The peatlands of North West Europe and Climate Change

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Contents

List of Figures

List of Tables

Abstract

Peatlands are one of the world's most important ecosystems. Covering a mere 3% of the world's terrestrial surface, peatlands contain 550 Gigatonnes (Gt) of carbon making them the most important long term carbon sink in the terrestrial biosphere. This ability of peatlands to store CO_2 and GHG means they have a net cooling effect on the global climate. It has been estimated that in the last 10,000 years since the last Ice Age the atmospheric carbon sequestered in peats has served to reduce global temperatures by about 1.5–2 °C. Humans have been causing the degradation of peatlands in NW Europe for centuries through management regimes such as the draining of peatland for forestry and agriculture. More recently humans have been causing the degradation of peatlands indirectly through anthropogenic climate change. The degradation of pristine peatlands causes the release of considerable amounts of CO_2, DOC, POC, sediment and N_2O causing peatlands to switch from net GHG sinks to sources. The degradation and destruction of peatlands causes the release of 3 Gt of CO_2 annually. Climate change is already affecting peatland ecosystems and its influence will only get stronger in the future. The response to changes in temperature, precipitation, hydrology and vegetation will vary greatly between ecosystems and regions. Some ecosystems such as palsa will be extremely vulnerable to a changing climate. The melting of permafrost will lead to the release of vast amounts of CH_4 and CO_2 that will provide a positive feedback to global warming. Some peatland ecosystems may benefit from a longer growing season with increased primary production. This may have a positive effect on C sequestration although a warmer climate with more regular droughts will cause the lowering of peatland water table and the invasion of vascular plants and trees onto the peat surface which will cause the loss of the peat C stocks. Over the short term these changes will likely see the release of vast amount of GHG that will create a positive feedback to climate change (Betts et al., 2004; Friedlingstein et al., 2010; Lewis, 2006). Peatlands however have adapted to climate change in the past and warmer temperatures will open up new territories for peatlands to colonise. It is unclear how the peatland ecosystems will respond to climate change in the future. What is clear is that the management of peatlands by humans will greatly influence the extent of peatland loss caused by climate change. In turn management decisions will affect the rate of mineralisation and the loss of the C sequestration capacity of peatlands this will affect the rate of future climate change. The rewetting and restoration of peatlands has the capacity to secure existing carbon stocks and reinitiate the C sequestration capacity of degraded peatlands. Peatland restoration projects have been recognised by the new Kyoto Protocol and should be incorporated into the very heart of the national greenhouse gas reduction strategy of Ireland, Britain, Finland, Norway and Sweden.

Introduction

The peatland of North West (NW) Europe are inextricably linked to local and global climate. The cool and humid weather off the region with high precipitation creates the water logged conditions which hinder aerobic decomposition of plant material and cause the formation of peat and peatland ecosystems (Mäkilä & Saarnisto, 2008; Moore and Bellamy, 1974). CO_2 is removed from the atmosphere through the process of photosynthesis and is sequestered in the form of peat over the course of millennia (Joosten & Clarke, 2002). Peatlands are the most efficient terrestrial ecosystems in storing carbon. They cover a mere 3% of the World's land area yet their peat contains as much carbon as all terrestrial biomass an twice as much as all the world's forests combined (Parish et al. 2008). They regulate the amount of CO_2, N_2O and CH_4 in the atmosphere and act as net greenhouse gas (GHG) sinks in their pristine state (Holden, 2005; Limpens et al., 2008). However degradation and land use change by human's as well as climate change are compromising this vital ecosystem service and switching the peatlands of North West Europe into GHG sources (Moore, 2002; Worrell et al., 2011). The release of vast amounts of GHG will have a positive radiative forcing (RF) on atmospheric temperature and create a positive feedback to climate change. Improving our understanding of the relationship between the peatland ecosystems of NW Europe and climate and the way that perturbations within peatland biomes affect climate is vital given the threat posed to human society and global biodiversity by climate change. This scientific literature review will try to illuminate current scientific understanding on the relationship between peatlands and climate change. It will firstly examine the effect that pristine peatlands have on global climate through the sequestration and production of GHG. It will then go on to examine what affect that the drainage and management of peatlands for agriculture and forestry has had on the GHG balance of peatlands. Finally this paper will examine the effect that climate change will have on the peatlands of NW Europe based on current evidence and future projections.

Section 1

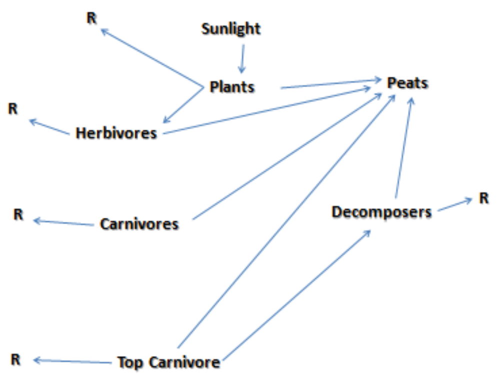
Peatlands

Peatlands may be thought of as 'organic wetlands (Charman, 2002). Peatlands are one of the world's most important ecosystems. Globally they cover over 400 million ha and are the most common wetland accounting for about a third of the estimated area of the world's wetlands (Parish, et al., 2008). Being a wetland ecosystem they contain the characteristic features of a wetland in that they are shaped by the presence of water and have soil conditions with low oxygen levels. Like other wetlands they have specialised biotas that have evolved to live in these conditions (Charman, 2002; Joosten & Clarke, 2002).

