic losses of US$10 Billion from direct damage to forests as well as impact on helth, tourism etc from the smoke haze (ADB and Bapenas 1998).

Investment in peatland fire prevention and control may be one of the most cost effective measures to reduce global GHG emissions. Since fire in peatlands may release very large amounts of greenhouse gases (over 2000 tones of CO2/ha for a severe fire in tropical peatlands), but fires can be often prevented through better water management and well as enhanced vigilance and fire control measures – investment on fire prevention and control can be considered a cost-effective way to reduce GHG emissions.

Fire in peatlands in the UK. Although many peatlands are naturally forested, many were cleared – in some areas as early as the mid-late Holocene. Particularly in the UK, much of this cleared area has been burned regularly since the 19th century to manage *Calluna vulgaris* (Heather) and sometimes grass, for sheep, deer and grouse production (Figure 2). The aim was to maintain cover of *C. vulgaris* (and grass for grazers) along with suitable nesting habitat for grouse. The impacts of fire on vegetation and erosion have been raised by some studies but land owners argue that managed burning reduces the likelihood of more intense accidental fires that are more likely to burn into the peat. Nevertheless measures are starting to be put in place in the UK to reduce the level of fire in peatland areas such as use of cutting rather than managed burning.

In some areas, fire risk can be compounded by an expansion in public access to peatlands for recreation. However, access management, for example through provision of surfaced paths and limited access points, can significantly reduce the area of land used by the public. Educational programmes have also been successful. For example the Moors for the Future partnership project "Moor Care Initiative" has provided leaflets and cigarette butt pouches to Peak District National Park visitors in the UK (Moors for the Future, 2006). Restoration initiatives that raise the water table may go some way towards reducing fire risk, though this is unlikely to reduce risk during prolonged droughts, when most wildfires occur. Alternatively, increased provision of firebreaks and fire-fighting resources in remote areas (e.g. water tanks and beaters) can minimize the impacts of wildfires (Reed et al., 2005). Such precautions may be particularly relevant around areas popular with visitors and tourists.

In Indonesia, fire prevention activities have involved the blocking of abandoned agricultural or forestry drainage channels, revegetation of degraded sites, fire awareness campaigns with local communities, provision of equipment and training for local volunteer fire prevention and control teams. No single measure will be effective in reducing the risk of fire in peatlands. However, wise preparation and the use of several mechanisms together can help protect peatlands from fires and in doing so, reduce emissions of GHGs.

In some cases fire has been used as a management tool by local communities and land owners in the management of peat soils.
For example in Wasur National Park near Merauke and Pulau Kimaam in SE Papua province In Indonesia local communities have for 1000s of years used fire to control vegetation growth to stimulate good grazing conditions for hunting of wildlife such as wallabies and deer in shallow peatlands (Silvius 1989 a and b).

Figure 9.3: Managed burning on UK peatlands for grouse and sheep management showing a fire being lit (top left), burning (top right), being put out (bottom left) and after the fire (bottom right) (photos: M. Reed)

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9.1.3 Rehabilitation of degraded peatlands

Rehabilitation of degraded peatlands can maintain carbon storage and other values.
In addition to totally protecting undisturbed or intact peatlands, it is also important to rehabilitate those peatlands which have to some extent been degraded. In some regions large proportions of the peatlands have been degraded by draining, in order to make the land suitable for forestry and agriculture. These degraded peatlands may represent the majority of peatlands in a given country. It is therefore advantageous to rehabilitate them so that they can continue to provide ecological services, such as carbon storage and water supply, as well as support biodiversity. The rehabilitation strategy will have to be developed in accordance with the specific situation at the various sites but may include the restoration of hydrology and natural vegetation or other management interventions. Initial work has indicated that provided the peatland has not been too degraded it will be possible to restore natural vegetation and stimulate further carbon sequestration.

Peatland restoration strategy for The Central European Peatland Project (CEPP)
The CEPP was started in 2000 to ensure that the natural heritage of peatlands would not be lost – indeed that it would be enhanced – during the challenging period of economic transition, stabilisation and growth. Its role was to assist eight participating countries to implement the Ramsar Convention through the development of a Strategy and Action Plan for Peatlands in Central Europe (Bragg et al. 2001). The strategy included an overview of peatland distribution and identified peatlands with a high biodiversity value. In addition, the strategy aimed to increase awareness about values and function of peatlands, including their significance with respect to carbon storage and flood attenuation and water purification.

Figure 9.4: Peatland restoration. Blocking of a canal in degraded peat swamp forests in Central Kalimantan, by local communities using manual traditional techniques (photo: Marcel Silvius, Wetlands International).

Restoration of degraded peatlands is often complex, expensive and takes significant time.
Changes to peat soil pH, nutrient status and structure are often irreversible. While vegetation recovery may be rapid in the immediate vicinity of a blocked channel, it may take decades for the effects to be felt down-stream. In addition to this, the scale of the problem is enormous. There are hundreds of thousands of kilometres of land drains and gullies throughout the world and the costs associated with blocking ditches and gullies on this scale are enormous. There are cheaper alternatives (for example using wood or heather bales on heather moorland), but in a system with a moderate slope, it is necessary to block channels every five to fifty metres for successful restoration. Costs can be minimised by the use of landscape models that can produce a programme of prioritised blocking locations. Priority locations are selected because they optimise the effects of blocking on the water table with the minimum number of dams (although channels on steeper slopes are likely to be eroding fastest) (Holden et al., 2006b).

Restoration or rewetting of peatlands reduces fire risk, CO_2 emissions and may generate benefits for biodiversity and local communities.
Peatland restoration through peatland rewetting and revegetation has been shown to significantly reduce fire risk and encourage regrowth of natural vegetation. In Belarus, restoration of Yelnia peatland through the blocking of over 20 major drains has led to elimination of regular fires. Large scale peatland rewetting is now underway in Belarus (see Box).

In Indonesia a pilot project to block drains in the abandoned mega rice project area in Central Kalimantan has demonstrated significant recovery of peatlands after the drains were blocked and water levels were maintained within about 40-50cm of the soil surface (Nyoman et al 2005, 2006) See

Figure 9.4. In the three years since blocking fires have been stopped, subsidence reduced and vegetation has recovered

In the UK blocking of drains has led to a significant reduction in dissolved organic carbon and particulate organic carbon in runoff water. This in turn has reduced downstream water treatment costs and reduced health risks (Holden et al 2006).

Restoring degraded peatland in Belarus. In 1950, there was 2 939 000 ha of natural peatland in Belarus. As a result of large-scale drainage programmes between the 1950s-1990, more than 54% of peatlands were drained for peat extraction and agriculture (First National Report on Implementation of the UNCCD in Belarus, 2002). This had a number of negative environmental impacts, including: biodiversity loss, local microclimatic changes, the drying of small streams and rivers, increased nutrient runoff and subsequent eutrophication of water bodies, shrinking fen areas, increased soil erosion, fires, huge net carbon dioxide emissions. In autumn 2004, UNDP and the GEF agreed to fund a US$3 million project to restore the degraded areas. 17 sites across Belarus with a total area of 42 000ha are in the process of being restored.

