CHAPTER 4:

CLIMATE IMPACTS OF PEATLAND FORESTRY

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4.1. Introduction

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

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

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

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4.2. Forestry in peatlands

4.2.1 Common principles

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

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

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

4.2.2. Extent of peatland forestry

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

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

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

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

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

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

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

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

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

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

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

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

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

4.2.3 Silvicultural practices

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

4.3. Ecological impacts of peatland forestry

4.3.1 Peatland carbon cycle

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

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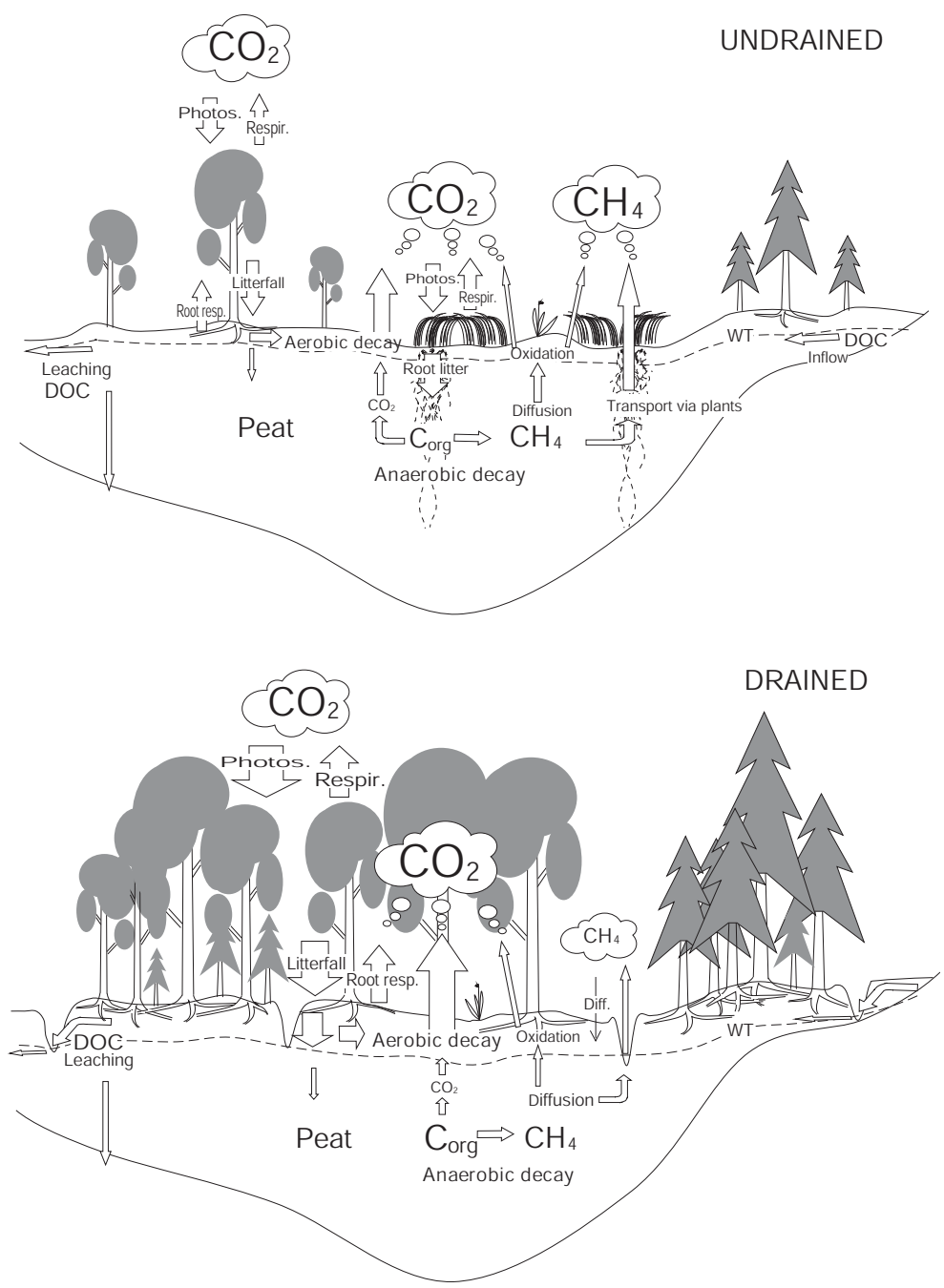