Peatlands as the name suggests are defined by the presence of peat (Charman, 2002; Lund, 2009). Most sources define peat as a substance that is composed of the partially decomposed remains of plants with over 65% organic matter and less than 20-35% inorganic matter (dry weight) (Charman, 2002; Joosten & Clarke, 2002). Any wetland area therefore where organic material has accumulated to a depth of 30 - 40 cm or greater can be defined as a peatland (Lund, 2009). However, as the definition incorporates all places in which peat has accumulated to a certain depth, it also includes non-active peat forming areas such as peatlands which have lost their peat forming function due to factors such as drainage or disturbance (Lund, 2009). The term mire commonly used throughout Europe and Russia is used to distinguish between actively peat forming ecosystems and those that are no longer forming peat. It includes all those ecosystems known commonly as swamp, bog, fen, moor, muskeg and peatland (Gore, 1983 a; Joosten & Clarke, 2002; Lund, 2009). Given the vast array of different mires all over the world a number of elaborate classification systems have been developed to subdivide these peatlands further based on features such as morphology, hydrology, stratigraphy, chemistry, plant species composition and vegetative structure (Charman, 2002; Gore, 1983 a). Given the complexity of these ecosystems and the fact that the vast array of criteria used to define them are continuous rather than discrete, these simplistic models often fail to fully subdivide all mires sufficiently (Charman, 2002; Gore, 1983 a).

Peat formation

Under normal circumstances energy from the sun enters ecosystems through photosynthesis. This results in the accumulation of plant material which may then eaten by herbivores or alternatively the plant may expire and its organic matter will enter into a decomposer food chain. In the majority of ecosystems microorganism will breakdown and respire this detritus until it is humified (Moore and Bellamy, 1974). In most ecosystems the cycling of matter through these processes is relatively fast and complete (Joosten & Clarke, 2002). However in mire ecosystems the decomposer food chain is impaired by the physical conditions, with which they are associated, namely the presence of waterlogged conditions which results in soil conditions with low oxygen levels (Mäkilä & Saarnisto, 2008; Moore and Bellamy, 1974). The large heat capacity of water causes the mire temperatures to be lower than the ambient temperature. Oxygen is limited as a result of its low diffusion rate in water. Both of these factors inhibit microorganisms that would otherwise decompose the organic matter (Joosten & Clarke, 2002). In the absence of decomposition much of the plant material produced by primary production in mires accumulates. The herbivore community in a mire ecosystem is only a small fraction of the plant material being produced. The composition of the undecomposed material which forms the peat is therefore mostly plant material with a small proportion of animal matter (Moore and Bellamy, 1974). Some Sphagnum peat can be constituted of as much as 99% organic matter (Gore, 1983 a). The energy flow in a mire ecosystem is described in Fig. 1. As can be seen from the system not all of the energy is released through respiration nor is it accounted for by the processes of succession or accumulation of living matter in the system (Moore and Bellamy, 1974). This incomplete cycling where plant production exceeds decay results in a positive carbon balance. As long as the system remains wet enough to inhibit decomposition then this energy will remain locked within the system as peat indefinitely (Moore and Bellamy, 1974).



The rate of peat accumulation is determined by the rate of decay rather than the rate of productivity (Mäkilä & Saarnisto, 2008). Water is the most important external factor limiting decay yet it is not the only factor that determines the character of the peat. The chemical and structural composition of the organic material also determines the extent of decay.

Fig. 1 The Energy flow within a mire ecosystem (From Moore & Bellamy, 1974)

The degree of decomposition is different for different plants, plant parts and the substances of which they are constituted. Therefore they contribute differently to the end composition of the peat (Joosten & Clarke, 2002). Many commonly found mire species such as Sphagnum species, other mosses, sedges, grasses, and woody plants are important in determining the composition of the peat (Gore, 1983 a; Joosten & Clarke, 2002).

A simplified two layer structure has can be used to describe functioning blanket bogs and raised bogs (Fig. 2). The surface layer made up of the upper 10-20 cm is known as the acrotelm. It is the oxic layer and it is therefore the biologically active layer where most of the decomposition takes place. This upper zone is where the living biomass of the bog is found. It is here that carbon is sequestered and peat is formed (Bain et al. 2012). As much as 10-20% of the decayed matter formed in acrotelm passes into the lower zone known as the catotelm. The catotelm is waterlogged and anaerobic and so very little decomposition takes place (Bain et al. 2012). The peat produced in the acrotelm may be stored in the catotelm for millennia. What little decomposition does take place in these anoxic conditions is returned to the atmosphere as carbon dioxide (CO_2) and methane (CH_4).