Restoration of peatlands can generate important new sources of sustainable livelihood as well as generating biodiversity and climate change benefits.
The restoration of peatlands can create new sources of sustainable livelihood as well as providing benefits for climate change and biodiversity. For example, restored peatlands in southern Thailand are part of an award winning ecotourism centre. In Indonesia in 2005, abandoned logging channels were blocked by local communities in one village in Central Kalimantan. This was initially undertaken to prevent peat fires but they subsequently were able to harvest over 2 tonnes of fish from the blocked channels which functioned like fish ponds. Restoration of tropical peatlands are essential to re-establish peat swamp forests and the associated timber and non-timber forest products such as rattan which sustain the local communities. In eastern Germany a new industry for sustainable biomass energy has been created following establishment of reed farms and wet alder forests on formerly degraded peatlands.

9.2 Modification of peatland management strategies

9.2.1 Improved water management

Improved water management is a fundamental step to support the sustainable management of peatlands.
Water is probably the most fundamental component of a peatland with most peatlands comprising 90% water. The extent, nature and depth of the peat are frequently a function of water extent and depth. Drainage thus has one of the most important and long-lasting impacts on peatlands. Drainage of temperate and tropical peatlands, to lower the water table by 1m, leads to a CO_2 emission of approximately 30 and 100 tonnes of CO_2/ha/year respectively (Wosten 2002, 2006). Drainage of water also leads to increased vulnerability to fire which is one of the most significant courses of peat degradation and GHG emissions. Fire does not normally occur continuously but when burning does take place, it may lead to the emission of up to 4000 tonnes of CO_2/Ha in the tropics and 2000 tonnes of CO_2/ha in temperate regions. Peatland fires are becoming more frequent in some regions such as South East Asia. This is generally a result of accelerated rates of land clearance as well as the large-scale drainage of peatlands. These extensive (more than 2 million ha of peatlands were burnt in the past 10 years) and persistent (many fires burnt for between 1-3 months) fires lead to large CO_2 emissions. Primarily as a result of persistent peatland fires Indonesia is now considered to have the 3rd highest CO_2 emission globally (Wetlands International, 2006)

While drainage has greatly improved the ability to farm peatlands, drainage leads to loss and subsidence of peat soils. A balance between drainage and conservation is needed in order to protect peatland soils.
Drainage has greatly improved the ability to farm peatlands. However it can lead to significant subsidence (dependent on drainage period and depth and temperature) of peat soils and large amounts of CO_2 being lost to the atmosphere – therefore contributing to greenhouse gas emissions. The excessive drainage of peatlands can also cause the shrinkage or loss of wetland area as well as the reduction of water levels in adjacent wetlands and mineral soils.

As peat subsides, the depth of the fertile topsoil also decreases. This means that further drainage, cultivation and pasture renewal are needed to maintain productivity, therefore increasing the cost to farmers. When managed properly, peat is a valuable, highly productive resource. To be able to farm on peat soils over the long-term, farmers must find a balance between keeping the water table low enough for production, but high enough to minimise peat loss and CO_2 emissions.

It is possible to use peatlands for agriculture without draining by using species of plant which require little or no drainage such as sago palm, or yams in the tropics or maintaining natural peatland sedges for hay production in the temperate regions.

Appropriate management is critical to maintain water pollution sink, flood control and water supply functions of peatlands.

Although it is not ecologically appropriate for peatlands to be deliberately used for water purification in heavily-polluted areas, in some regions they may be found downstream of polluting operations. As a result, they play an important role in the removal of pollutants from streams. For example in the river systems of the highveld in South Africa, peatlands downstream of industries and mines are important for filtering out and temporarily storing pollutants. These include waste matter such as uranium from gold mining operations. As a result subsequesnt degradation of peat or extraction of peat for use in horticulture may lead to significant pollution (Wetfix, 2006).

Drainage and gully erosion are major causes of peatland degradation and associated loss of carbon storage, biodiversity and ecosystem services around the world.

Gully erosion occurs when vegetation cover is lost, for example due to inappropriate burning or overgrazing. The effects of gullying and drainage are similar, though the problems associated with gullying tend to be more severe. Drainage of lowland peats in much of the western world took place mainly in the 19^{th} century to improve land for agriculture by lowering the water table. But in the 1960s and 1970s, upland drainage became more common. In Europe, this was carried out primarily to improve land for grazing and grouse production, and remove hazards for stock (Ratcliffe and Oswald, 1988). However, there is little evidence that these aims were met (Stewart and Lance, 1983).

Restoration of Reitvlei Peatland for water storage and water quality improvement

Rietvlei peatland near Pretoria, South Africa was recently rehabilitated by a team of more than 50 local people through the Working for Wetlands programme. This degraded peatland that that was drained in the 1960s for peat mining, dryland cropping and irrigation purposes. In addition, poorly timed annual burning of reeds led to peat fires, while upstream, urban townships and industrial areas caused serious pollution. Rietvlei supplies nearly 20% of the water supply of South Africa's Capital City Pretoria and is owned by the municipality. Since the remaining 80% of water has to be bought in from elsewhere, there is a strong economic imperative to manage water wisely and to restore the former diffuse flow of water through the wetland, which previously provided natural water purification services free of charge. Working for Wetlands has enabled the diversion of water from the central drainage canal out to the edges of the peatland, re-flooding previously dried-out areas. Small gabions (rock-filled wire baskets) placed at 30 m intervals allow the water to back up and then overflow into the desiccated wetland. This has enhanced the water storage as well as the water quality (Working for Wetlands, South Africa, 2006).

By lowering the water table, drainage and gullying increase the air-filled porosity of the peat, leading to shrinkage, cracking and subsidence, increasing decomposition rates and altering microbial processes. These changes have important implications for peatland hydrology, water quality and ecology. For example, aerobic decomposition in drained peats enhances the mineralization of nutrients, leading to significant losses of carbon, phosphorus, nitrogen and sulphur, which may affect soil fertility. Water flow paths through and over peatland soils are altered (Holden et al., 2006a), leading to a complex hydrological response (Holden et al., 2004). This includes increased loss of particulate and dissolved organic carbon in stream water, and has implications for the carbon balance of drained or eroding peatlands. As outlined earlier, as the water table is lowered, peatland fires become more of a risk. This is a problem that can only get worse under projected climate change scenarios and is likely to further contribute to carbon losses in a possible positive feedback loop (Hogg et al., 1992). Thus, drainage and gullying can lead to environmental problems of increased fire risk, increased incidence and severity of down-stream flooding, carbon loss, water colouration, changes in peatland ecology including reduced biodiversity, and the sedimentation of reservoirs and fish spawning beds. Careful management is therefore paramount in ensuring the adequate control of these practices.

Blocking drains and gullies in peatlands can stem carbon losses, and sequester and store carbon as channels re-vegetate.

Blocking drains and gullies in peatlands can reduce subsidence and fires and hence stem carbon losses, and sequester and store carbon as degraded peat and channels re-vegetate. Nyoman et al 2006, Worrall et al. (2003). Blocking ditches and gullies is undertaken to raise the water table to its former level and to re-wet the peat. If this does not lead to natural re-vegetation, reseeding or planting of wetland species can be used (Price, 1997; Evans et al., 2005; Nyoman et al 2005, Chen Ke Lin et al 2006).

9.2.2 Modification of agricultural practices

Conversion of natural peatlands for agriculture is one of the main root causes of the loss of peatland biodiversity and functions.
In terms of area of peatland affected the most extensive impacts on natural peatlands have come from the drainage and utilization of peatlands for agriculture purposes.. Agricultural use generally involves the drainage of peat (by 30cm-1.5 m) and the replacement of the vegetation with agricultural species such as potatoes, cabbage, vegetables, oil palm, maize, buckwheat, or pineapples. The selection of species depends on the climatic and ecological situation and water levels (degree of drainage) as well as the macro-economic and agriculture commodity situation. The uses of the area have varied over time.