Figure 4.3. *C cycle in undrained mires and peatlands drained for forestry.*

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

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

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

4.3.2. Impacts of drainage on the carbon cycle

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Decomposition

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

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

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

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

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

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

Leaching

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

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

Soil CO_2 efflux

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

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

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

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

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

Soil C stocks and NEE

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

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

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

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

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

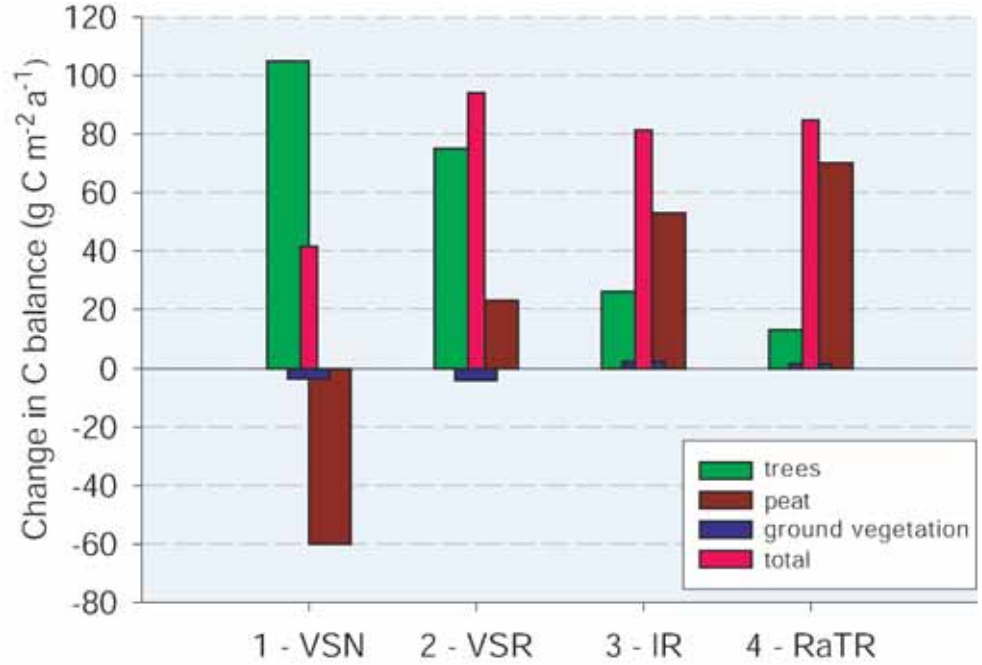

Figure 4.4. *The change in the C balance of the tree stand, ground vegetation and peat soil in four sites on Lakkasuo mire, Central Finland (Minkkinen et al., 1999). C balance of a peatland after drainage for forestry is strongly dependent on the site type and the consequent differences in influx (primary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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

4.3.3. Impacts of other silvicultural practices

Fellings

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

Even though thinning and clear-felling somewhat increased soil temperatures (about 1°C), no significant increase in heterotrophic soil respiration of the old organic matter (excluding the roots from cut trees), was detected in an experiment in Central Finland (Minkkinen, unpublished data). Instead, the total soil respiration was quickly reduced by the same proportion

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of root respiration, as was estimated by trenching before the fellings. The small rise of water table (10 cm) slightly reduced CH_4 consumption at the site. Nitrous oxide emissions increased from slash piles, probably because of a liberation of N into the soil from decomposing needles. No impact was seen elsewhere in the clear-felled area.

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

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

Soil preparation

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

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

Fertilization

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

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

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

4.4. Radiative forcing from peatland forestry

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

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

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

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

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

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

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

4.5. Conclusions

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

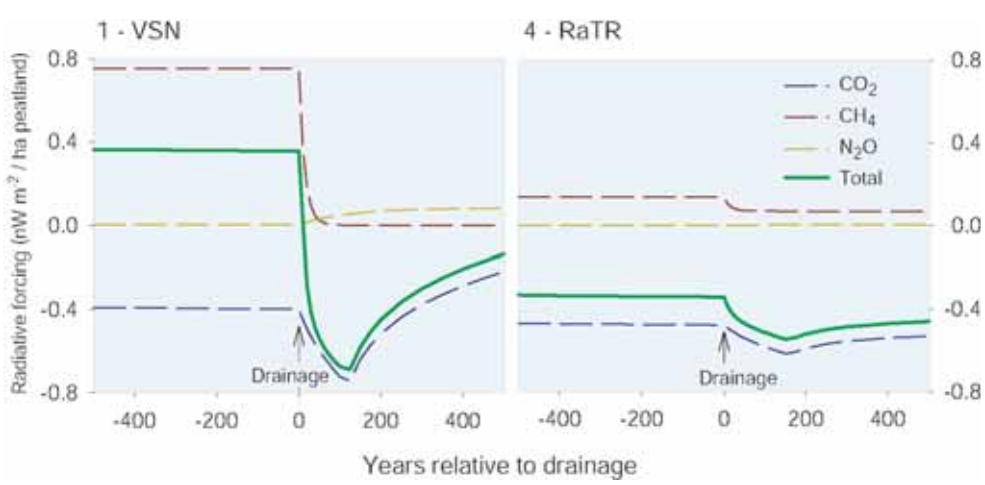

Figure 4.5. *The impact of drainage on the radiative forcing of a mire ecosystem in the minerotrophic (1 - VSN) and ombrotrophic (4 - RaTR) sites in Lakkasuo mire, Central Finland (Laine et al., 1996). The flux estimates behind this figure are slightly different from that in Figure 4.4, but the direction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

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

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

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

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