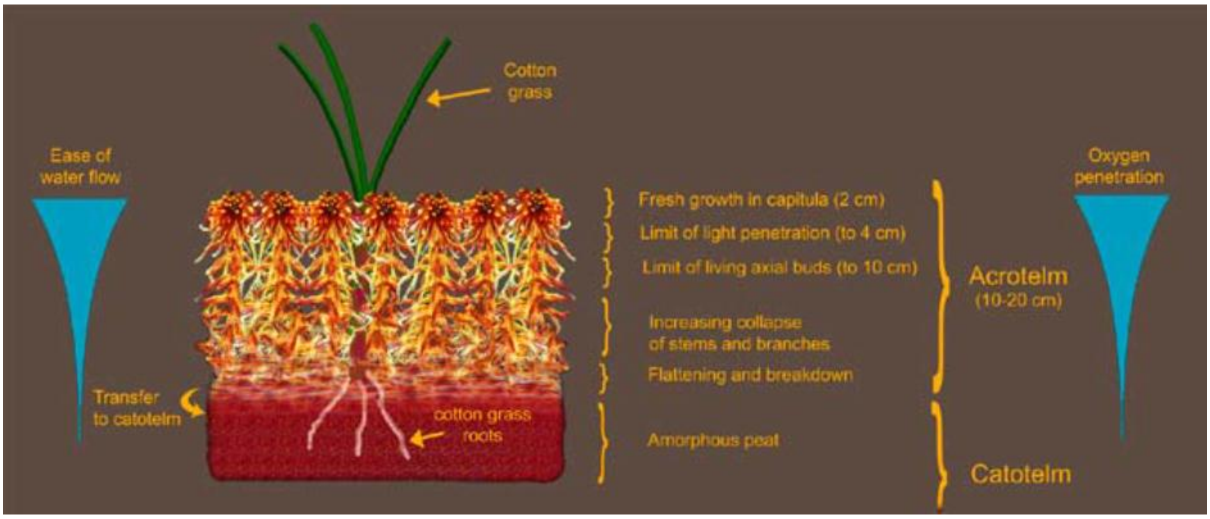


Fig. 2 The peat structure of a typical functioning bog (from Bain et al., 2011)

In boreal mires approximately 4-10% of the carbon that is annually photosynthetically fixed is returned to the atmosphere as CH_4 (Mäkilä & Saarnisto, 2008). The acrotelm has a large pore structure with facilitates the storage of large amounts of water. The specific yield which is the ease with which water can be drained out of the peat by gravity and the hydraulic activity which is the rate at which water can move through the peat is high in the acrotelm. The pore size in the catotelm is smaller because the peat is highly decomposed. As a result of its small pore size the specific yield and hydraulic activity in the peat decreases with depth (Fig. 2). Water retention increase with depth and therefore the catotelm is permanently waterlogged (Stack, et al., 2008). Due to the low hydraulic conductance the nutrient transfer in this lower layer is extremely low. Up to 95% of water run-off occurs in the uppermost 10cm (Bain et al. 2012). The relationship between bogs and climate is a dynamic one. A bogs existence is reliant on maintaining a balance between the growth of plants such as Sphagnum, the decomposition of peat, the water table, and the thickness of the aerobic acrotelm. Peat production is therefore vulnerable to changes in humidity (Seppä, 2002).

Two major classification systems are used to distinguish between most types of mire based on what their nutrient source is. Bogs are classified as *Ombrotrophic* mires. This means that vegetation relies on precipitation and atmospheric inputs for their nutrient and water supply (Doyle, 1990; Raeymaekers, 1999). Bog floras receive their supply on nutrients exclusively from precipitation and atmospheric fallout. Bogs are not in contact with the under lying mineral soil (Raeymaekers, 1999). Under low nutrient condition peat forming Sphagnum moss species replace ‘brown moss’ species. Sphagnum species utilise what little nutrients are supplied by the rain water. They take up the cations from the minerals and release hydrogen ions. This acidifies the bog (Raeymaekers, 1999). The nutritional status of bogs is therefore oligotrophic and the characteristic vegetation has evolved to thrive in these nutrient poor conditions (Gore, 1983 a).

Minerotrophic mires such as fens receive there nutrients from their water supply. There nutrient status may therefore be eutrophic, mesotrophic or oligotrophic (Gore, 1983 a). Examples of these different types will be discussed later.