Agriculture in peatlands in Heilongjiang, China
The natural wetlands in China and the biodiversity they support are under a constant threat of degradation, mostly associated with human development pressures, such as drainage, over-use of water resources, conversion to agricultural uses, unsustainable harvesting and resource use. Sanjiang Plain in Heilongjiang Province is a vast, low lying alluvial floodplain of about five million hectares in the apex of the Heilongjiang and Wusuli rivers which is dominated by peatlands. In its former natural state, the floodplain ecosystem consisted of a mosaic of sedge and reed marshes, wet grass meadows, ox-bow lakes, riverine scrub and wooded hummocks of birch and poplar. During the past 30-40 years more than 90% of the peatlands of the Sanjiang Plains have been converted to agriculture – primarily for rice and soya bean production. This has led to significant degradation of the natural resource, loss of locally and globally important biodiversity, significant GHG emissions, and lessening of flood mitigation capacity.

Agriculture on peatlands which involves drainage will lead to the loss of peatlands and its associated functions and cannot be classified as sustainable peatland management.
Any agriculture practice which involves drainage of peatland will lead to loss of the peat layer through oxidation, compaction and erosion. In addition the natural processes which lead to peat formation stop so that no further growth of the peat layer takes place. As a result, drained peatlands will continually subside and eventually (provided that the drainage continues) the entire peat layer will be lost exposing the underlying mineral soil.

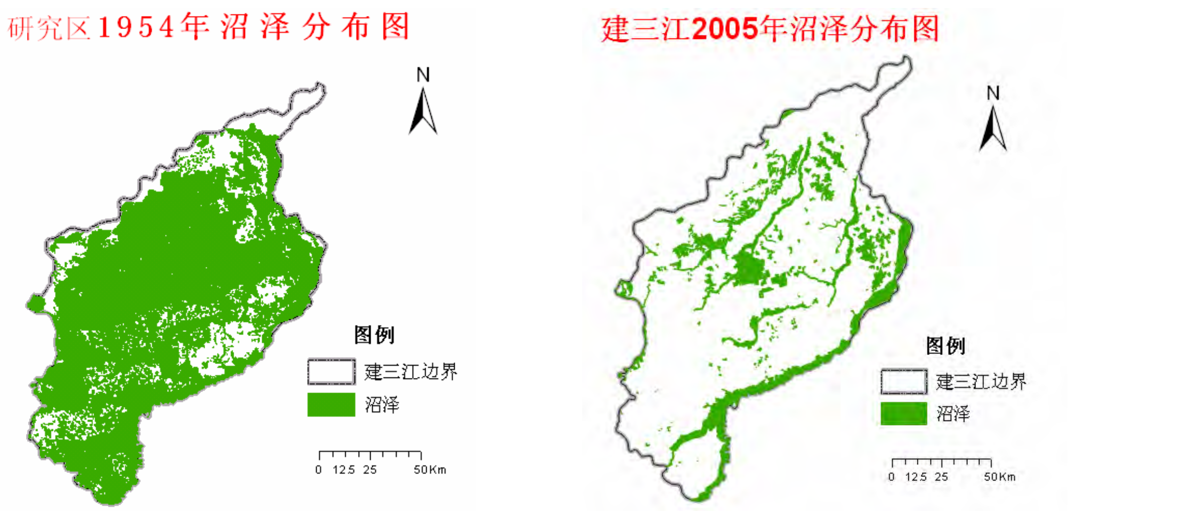

Figure 9.5: Change in area of remaining natural wetlands/vegetation in Sanjiang plan 1954-2005 (Source UNDP-GEF 2007).

Agricultural drainage in peatland areas is frequently badly designed and leads to peat degradation as well as reduced agriculture yields.
In many places the agricultural drainage system may be too deep and with inadequate water management systems which leads to overdrainage of the peatlands. In Malaysia for example most of the drains developed in peatland areas were based on the designs of drains in mineral soils. As a result they lowered the water levels too much and led to rapid subsidence. The peatlands of West Johor in Peninsular Malaysia were drained in the 1970 with funding and technical guidance from the

World Bank. As a result of poor design and lack of water management structures there was overdrainage and severe subsidence – leading to the collapse of most of the infrastructure, failure of the agriculture projects, flooding of coastal towns, acidification of water supply and other problems.

Agricultural production techniques that maintain /increase peatland carbon store need to be developed and promoted.
Agricultural or agroforestry activities that do not involve the drainage of peatlands, that maintain natural water levels and that can maintain or increase the natural carbon stores should be developed or promoted, in contrast to those techniques that drain or lead to the loss of carbon storage. Examples include sphagnum farming, reeds, alder, jelutong, sago, and hay making.

9.2.3 Modification of livestock management on peatlands

In many parts of the world, grazing-induced erosion is a major cause of peatland degradation.
Erosion induced by overgrazing is a major cause of peatland degradation in many parts of the world (Evans, 2005), and in some areas this is expanding rapidly. Peatlands generally cannot sustain high stocking densities. *Calluna vulgaris* (Heather) which grows on upland peatlands in the UK is only of high value as a winter feed for hill sheep when grasses have died back, and only grows when grazing is below 2 sheep ha^{-1}. In Europe, in response to headage payments from the Common Agricultural Policy in the 1970s and 1980s, stocking levels increased above this level on many moorlands (e.g. 29% of UK moors were stocked above this level in 1977 and by 1987 this had increased to 71%). However grazing densities of over 0.55 sheep ha^{-1} can instigate erosion in some areas (Rawes and Hobbs, 1979). Similar problems have been described in Ruoergai Peatland in China (Chen, 2006 – see Box) and the lesthoto Highlands in Southern Africa.

Management of the Peatlands of the Ruoergai Plateau, China
The Ruoergai peatlands cover an area of about 500,000 ha at an altitude of 3400-3900m in the eastern edge of the Qinghai-Tibetan Plateau. The peatlands are found in a shallow basin surrounded by hills and mountains, and comprise part of the headwaters of the Yellow River. The peatlands are of global importance for conservation of biodiversity as well as playing important roles in water storage and supply, and carbon storage. The peatlands are also important for the livelihood of local communities especially nomadic Tibetan herders.

In late 60's-early 70', as part of a national agricultural expansion scheme about 300 km of drainage canals were dug by more than 50,000 workers to drain many of the peatlands to allow increased access by grazing animals. With rapid economic development and population increasing, the overgrazing severely damages large piece of grasslands in this area. The population of livestock has increased from past 400,000 to 1 million now. The peatlands degraded as the canals are still used to drain the water and the grass could not regenerate and the desertification expands year by year. The local government has realized that the overgrazing leads to peatland degradation and some measures have been taken to reduce the economic loss of the local communities. Some parts of peatlands are strictly protected for hay production for livestock in winter. Local herders are encouraged to reduce the numbers of the livestock by increasing their quality or breeding in captivity to encourage rangeland regeneration. More than 50km of drains have now been blocked in a pilot project to retain water resources and restore peatlands

Grazing can have a major impact on peatland vegetation dynamics which can affect carbon storage as well as biodiversity.
A number of studies have examined the effect of grazing on peatland vegetation dynamics. Grazing has a profound effect on species composition (favouring grazing-tolerant species such as tussock grasses), and depending on its intensity, can reduce competitive vigour and potentially kill plants. Overgrazing has been blamed for the loss of heather moorland in some peatland areas in the UK (Shaw et al., 1996). Grazing can have both positive and negative effects on seed dispersal. Browsing can prevent tree establishment and prevent seed production, but consumption of seeds and dispersal in dung can cause some species to spread (Thompson et al., 1995). Since the mid-1990s, there have been calls for a large reduction in sheep numbers on peatlands in the UK (Thompson et al., 1995). This trend is likely to continue as the EU's Common Agricultural Policy shifts from headage to area-based payments, reducing the incentive to overstock and taking a more environmentally-considerate approach. This highlights the importance of policy considerations when managing peatlands, as they can both encourage and discourage appropriate management and wise use.