Peatland formation

Generally speaking bogs and mires develop when an excess of water accumulates on land. There are two major pathways to peat formation called *terrestrialization*, or *plaudification* (Charman, 2002; Joosten & Clarke, 2002; Schouten & Noreen, 1990)

Terrestrialization is a process whereby a water body such as a shallow lake or pond becomes overgrown with aquatic vegetation (Gore, 1983 a, Schouten & Noreen, 1990). The initiation is a topographic rather than a climatic process. The transformation of the lake is dependent on a range of variants such as the trophic condition of the lake, the catchment area and the lake topography (Gore, 1983 a). With time a fen is formed and peat layers begin to accumulate (Schouten & Noreen, 1990). This process continues to the point where the water table is at or below the surface for at least a part of the year. In the Northern areas of Europe, Asia and America south of the alpine pine belts where the climate is sufficiently cool and wet this succession will continue after the basin has been filled up with peat (Schouten & Noreen, 1990). Sphagnum and other peat forming species may then colonise the mire and accelerate the rate of peat formation (Charman, 2002). Peat will continue to accumulate and eventually a bog will form on top of the fen (Schouten & Noreen, 1990).

The other form of mire initiation is *plaudification*. Plaudification describes the formation of peat directly on mineral soil without an aquatic phase (Charman, 2002). This is a process whereby land adjacent to mires which was previously dry and maybe even vegetated is blanketed by the expanding mire. The land may previously have been woodland grassland or bare soil exposed by the retreat of the glaciers during the periglacial period (Charman, 2002). The process is initiated when either allogenic or autogenic factors cause a change in the hydrology of the area. Allogenic process may include an increase in local precipitation as a result of deterioration in the micro or macroclimate such as happened during the Atlantic period (8,000 – 5,000 B.P.) (Gore, 1973 a; Gore, 1973 b; O'Connell & Molloy, 2001; Verrill & Tipping, 2010). It is thought that the anthropogenic deterioration in soil conditions initiated during the Neolithic Land-nam and also evident during parts of the Bronze and Iron Age in Ireland and Britain was another cause of plaudification. Extensive cutting or burning of woodlands and exhaustion of the soil by over farming caused deterioration in many soils adjacent to mires (Gore, 1973 b; O'Connell & Molloy, 2001; Verrill & Tipping). The removal of tree cover would have increased soil wetness (Gore, 1973 a).The spread of peat within a region after excessive accumulation would be an example of an autogenic factor. In the

Northern Hemisphere this process peaked during the early Post glacial period during ~ 6500 B.P. and ~4800 B.P. and ended around about 2000 B.P. (Schouten & Noreen, 1990).

World Distribution

As has been said peat may be found anywhere where the decomposition of plant material is suppressed by waterlogging. Mires have therefore existed ever since wetland plants first evolved. The coal and lignite we are familiar with today originates from peat which was formed in tropical mires during the Upper Carboniferous (320 - 290 million years ago) and the Tertiary (65 - 3 million years ago) periods (Joosten & Clarke, 2002). It ranges in character from the moss peat found in the arctic, subarctic and boreal regions; to the reed/sedge peat and forest peat of the temperate regions; to the mangrove and swamp forest peat in the humid tropics (Montarella et al. 2006).

Mires are found across the world from the tropics to the north in 175 countries worldwide (Bain et al 2011). Peatlands defined as soils with <50% organic matter and < 30cm deep cover between 386 and 409 million ha. Tropical peatlands cover ca.40-50 million ha globally (Clough et al. 2008). Mire formation is as we know favoured by conditions which inhibit the decay of organic material. Mires are therefore predominantly found in the cool and wet continental boreal and sub-arctic regions. The northern mires cover ca. 3,460 -103 km^2. The Russia, Canada, the USA and Indonesia are the countries with the greatest peatland areas. They account for < 60 % of the global peatland area (Parish et al. 2008). Topography also plays its part in the distribution of peatland as a large quantity are found in the flat land areas in western Siberia, Hudson Bay , the South East Asian coastal plains, and the Amazon Basin. Figure 3 gives an indication of the global distribution of peatlands (Bain et al. 2011).

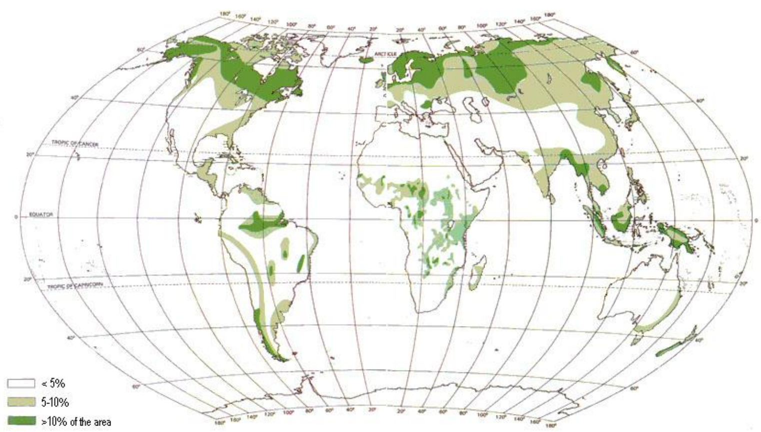


Fig. 3 The global distribution and land cover of mires and peatlands (LIFE, 2009)