Reduction and removal of grazing from peatlands can stop degradation and lead to recovery of peatlands, but other measures may be needed to restore peatland functions and vegetation.
The effect of reducing and removing grazing from peatland has been investigated in a number of exclosure studies. Rapid recovery only occurs in the total absence of grazing (Marrs and Welch, 1991), but a combination of herbivores using the land (including grazers and browsers) at different intensities and times of year, is likely to optimise biodiversity (IEEP, 2004).

Some low intensity management of livestock may locally enhance biodiversity.
Grazing and the cutting of hay in shallowly drained meadows reduce the competition between grasses and other plants and also create possibilities for pioneer species to colonise the area. Lightly grazed areas also may have a mosaic of microhabitats with differential relief and nutrient status. As a result, peatland grazing meadows that have been managed for long periods in a traditional low intensity manner for hay cutting or light grazing may support a higher diversity of plants (including many rare or restricted species). Following the abandonment of grazing meadow management in Eastern Europe in the 1980s and 1990s following political and economic changes, the quality and natural diversity of these grazing meadows declined.

Guidelines issued by the New Zealand government to livestock farmers operating in peatland areas.
1. Avoid deep drainage
Deep drains in peat cause over-drainage and rapid subsidence of peat soils. As the peat dries it shrinks and cracks, making soils difficult to re-wet. Rainwater flows down into the subsoil through cracks in the peat. When peat dries it becomes waxy and cannot reabsorb water easily. By keeping drains shallow, you will help keep the water table high enough to protect your peat.
2. Maintain the water table over summer/dry periods
Keeping the water table high in drier periods is important for pasture growth and maintaining soil quality, for both peat and mineral soils. This can be achieved by putting weirs or stop gates in your drains. Water table management can be difficult to achieve on an individual farm. This is because ground water is a resource that spans property boundaries. It may be useful for you to get together with your neighbours to discuss summer water-table management. Better water-table management will minimise shrinkage, allowing you to extend summer grass growth and farm your peat soils profitably for longer.
3. Fence drains and spray weeds
Controlling weeds and fencing drains to exclude stock reduces maintenance costs associated with machine cleaning drains. Weeds should be sprayed in summer (January/February) before they seed. Note that glyphosate (‘RoundUp’) is the only herbicide approved for use over water. Less machine cleaning of drains saves money. It also reduces impacts on water quality and the risk of drain deepening. Drains only need to be cleaned if their ability to function has been reduced by silt or weed growth. Most silt in drains comes from stock damaging the drain banks. By fencing off your drains you will greatly reduce the need to machine clean them. A single electric wire is usually enough on most dairy farms.
4. Do not deepen drains during maintenance
Continual deepening leads to over-drainage, and makes pasture less productive. Keep your drains shallower and protect your valuable pasture.
Source: Government of New Zealand (http://www.ew.govt.nz/enviroinfo/land/management/peat.htm

9.2.4 Modification of forestry practices

Management or rehabilitation of natural forest on peatlands is an important management strategy.
Peatlands in many regions of the world (e.g. the Boreal zone, Africa, South-east Asia and North and South America), are naturally forested. They therefore need to be managed

Clear felling, over extraction and high impact logging techniques in forested peatlands are a major cause of peatland degradation, leading to a loss of biodiversity and reduction in carbon storage.
Clear felling and over extraction of trees in forested peatlands may lead to changes in peatland water balance, as well as degradation and loss of biodiversity. In tropical peat swamp forests, large-scale harvesting leads to the drying of surface peat layers and increases the chances of fire. In addition, the open conditions are often not suitable for the growth of most peat swamp forest species, leading to the

development of secondary forests dominated by a limited number of pioneer species. High impact log extraction techniques include the use of heavy excavators. These compact peat, and alter the drainage of peatlands prior to logging in order to facilitate access. Logs may also be extracted via drainage canals. Such logging techniques have been shown to significantly reduce the chances of natural regeneration and the drainage leads to significant subsidence (Danced, 2003)

Figure 9.6 Jelutung (Dyera sp), an indigenous latex producing peat swamp forest species is being planted along the banks of blocked drainage channels in Central Kalimantan (picture: Marcel Silvius, Wetlands International)

Forest resources from peatlands which are naturally forested can be sustainably harvested using low impact logging/extraction techniques, while maintaining biodiversity and carbon storage.
Resources can be sustainably harvested however, using low impact logging/extraction techniques, while also maintaining biodiversity and carbon storage. For example, in Southeast Asia, peat swamp forest has been logged in many places using the so-called “kuda-Kuda” system, where trees are hauled along skid tracks by manual labour or winches to railway lines placed on logs laying on the peat surface. The rail systems do not involve any drainage of the system and so do not induce subsidence or other problems. In Europe extraction routes over deep peats can be covered with a layer of logging residues (known as “slash roads”). Research in the UK has shown that peat soils under slash roads exhibit only minor changes despite high levels of traffic, and can improve vehicle traction (Wood et al., 2003). In boreal peatlands in Canada, trees are harvested in winter when the peatland is frozen. This results in little or no impact from the log extraction activities.

Afforestation of naturally unforested peatlands can have important negative effects on peatlands.
The afforestation and associated drainage of naturally un-forested peatlands can cause significant changes in hydrology and ecology, leading to a reduction in water quantity and quality, loss of biodiversity and reduced carbon storage. Although forest managers now attempt to maintain or increase biodiversity through careful planting design (Anderson, 2001), there is a still an increasing area of peatland that is being commercially afforested in some countries. For example in the UK, nine percent of upland moors have been afforested (Cannel et al., 1993), mostly in the form of commercial coniferous plantations, and this area is growing. In Finland and Russia the area of afforested peatlands is very large, covering millions of ha of peatlands. In Indonesia large scale tree plantations have been developed in peatlands to supply pulp and paper mills. These plantations are currently in Sumatra and cover an area of about 800,000 ha. The main tree planted is *Acacia crassicarpa* which is not an indigenous peat swamp forest tree species. The peatlands are thus drained to a depth of 0.8-1.5 metres. Although the trees are relatively fast growing and achieve canopy closure in one year, they are harvested on a 4-5 year cycle which leads to regular clearance and opening up of the land. High levels of peat subsidence have led to significant management problems which are now being assessed (see Box 9.9).