The peatlands of North Western Europe

In Europe, peatlands extend to ca. 515,000 km^2 (Bain et al. 2011). They cover 5–6% of the European continental land surface and are concentrated in northern and temperate lowlands (Drösler et al. 2008). The cool and wet climate of N to occur within a mean annual air temperature range of –12° to 5°C and a mean annual precipitation range of 200 to 1000 mm (Zicheng, et al., 2009).The vast majority of Europe's peatlands are found in NW Europe with one third of them occurring in Finland, and over a quarter in Sweden (Montarella et al. 2006). Throughout NW Europe the bedrock geology determines the major patterns observed within the landscape. The expansion and retraction of glaciers during the Ice Ages of the Pleistocene have modified the landscape. These two factors of underlying geology and its modification by the Ice Ages have determined peatland distribution in Ireland, Britain, Sweden and Finland. Within these landscapes in areas where drainage is impeded or precipitation is high and evapotranspiration low waterlogging occurs and peatlands are found (Hammond 1981).

Many of the peatlands and peat deposits that are evident in the British Isles and Fennoscandinavia today have developed since the Late Devensian period some 15,000 years B.P. following the last Ice Age. Although Britain and Ireland are small in terms of size they contain extensive peatlands in terms of proportional area (Gore, 1983 b). The high mid-latitudinal position of the islands and there cool temperate and maritime climate with high humidity and precipitation are well suited to peat formation (Gore, 1983 b). The prevalence of flat or concave topography as well as geological features and poorly draining soils that are a legacy of processes related to glaciation and periglaciation on the two islands also promote the formation of peat (Gore, 1983 b). The Republic of Ireland is third only to Finland and Canada in proportional area of peatland cover with Northern Ireland and Scotland coming ranked sixth and seventh respectively (Gore, 1983 b). In Ireland it is estimated that peatland covers ca. 20% of the Island (Renou Wilson et al. 2011). Peat soils in Ireland cover approximately 1,466,469 ha or ~20.6% of the country (Connolly & Holden, 2006). Britain has between 9-15% of Europe's total peatland area. The total area of peatland habitats in the Britain is at this point in time it is estimated at over ca. 22,000km^2 (Bain et al., 2011; Worrall et al. 2011). In the Britain peatland habitat covers about 9.5% of the land area, with the majority in Scotland (Bain et al., 2011). In the Britain total intact deep peat covers an area of 17,125km^2. The vast majority of deep peat is found in Scotland (17,269 km^2) with smaller amounts found in England (6,799 km^2), NI (1,700 km^2) and Wales (706 km^2) (Worrall et al., 2011).

In Finland and Scandinavia or Norway and Sweden, mires are a major component of the natural landscape. They extend over vast areas and may cover ca. 10 to 30% and in the north as much as 50% of the terrain (Pakarinen, 1995). As in Ireland and Britain climate plays a key role in the prevalence of mires. Evaporation remains below precipitation during the growing season and a large proportion of the spring melt is retained in the mires (Gore, 1983 b). Up to 40% of the mires found on land which was above sea level during the last Ice Age are ca. 8000 years old B.P. (Gore, 1983 b).

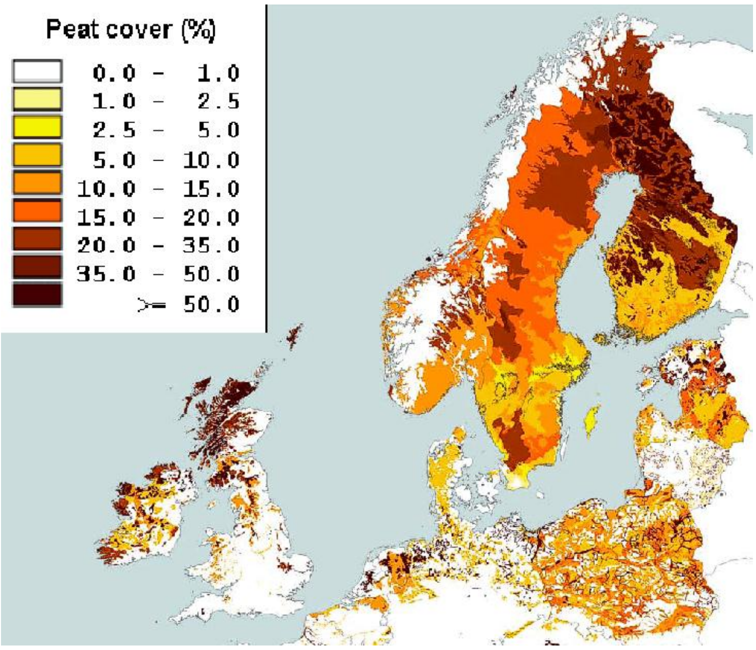
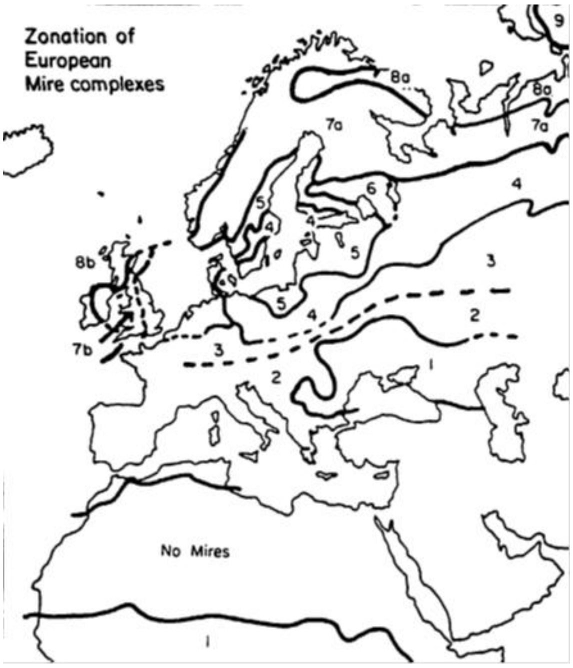


Fig.4 The relative cover (%) of peat and peat-topped soils in the SMUs of the European Soil Database (Montarella et al., 2006)

Post glacial isostatic uplift lifted the younger lands of Fennoscandinavia above the Baltic Sea. From a global point of view the mires of Fennoscandinavia are among the most diverse and well preserved in the world (Gore, 1983 b). Finland is the mire-richest country in the world with mires covering ca. 30–35 % of the total Finnish land area (Seppä, 2002; Turunen, et al., 2002; Vasander et al., 2003)**.** The total area of peatlands with a peat layer over 30 cm thick, and exceeding 20 ha, is 5.1 million hectares. In Sweden the peat covered land area 10.4 million ha. Most of the mires are concentrated in the flatter elevations of Northern Sweden (Vasander et al., 2003). Norway is home to some 3 million ha of peatlands. The majority of the mires are found below the timberline and cover ca. 2 million ha. The remaining 1 million ha of mires are found in upland areas (Gore, 1983 a).

The Peatland types of NW Europe

Beyond the classifications of ombrotrophic or minerotrophic mires can be further subdivided based on characteristics such as ecology and vegetation, stratigraphy and development history (Doyle, 1990). The main peatlands found in Ireland and Britain are 1) Raised bogs, 2) Mountain blanket bogs, 3) Atlantic/Western blanket bogs, 4) fens (Bain et al. 2011; Doyle, 1990; Douglas et al., 2008; Renou Wilson et al., 2011). The fens can be further subdivided into lowland fens; and, upland flushes, fens, and swamps. (Worrall et al. 2011) In the Britain, blanket and raised bogs constitute 95% of all peatland habitats (Bain et al., 2011). A rich diversity of different mire ecosystems are found across Fennoscandinavia has as result of the variations in latitude, altitude, oceanity and geological conditions within the three countries (Gore, 1983 a). Figure 5 shows the zonation of the different mire complexes.



Zones: (1) Valley bogs and flooded fens, (2) Tertiary valley mires, (3) Continental raised bogs, (4) Plateau raised bogs, (5) Concentric raised bogs, (6) Eccentric raised bogs, (7a) Aapa mires, (7b) Ridge raised bogs, (8a) Palsa mires, (8b) Blanket bogs, (9) Artic bogs. Zones 3-6, 7b & 8b refer mainly to ombrotrophic bogs

Fig. 5 The spatial zonation of European mire complexes (from Moore & Bellamy, 1974)

The climate of southern Fennoscandinavia and the west coast of Norway is like that of the British Isles oceanic in nature and with mild and wet winters due to its latitude and the influence of the Gulf Stream (Gore, 1983 b). The climate is suitable for the formation of raised bogs. Figure 5 shows the distribution of the raised bogs in this region of Fennoscandinavia. As one moves north in Fennoscandinavia the raised bogs give way to firstly the loosely defined mire complexes then the aapa mires and in the high latitudes palsa mires and tundra mire ecosystems are typical (Charman, 2002; Moore & Bellamy, 1974). Excluding blanket bogs, Finland has almost all of the mires commonly associated with the boreal region. Various raised-bog, plateau bogs, concentric bogs, eccentric bogs and Sphagnum fuscum bogs, southern aapa mires, main aapa mires, northern aapa mires, palsa mires and orohemiarctic mires (Seppä, 2002; Turunen, et al., 2002). In general however raised bogs characterise southern Finland, open aapa mires and sloping fens northern Finland. In the far north of Lapland, palsa mires form the northernmost, periglacial mire complex type in Fennoscandinavia (Fig. 5, 8a) (Seppä, 2002).