Afforestation of peatlands is often associated with drainage and fertilizer application which together lead to major ecological changes.
Drainage ditches lower the water table, while the trees, whose roots reach far deeper into the soil profile than the natural vegetation can cause the water table to lower even further. Compression and

shrinkage can lead to subsidence and cracking of the peat surface (Shotbolt et al., 1998; Anderson et al., 2000). This alters conditions for ground layer plants, and reduces the availability of the fresh-water habitats that characterize many peatlands (e.g. blanket bogs). Fertiliser application can change the species composition of ground layer plants by altering the nutrient availability and pH. Also, as the trees grow, they can change the microclimate for the ground layer plants. This in turn can lead to further changes in species composition. Combined with changes in the soil, these alterations can cause an increased prevalence of earthworms, slugs, moths and beetles (Makulec, 1991), while spiders and wasps may become less abundant (Coulson, 1990). Birds that favour open ground are gradually replaced by forest birds. These impacts can be felt far beyond the forested area, as bird communities may be affected up to a kilometer from the edge of the plantation (Moss et al., 1996). Although tree felling can cause the water table to rise again, the soil structure (and consequently the drainage) may have been irreversibly altered by the forest. Given the unique biodiversity value of many peatland habitats, these types of environmental changes may be associated with the loss of rare and endangered species, and are therefore cause for concern.

Acacia plantations on peat in Indonesia
Peatlands in Indonesia have, over the last 20 years been developed as large scale Acacia plantations for pulp for paper production. Plantations covering hundred thousands of hectares have been developed in Riau and South Sumatra provinces, particularly by Riau Andalan Pulp and Paper (RAPP, APRIL) and Asian Pulp and Paper (APP, Sinar Mas, Indah Kiat). Plantations by APRIL were until recently quite deeply drained by 1-1.5 m – but recently the company has altered its water management strategies by raising water levels to decrease subsidence and greenhouse gas emissions from the peat. In some areas the plantations have been established in already deforested or heavily degraded peatlands, whereas in other areas they were developed in peat swamp forests. A recent area of contention between both companies mentioned as well as conservation organizations is the Kampar peninsula in Riau province, which is the largest remaining relatively undisturbed peat swamp forest area in Sumatra. Options for win-win scenarios are being considered by APRIL and where the pulp plantations could be developed in a narrow ring around the peninsula and the peat swamp forests further inland would be protected for their biodiversity values as well as their essential water management functions for the plantations (preventing floods and droughts). The company's management capacity could be used for monitoring as well as for fire control.

Although trees sequester and store carbon in their biomass, the changes that take place in the soil after peatlands are afforested lead to significant carbon loss.
Peatlands are significant carbon stores – for example, peatlands are the UK's largest carbon store, holding more than all the forests of the UK, France and Germany combined (Worrall et al., 2003). Although trees sequester and store carbon in their biomass, the changes that take place in the soil after peatlands are afforested lead to significant carbon loss (Cannell et al., 1993). Research has indicated that peatland afforestation can result in a net release of carbon dioxide into the atmosphere (Holden, 2005). Fertilization of peat soil also leads to significant emissions of N2O which has a global warming potential 310 times higher than that of CO_2.

Improvement of management measures for forest plantations on peatlands can reduce losses of biodiversity) and GHG emissions while at the same time reducing risks for production.

9.2.5 Modification of Peat extraction

Peat extraction operations can affect biodiversity and impact GHG emissions both directly and in adjacent areas
The extraction of peat for use in energy generation or horticulture is one of the significant uses of peatlands worldwide although the area used is much less than for forestry and agriculture purposes. Extraction of the peat normally involves the clearance of surface vegetation, drainage of the peat and extraction using machinery. The extracted peat is then stockpiled before transportation and utilisation. The clearance of the vegetation directly impacts the biodiversity while the drainage and extraction of the peat often leads to changes in the hydrology of adjacent areas which can affect GHG flux.

Use of peat as a substrate for horticulture is a significant source of peatland degradation and carbon emissions. This problem can be reduced through the careful site selection and development of appropriate alternative growing media.

The extraction of peat for horticulture has led to significant impacts on conservation sites in some countries and led to long term conflicts. This has in turn stimulated consumer boycotts of horticultural peat in some countries. In response the peat industry has developed codes of practice which ensure that mining is focused on those sites with little conservation value. These include abandoned agricultural land. In addition, in some countries the peat industry has actively developed post-mining restoration techniques and also introduced sphagnum farming methods. Development of alternatives to use of peat for horticulture such as compost or coco-peat (from processed coconut husk fibres) is also underway.

9.3 Integrated management of peatlands

Unisectoral planning of peatland management/use is one of the root causes of peatland degradation and so peatlands must be planned and managed in an integrated manner.

In most countries there is significant conflict between different user groups or economic sectors such as forestry, agriculture, water supply, industry etc and also between government, private sector and local communities over the development and management priorities and strategies for peatlands. These contrasting interests lead to conflicting decisions and ad-hoc planning. This is further complicated by the fact that large peatlands are single hydrological units which may cover up to one million ha – which may be subdivided by different administrative boundaries and land use zones. Since each part of the peatland is interconnected, drainage or vegetation clearance on one site will have an impact on other portions of the same zone. It is therefore imperative that each hydrological unit is addressed as a single entity for the purposes of development planning and management and care should be taken that activities approved for different parts of the peatland are compatible.

Peatland and livelihood in Indonesia

A number of people in the sub-village of Muara Puning, Central Kalimnatan, have built long ponds in the peatland (length: 10-50m; width 1.5-3 m; depth: 1-2m), see pictures below. The ponds, known locally as *beje*, trap wild fish when the river overflows during the rainy season (October – February). The fish are left in the pond for several months, and are harvested (while at the same time scooping out sediment) throughout the dry season, from July to September.

In the same area channels up to 10km long were dug into the peat swamp forest by local community, members to facilitate logging activities. During the dry season these lead to drying of the peatland and increase the susceptibility to fire. The Climate change Forest and Peatland (CCFPI) Project worked in 2003-2005 with local communities to blocking the ditches to reduce fires and associated GHG emissions. The blocking of the channels resulted in the formation of a number of ponds which are similar to *beje* ponds. A total of at least 16 species of fish (including *Chana sp., Clarias sp., Anabas testidineus, Trichogaster sp.,* among others) were recorded in the blocked sections of the ditches with a total weight of almost 2 tonnes were harvested by the local community in 2004. This acted as a strong incentive for other community members to block other abandoned channels in the peatlands

Peatlands across the world are managed by a number of different people and groups, each with different aims, values and goals.

However, it is increasingly recognized that local communities are critical stakeholders within peatland management systems. Community-based approaches can be used to raise awareness about climate change and the key role that appropriate peatland management can play in terms of carbon storage and sequestration. Local involvement can also promote sustainable management and avoid conflicts, as different stakeholders familiarize themselves with the views of others (Mathews, 1994). Vitally, local participation in peatland management can also achieve social goals; contributing towards poverty alleviation, increased livelihood sustainability and social empowerment (Middendorf and Busch, 1997).

Communities can be included in peatland management through use of a number of different methods and mechanisms.
In helping decision-makers and researchers learn about different uses and understandings of peatland areas, methods such as transect walks with land managers, livelihood analyses and the development of participatory resource maps can be useful. Local knowledge about the peatland can be used to complement more technical scientific knowledge and together, combined knowledge can contribute to more acceptable, appropriate, and ultimately more sustainable management and policy (Berkes, 1999; Kelsey, 2003). Communities can also be involved in monitoring and assessment exercises. These kinds of activities can provide decision-makers with important information on the rate and nature of any changes to the peatland, assisting the development of policies for more sustainable use of the resource. Box provides an example of the ways in which different stakeholders were involved in a study of the sustainable management of the Dark Peak area in the UK's Peak District National Park.