Raised bogs

Raised bogs are peatlands that are characterised by a distinct convex peat dome where peat can obtain thicknesses of 9 to 12 metres (Charman, 2002; Worrall et al. 2011). Raised bogs initial peat formation occurred in post-glacial lakes or saturated hollows in the landscape following the Midlandian Cold stage. These poorly drained hollows filled with marl and glacial drift accumulated plant material and debris over time. The accumulation of debris decreased the water dept. The lowering of the water level encouraged the encroachment of the lake and the subsequent further diversion of water. This resulted in raising the local water tables and which initiated plaudification. Terrestrial eutrophic and /or mesotrophic peats then developed. The lateral and vertical accumulation of peat and plant debris raised the mire above the ground water level. Plants were now dependant on rainfall as the main source of nutrients. Under these conditions Sphagnum mosses and terrestrial ombrotrophic peats became dominant. The normal process of peat formation resulted in the raising of the dome and the formation of the raised bog (Worrall et al. 2011).

In Ireland raised bogs are found in the Central Plain of Ireland (Douglas et al., 2008; Hammond 1981). It is low-lying and flat with Carboniferous limestone bedrock. Glacial drift deposits have formed a gently undulating topography. Given the nature of the topography drainage is poor. The glacial drift and waterlogged conditions have given rise to the raised bogs found in the midlands today (Hammond 1981). The largest complex of lowland raised bogs in England can be found in Yorkshire in Thorne, Hatfield, and Crowle Moors (Worrall et al. 2011).

The hemiboreal and south boreal zones of Fennoscandinavia contain what is thought to be the largest concentration of raised mire ecosystems in Europe (Fig.5, 4 & 5) (Charman, 2002; Pakarinen, 1995). The Baltic raised bogs and the Atlantic raised bogs differ in their morphology. Baltic raised bogs have a domed surface while their Atlantic equivalents have more of a plateaued surface (Gore, 1983 a). Both however may be defined as raised bogs as the centre of the bog is above the level of the mineral soil (Seppä, 2002). Both types of raised bog are ombrotrophic and share a similar morphology due to their similar development history. They both developed due to the continuous accumulation of nutrient poor Sphagnum peat which eventually lifted the bog surface out of reach of the mineral soil and water table (Seppä, 2002).

Blanket Bogs

Blanket bogs area rare ecosystem type as they are restricted to region with a hyperoceanic climate. In Europe they occur in Iceland, Ireland, Britain, coastal Norway and the Faroe

Islands (Gallego-Sala et al., 2010; Gallego-Sala & Prentice, 2012; Gore, 1983 b). Blanket bog formation requires a specific set of extreme weather conditions.

- Minimum of 1000mm rainfall
- Minimum of 160 wet days (>1mm rain).
- Mean temperature of < 15 °C for the warmest month.
- Minor seasonal fluctuation in temperature (Charman, 2002).

The development of blanket bogs is closely linked to the climatic deterioration within the post-glacial period. The presence of pine stumps in the basal peats of blanket bogs are testament to the favourable growing conditions prior to the down turn. Mountain blanket bogs began to develop prior to 4,000 B.P. The date of Blanket bog initiation differs from place to place and ranges from 4,150 - 2,150 B.P. (Worrall et al., 2011). Blanket bog formation is also closely linked to the elm decline in and human activity in the early Neolithic. The actions of prehistoric man in forest canopy clearance, grazing and firing of the vegetation led to the removal of forest cover and an associated decrease in evapotranspiration and an increase in water run-off. These human activities and a concurrent decline in northern European climate created the conditions for blanket bog initiation (Charman, 2002). Under these wet condition the process of plaudification occurred (Westhoff, 1990). The mid-Holocene climatic deterioration in tandem with the activities of Neolithic and Bronze Age man are thought to be responsible for the spread of both upland and Atlantic blanket bogs. The removal of tree cover resulted in the deterioration of the soils and the development of peat in waterlogged conditions (Gore, 1983 b).

In the West of Ireland and Scotland the climate is characterised by its extreme oceanity here the precipitation levels are so high that blanket bogs have developed in the lowlands. This hyper-oceanic variant of blanket bog known as Atlantic or western blanket bogs are found in the Atlantic coastal plains of Western Ireland and Western Scotland. This habitat is unique to these countries and as a result its protection is of global significance (Douglas et al., 2008; Westhoff, 1990). In Ireland Atlantic blanked bogs are characterised by species such as Sphagnum subsecundum and Schoenus nigricans (Pakarinen, 1995).

The development of Atlantic blanket bog is thought to have been caused by the same factors that initiated the formation of the other blanket bogs. The development of Atlantic blanket bogs is strongly linked to the hyperoceanic climatic conditions of the Atlantic seaboard. Here high annual humidity, cool summers, high annual precipitation (>1250 mm yr^{-1}) and persistently wet weather (> 225 rain days yr^{-1}) enables the development of blanket bog at low altitudes (Gore, 1983 b). Although oligotrophic Atlantic blanket bogs are fed by the nutrient rich rains of the Atlantic coast. As a result species such as *Schoenus nigricans* and *Molinia caerulea* which are generally associated with more nutrient rich mires are dominant (Gore, 1983 b).