Despite benefits, participatory approaches are not without their problems.
Participatory approaches have certain constraints (see Cooke and Kothari, 2001; Hickey and Mohan, 2004; Stringer et al., in press). For example, by involving local communities, expectations can be raised. If these expectations are then not fulfilled, it can lead to disillusionment. Participatory approaches also acknowledge diversity and complexity, rather than simplifying environmental management situations. Although this is one of the main strengths of the approach, and by involving local people in the analysis and interpretation of results, errors can be avoided, taking several different diverse viewpoints in account can make the results difficult to analyse and interpret. Finally, there is a danger that participatory methods can be applied mechanically, without an appreciation of underlying principles. This can prevent the benefits of participatory approaches from being realised.

Peatland management in the UK's Peak District National Park.
The Peak District National Park straddles four UK Government regions that together contain around 48% of England's population. The 22 million people a year that visit the Park make it one of the most visited National Parks in the world (Peak District National Park, 2004). But the demands these visitors place on the landscape must be balanced with the 38,000 residents who live there (Office of National Statistics, 2003).

The Dark Peak part of the Park is characterised by extensive heather moorland and blanket bog, surrounded by enclosed pastures in deep, narrow valleys. These habitats are both nationally and internationally important for their biodiversity. However, the multiple competing uses of the Park place complex demands on the landscape and current management practices fail to integrate the range of social, economic and environmental pressures. Management plans are required that can adapt to social values and changing scientific understanding.

Dougill et al. (2006) carried out a scoping study to explore the different kinds of moorland use and management employed by different stakeholders in the Dark Peak. A variety of participatory methods (including semi-structured interview and focus groups) were used to enable the researchers and other stakeholders to learn about the different management goals of each group. Scenarios of likely future change to the peatland were developed and discussed with the stakeholders, and indicators were identified that could be used to monitor progress towards the management goals under the different scenarios. The research is ongoing but preliminary analyses suggest that by bringing different stakeholders together to learn from each other, more sustainable management strategies are possible.

Generally, a combination of top-down and bottom-up peatland management approaches is favoured, since it is sometimes necessary to increase local awareness of changes and threats to the peatland, and build local capacity for monitoring and more sustainable management.

Effective approaches to peatland management problems can be developed in regions with shared management issues or transboundary problems related to peatland management.
There is a need to develop mechanisms to harmonize or integrate approaches to the management of peatlands across regional or national boundaries or within river basins. The need for transboundary cooperation is needed where the peatland physically crosses the boundary or there are issues of common concern on peatland management that can be shared and discussed between neighbouring

countries. Examples of this include the exchanges by the Grupo Paramos on the Andean peatlands in South America or between the countries of Southeast Asia working through the ASEAN Peatland Management Initiative (APMI). There is also a need to explore the options for the transfer of resources between different stakeholder groups that play a role in the management of peatlands – for example – upstream pastoralists or farmers could moderating peatland management practices to benefit downstream users of the water supply.

9.4 Peatlands in relation to policy processes

9.4.1 Peatlands and policy

Policy frameworks tend to treat peatlands either as forests or marshes, and often fail to recognize the special eco-hydrological characteristics of peatlands which are so important for their sustainable management.

Even the Ramsar Convention on Wetlands classifies peatlands under these two wetland types and does not provide peatlands with a class of their own (Peatlands are generally treated in sectoral policies such as policies on agriculture, water management, (bio)fuels, conservation or infrastructure development. There are few countries that have national or local peatland policies or strategies which are integrating the multi-sector interests in peatlands.

There are around 40 countries with National Wetlands Policies, of which only some are mentioning specifically peatlands. Also in many national Biodiversity Conservation Strategies peatlands are not mentioned or not recognized as a priority for biodiversity conservation. The limited prioritization of peatlands conservation for biodiversity conservation maybe partly due to their relatively low biodiversity combined with a lack of awareness on their high degree of biological diversity at habitat level and the relative high occurrence of characteristic species and endemics. Sometimes it may even be linked to a plain lack of awareness of the existence of peatlands and their special management needs.

> **Russian Peatland Policy and Action Plan**
> Peatlands cover over 8% of the Russian Federation and are sites of nature management within many sectors of the economy. The complex nature of peatlands requires an integrated approach and the Russian economy has traditionally been organized according to a sectoral principle. To integrate modern methods of peatland conservation and wise use into the system of state management a federal Action Plan on Peatlands Conservation and Use was developed by all interested stake holding organizations under supervision of the Ministry of Natural Resources (Wetlands International 2003).

A review of policies and practices in tropical peat swamp forest management in Indonesia (Silvius and Suryadiputra, 2005) found that while many of the Indonesia sectoral policies and legislation bear great relevance to peat swamp forest management, only few refer to or address specifically the special management requirements that are linked to the specific eco-hydrology of peat swamps. The most important one in that respect is Presidential Decree No. 32/1990, which stipulates that peat areas deeper than 3 meters should not be developed but retained in view of their water retention capacity. However, even this policy fails to recognize the need for an ecosystem approach when dealing with rain-fed peatlands: The policy allows reclamation and drainage of the outer zone of a peat dome with a depth of less than 3 meters which will invariably lead to subsidence of the deeper parts of the dome, a process that could continue until the entire dome would be lower than 3 meters and thus “eligible” for reclamation.

In many countries relevant policies and government regulations are clearly conflicting and can lead to confusion. It would be pertinent to review policies in this light, and to develop guidelines for land-use planning and management of peatlands taking into account their multi-functionality and their ecological and hydrological characteristics. In this regard it is important to note the particular applicability of an ecosystem approach for peat swamp management, as intervention in one part of the ecosystem can have significant impacts on other parts. Moreover, the management of separate peat land areas can not be seen separate from the management of their surroundings and the ecological and hydrological interconnections between the different habitats and land-uses within the entire water catchment.

The UNCCD’s definitions of desertification and land degradation focus mostly on desertification and land degradation in arid, semi-arid and dry sub-humid areas However, a number of countries have used the UNCCD framework and its synergy with the other Rio Conventions to address problems of peatland degradation.

For example, large scale drainage projects in Belarus in the 1950s-1990s led to extensive peatland degradation (Box 9.14). Funding was obtained for activities to reduce the degradation through the

GEF's Operational Program 12 (Integrated Ecosystem Management). This program funds projects that address ecosystem management in a way that optimizes goods and services in at least two of the GEF focal areas.

Definitions used within the Article 1 of the UNCCD

- **Desertification** means land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities;
- **Land degradation** means reduction or loss, in arid, semi-arid and dry sub-humid areas, of the biological or economic productivity and complexity of rainfed cropland, irrigated cropland, or range, pasture, forest and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns, such as: (i) soil erosion caused by wind and/or water

(ii) deterioration of the physical, chemical and biological or economic properties of soil

(iii) long-term loss of natural vegetation

9.4.2 Addressing root causes and enhancing Implementation Mechanisms

In developing countries and countries in transition where poverty may be a root cause of unsustainable peatland resource exploitation, development may be the only way to create opportunities for peatland conservation.

On the other hand, where development of peatlands coincides with the need for drainage and mining, it will generally be unsustainable and not be conducive to the conservation of the peatland carbon store and the biodiversity. Without appropriate economic alternatives and incentives it may often be impossible to maintain and manage conservation areas or invest in rehabilitation of degraded peatlands. Poor people must have a livelihood before being able to refrain from over-exploitation of natural resources. In poverty stricken regions, governments have to generate sufficient economic growth – even by unsustainable means – before being in a position to take environment into consideration. Therefore, incentives for short-term unsustainable development, including for instance logging and land conversion remain high. Development therefore is key to peat swamp forest conservation and the sustainable management and rehabilitation of degraded peatlands.

Considering the declining incomes from agriculture and forestry on peatlands there is a pressing need to enhance alternative income opportunities to local rural populations. In the meantime it is important to ensure that their lands and resources are no longer degraded and where agriculture and plantation forestry on peat is practiced it is optimized in terms of sustainability. Without sufficient revenues from the land, poor people may be forced to go for the cheapest but not necessarily the most sustainable land-use management options. This could e.g. include the use of fire for land clearance.

9.4.3 New emerging innovative options

The huge, but only recently recognized, CO2 emissions from tropical peatland deforestation and degradation, represents one of the single largest but also most concentrated sources of greenhouse gas emissions and provides as such also a major opportunity to reduce current global greenhouse gas emissions (Silvius 2006, Hooijer et al. 2006).

Whereas tropical deforestation in general covers hundreds of millions of ha worldwide and generates emissions of 1-2 billion tonnes of carbon dioxide, the degradation of peat swamp forests which is mainly confined to 12 million ha of degraded peat swamps in South-east Asian leads to a larger total emission. However, significant emission reductions can also be achieved through peatland conservation and restoration in other parts of the world. The linkage to poverty issues (see chapter 3) and biodiversity loss ties it to two other globally recognised priorities.

Some newly emerging possibilities for conserving peatlands, particularly for their carbon storage function, are payments for Reduced Emissions from Deforestation and Degradation (REDD) as currently being developed by Parties to the UNFCCC. The World Bank and other institutions are exploring options to establish REDD funding mechanisms to support pilot schemes, including the option of carbon fund payments to national & local governments which need to be based on a national baseline monitoring, and the option for payments to private and community stakeholders and beneficiaries for their "environmental services". As peatlands cut across all forest management, conservation and land use (production, industrial and agricultural crops) types, maintaining the welfare of traditional local communities in peatlands is a major concern.

Parallel to this there are numerous private sector initiatives which indicate a strong interest in investment in avoided emissions through peatland rehabilitation and reforestation as a means to compensate for industrial emissions elsewhere. Some investors see opportunities for trade in "Carbon futures". These interests could well provide the local people in peatlands with opportunities for development of a new community-based public service. According to Butler (2007) preserving tropical forest and peat swamp that would otherwise be converted and collecting the resulting recurrent revenue provided by the carbon offset market may be more lucrative for landowners in some areas than conversion to palm oil. With a carbon price range of US$ 14 to US$ 22.12 similar level profits may be derived over a period of 25 years.

However, much will depend on how the funding is used and how much of it can be channeled to local stakeholders. The carbon market provides a significant opportunity for a pro-poor approach, in which consideration should be given to the equitability of the development in terms of revenue sharing between investors and local stakeholders. Funding schemes that will enhance access of local stakeholder groups to carbon funding, e.g. through special REDD micro-financing facilities, could create hitherto un-existing economic incentives and help to empower these stakeholders. This would increase chances of successful development of an innovative community based environment management service sector as part of the voluntary carbon market.

Carbon financing mechanisms (CDM): Under Article 3.4 of the Kyoto Protocol, activities that enhance carbon sequestration in agricultural soils can be counted towards emission reduction targets, and can be traded on the international carbon market via the Protocol's "flexibility mechanisms". Since a large proportion of peatlands are extensively grazed or under some form of agriculture, money from this source could be used to finance drain and gully blocking on a far larger scale than is currently possible.

The option for local communities to provide services to the emerging carbon market in terms of peat swamp forest conservation and restoration represents a major opportunity for linking climate change mitigation to poverty reduction.

It also enhances options for other types of strategies or combinations, particularly relevant to countries with no substantial agricultural subsidies, for instance the development of innovative financial instruments such as Bio-rights (Silvius et al, 2002). The Bio-rights approach involves establishment of business contracts, providing micro-credits for sustainable development in exchange for the conservation or rehabilitation of globally important biodiversity or environmental values. The business partners are "the global community" (represented by a broker, e.g. NGO or bank) and a local partner, e.g. a local community or a major community-based stakeholder group. The local (community) business partner will pay interest over the micro-credit not in the form of money, but in terms of biodiversity conservation services – defined by mutually agreed environmental or biodiversity related indicators. An indicator frequently used is for instance the survival rate of planted tree seedlings after 5 years of reforestation. The micro-credit level is linked to the opportunity costs of sustainable use and conservation of the natural resource base and biodiversity. As such, the Bio-rights approach removes the incentive for unsustainable development and allows the public value of key biodiversity wetland/peatland areas to be transferred over time to local stakeholders as a direct economic benefit. The incentive can be increased by allowing the credit itself also to be repaid through such services, enabling the development of community-based revolving funds for sustainable development. This again will trigger community-based monitoring, as the whole community will stand to loose if the business is unsuccessful.

The Bio-rights approach can of course also include such indicators as carbon store conservation and carbon sequestration, as well as the maintenance of wider ecosystem services such as water management and biodiversity values. As the micro-credit levels in the Bio-rights approach are directly related to the opportunity costs of sustainable development and conservation, the approach does not require economic valuation of biodiversity or the ecosystem services that are maintained. This distinguishes it from Payments for Environmental Service (PES). Bio-rights schemes are operational in the buffer zones of the Berbak national park in Jambi, Sumatra, and are also used in many other community-based wetland restoration projects in Indonesia, such as in the Tsunami hit region of Aceh (involving sustainable coastal development and mangrove reforestation) (See www.bio-rights.org).

Evidence has been accumulating that in many cases, natural peatland habitats generate marked economic benefits, which exceed those obtained from habitat conversion.

Economic costs associated with damage to ecosystem services can be substantial: for example, the damage of the 1997 Borneo fires to timber, tourism, transport, agriculture, and other benefits derived from or linked to the forests is estimated at $4.5 billion in addition to the actual cost of fighting the fires (Tacconi 2003).Significant investments are often needed to restore or maintain non-marketed ecosystem services: e.g. the costs of flood prevention in down-stream areas.

Payments for Ecosystem Services are already operational in many parts of the world, but so far not yet practices in peatlands. In some regions these innovative payment schemes are supported by policies and trust fund constructions. However, techniques and local capacity for monetizing ecosystem functions are generally under-developed. Some ecosystem functions can not be valued, because their precise contribution is not known and indeed unknowable until they cease to function. Other functions cannot be monetized because there is nothing equivalent to be put in their place: intrinsic values are by definition without price. Consequently, any weighting can only be partial and whole ranges of values, benefits or disadvantages escape monetary evaluation. Studies valuing multiple functions and uses, and studies which seek to capture the 'before and after' states as environmental changes take place, are rare. By and large it is the latter types of analyses that are most important as aids to more rational decision taking in ecosystem conservation versus development situations involving different stakeholders (local, national and global). Aggregate (global scale) estimates of ecosystems value are problematic, given the fact that only 'marginal' values are consistent with conventional decision-aiding tools such as economic cost-benefit analysis. Despite these difficulties, valuation data are useful in decision-making by illuminating tradeoffs.

Valuation studies of industrialized countries focus on recreational and existence values held by urban consumers (travel cost models, contingent valuation). In developing countries, on the other hand, ecosystem values related to production and subsistence remain relatively important, although this is changing in regions characterized by rapid urbanization and income growth. In general, valuation data provide support for the hypothesis that net ecosystem service values diminishes with biodiversity and ecosystem loss.

9.4.4 The need for local policy embedding of innovative mechanisms

There is an urgent need to create an enabling policy environment for innovative mechanisms such as the emerging market in Verified Emission Reductions from peatlands and forests, to provide the necessary legislative basis for long-term commitments from all stakeholders and management frameworks that will provide carbon buyers with sufficient guarantees that their investments – represented by the preserved and rehabilitated sub- and above-soil carbon store – are safe.

This will require more than the usual five year plans, and commitments must be binding well over legislative periods of current elected authorities.

Also for carbon projects that are based on business deals at the local – community – level such long-term commitments are needed. For instance, investment in reforestation of community owned buffer zones adjacent to protected areas, need contracts that are also binding to the future generations. This poses considerable new challenges, as it is unpredictable what incentives or disincentives may arise in the future, that may tip the balance and will change priorities of local stakeholders.

Many other risks need to be assessed in relation to selling and buying of avoided emissions from peatlands, including the risk of fires. This risk is particularly predominant in South-east Asia during the recurring El Niño events, but also occurs in large parts of eastern Europe. Such risks may need to be covered by newly developed structural assurances through new government policies and legislation, and perhaps involving also new insurances catering for this sector.

Current developments of REDD and private sector initiatives are pushed hard to become operational soon. However, the question arises whether these ideas and initiatives have sufficiently matured. Immature ideas and projects will lead to failures and disappointments and can discredit and endanger the new emerging sector, affect carbon price and create risks that so far have not been part and parcel of community- and government-based natural resource management planning. It is very important that any voluntary carbon credit scheme will adhere to a same set of standards and criteria. For peatlands, with their special eco-hydrological character and management requirements, as well as their complex social and economic setting, these have not yet been developed. Pilot schemes will be needed, and therefore there is a strong need for coordination and sharing of lessons learned between all projects and efforts that relate to peat CO_2 management and the promise this holds for poverty reduction, biodiversity conservation and climate change mitigation.

It is now widely recognized that the peat issue is part and parcel of the REDD agenda. This creates a strong basis for international cooperation and support. There are many signals of strong interest to assist from both the donor community as well as private sector, and many initiatives are being developed as we speak. A new market is emerging that can be supplied by a community-based service sector. It will create significant opportunities for local community development as well as private sector investments. However, there is an urgent need to create and enabling policy environment for these developments. Voluntary carbon initiatives will require certain guarantees that the investments will not be in vain and can be efficiently channeled to where they can be most effective. For effective development of the REDD market long-term commitments are needed backed up by policies and legislation.

9.4.5 Harmful subsidies, policies and taxes

Peatlands have been subject to a wide array of perverse and harmful incentives in the form of policies and subsidies.
In many countries peatland drainage is still encouraged under various kinds of policies, subsidies and tax breaks. There are ample examples in western Europe where high mountain peatlands were significantly affected by former EU subsidies for sheep (encouraging more sheep to be held then the carrying capacity of the peatlands) and afforestation subsidies and tax breaks.

Biofuel policies and subsidies
A clear example of a policies and subsidies that may generate the opposite effect of their original intention are the recent policy development in the EU regarding biofuels for transport and energy:

- The EU promotes the use of 10% bio-diesel by 2015. This could come from locally produced biofuels, e.g. rapeseed, corn, but also from imported biofuels such as palm oil.
- The Netherlands recently committed to 740 million Euro in subsidies in support of the develop power stations.

The increased international interests in biofuels created a significant reaction in South-east Asia stimulating plans for rapid expansion of palm oil, particularly also to cater for the international growing biofuel market (ref…). However, a substantial part of the South-east Asian palm oil plantations occur on peatlands and the palm oil expansion is expected to take place largely on peatland areas (Hooijer *et al*. 2006). Oil palm plantations require relatively deep drainage (at least 60cm), causing significant CO_2 emissions. Use of palm oil could therefore lead to 3 to 10 times more CO_2 emissions compared to using fossil fuels (Silvius, 2007).

There are also ongoing EU subsidies for e.g. planting corn, which creates a perverse incentive to planting this crop also on peatlands – which otherwise would be avoided being marginal soils.

Sometimes land tenure is linked to productive use of peatlands, providing a disincentive to conservation and restoration. In Indonesia, for instance, there are local policies that require clearance of land every three years, without which the land tenure can be lost. This creates an incentive for burning as it is the cheapest option for land clearance.

Basic approaches for sustainable peatland management to be considered in policy and practice.
In defining requirements for management of peat swamp forests, we can draw up the following basic lessons to be considered when planning interventions in peatland areas.

Precautionary approach

1. In planning of land-use in peatlands, it is advisable to use the precautionary approach. Large scale developments in peatlands should be pursued only after considerable research and after successful completion of pilot projects.

Ecosystem approach

2. Land-use planning in peatlands should follow the ecosystem approach, taking special account of the hydrological vulnerability of peat domes and the ecological relationships with the surrounding habitats and land-uses. Particular regard should be given to the place of the area within the water catchments/ water shed, and the potential impacts of and on upstream and down stream habitats and land-uses (including potential land-uses). It may even be necessary to consider multi-river basin complexes, as multiple watersheds may be dependent on shared peat domes, and impacts on one river basin may affect the shared hydrological basis.

Integrated approach

3. Wise management of peatland ecosystems requires a change of approach from single sector priorities to integrated, holistic planning strategies, involving all stakeholders to ensure that consideration is given to potential impacts on the ecosystem as a whole. Land-use planning in peatlands should involve all relevant sectors and major stakeholder groups, including local people, from the outset of development planning. A precondition for successful integrated planning is the (enhancement of) awareness of the various groups regarding peatland ecology and hydrology, and the full scale of values that peatlands may have.
4. The use of a peatland for a specific purpose may have considerable side effects and all other functions must be taken into account in the full assessment of the suitability of a particular use.

5. With respect to side effects, a use could be considered permissible when:
   - Negative side effects will not occur
   - The resources and services affected will remain sufficiently abundant, or
   - The resources and services affected can be readily substituted, or
   - The impact is easily reversible

   In all other cases, an integrated cost benefit analysis should be carried out involving thorough consideration of all aspects of the proposed use.

<u>Allocation of land-use status</u>

6. Allocation of land-use status in peatlands should take account of the hydrological vulnerability of peat swamp forests, their susceptibility to chemical erosion (oxidation – leading to CO_2 emission), their vulnerability to fires and their values for biodiversity conservation, water retention and climate change mitigation.

9.4.6 Synergy between conventions to develop integrated policy frameworks

Peatlands are a habitat where current global priorities in climate change mitigation, combating land degradation, stopping the loss of biodiversity and reducing poverty are coming together.

	Impacts:	**Business of:**
Win 1	**Climate change**	**United Nations Framework Convention on Climate Change**
Win 2	**Land degradation**	**UNCCD, World Water Forum**
Win 3	**Loss of biodiversity**	**CBD, Ramsar Convention**
Win 4	**Poverty**	**Commission on Sustainable Development**

9.5 Conclusion

This Assessment has indicated in various ways the disproportionate relevance of peatlands in relation to climate change mitigation, combating land degradation, biodiversity conservation and poverty reduction, and as such the need to consider peatlands within the context of the major global policy platforms, including the UNFCCC, UNCCD, World Water Forum, CBD, Ramsar Convention on Wetlands, and the Commission on Sustainable Development. The synergy between the conventions and policy platforms in this regard calls for enhanced coordination and cooperation. The donor community is increasingly recognizing the need for integration of these agendas, but current global policy processes fall short of the required sharing of lessons learned and development of inter-sectoral approaches to the conservation and wise use of peatlands world-wide.

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