Mountain blanket bogs are distributed as the name suggests in the upland regions of Ireland and Britain at altitudes < 152 m (Douglas et al., 2008; Westhoff, 1990). In Ireland upland blanket bog formation is common in the upland areas of the west for example in the mountain ranges in Donegal, Galway and Kerry. In regions to the east where precipitation is less blanket bog is restricted to mountainous areas (Hammond 1981). Most peatlands in Britain are blanket peats which occur on flatter parts of the uplands. Britain contains ca. 13% of the world's resource of blanket bog (Worrell et al. 2011). The largest area of blanket bog in Britain is in northern Scotland in the Flow Country (Worrall et al. 2011). Raised bogs can be found along the Atlantic Coast of North West Europe (Douglas et al., 2008).

Fens

Fens are minerotrophic ("mineral-fed") mires which develop where the mire vegetation is still in contact with the enriched mineral water. The water source may be the result of water run-off from the mineral soil or seepage. Fens may develop into bogs with time (Raeymaekers, 1999). Fens may be divided into basin, valley, floodplain and sloping fens and have varying nutrient statuses and pH of the ground water (Worrall et al. 2011). The nutrient status of fens varies depending on their location and local geology (Worrall et al. 2011). Fen peatlands can be found throughout Ireland and Britain in waterlogged areas which are fed by calcareous water supply (Doyle 1990). The Fen mire ecosystems commonly found in Fennoscandinavia such are pine fens, eutrophic fens, spruce swamps and open fens where trees are absent (Seppä, 2002). Pine fens as the name suggests are forested mires. Pine and *Betula nana*, *Calluna vulgaris* and *Ledum palustre*, dominate their vegetation. The peat layer may be several metres thick and formed by brown *Sphagnum* species (Seppä, 2002). Eutrophic fens are restricted to a few areas in Finland. They have rich vegetation and a high pH and nutrient levels due to the carbonate nature of the underlying bedrock (Seppä, 2002). Spruce swamps are forested mires which are characterised by a dense canopy of spruce with an understory of birch and alder. In the field layer tall grasses dominate. Peat layers are well decomposed and usually ca. < 100 cm (Seppä, 2002).

The Boreal Mire Complexes

The boreal mires found in Fennoscandinavia are mire complexes where bogs and fens occur next to each other. These boreal mire complexes are generally flat landscape features, which are shaped by the thawing in spring and the compression of snow cover in winter (Raeymaekers, 1999). Raised bogs, aapa mires and palsa mires are the major mire types found across Fennoscandinavia. In many regions the combinations of environmental gradients varies in different parts of the mire. Here are composed of a number of mire types

(Seppä, 2002). These mire complexes are named after the dominant peatland type in the region (Charman, 2002). They have a regional distribution and have distinct differences in their peat stratigraphy and general arrangement of vegetation types typical of either ombrotrophic or minerotrophic mires (Pakarinen, 1995).The regional differences between climatic conditions in maritime and continental regions means that it is problematic to use vegetation or nutrient status to compare mires in Ireland the Britain and Fennoscandinavia. For example ombrogenous blanket bogs of the oceanic flank of Ireland characterised by *Sphagnum subsecundum* and *Schoenus nigricans*, are comparable to the *Schoenus* fens of Sweden and Estonia (Seppä, 2002).

Aapa mire complexes

North of this bog zone aapa mire complexes dominate the northern boreal zone in Finland, Sweden, Norway, central and northern Sweden and north western Russia (Fig. 5, 7a) (Pakarinen, 1995; Seppä, 2002). Mixed mire complexes exist in the transition between to aapa mire dominated mire complexes. Here northern raised bogs can be found alongside southern aapa mires (Pakarinen, 1995). Appa mires are concave and characterised by narrow elongated ridges of peat known as 'strings' which are intertwined amidst hollows and pools known as 'flark' (Charman, 2002). This 'flark-string' pattern is one of their defining features. The mineral rich run-off that collects in the pools and hollows means that the flarks are characterised by fen vegetation. Oligotrophic-ombrotrophic *Sphagnum* bogs are typically found in the margins or locally on peat ridges (Pakarinen, 1995). Around aapa mires where the peat may be shallow bog woodlands can become established (Raeymaekers, 1999). The centre of aapa mires lies below the level of the surrounding mineral ground giving them a flat or concave shape (Seppä, 2002).

The spring thaws and cold winters are vitally important in the formation of aapa mires microtopography. In late spring aapa mires become flooded by rapid snow melt. This keeps the pools minerotrophic and so prevents *Sphagnum* species from establishing and as a result the hummocks that are associated with raised bogs are not formed (Raeymaekers, 1999). The appearance of aapa mires changes as one moves north and the shaping forces of spring thaw and winter freeze become more influential. In the middle of the boreal area aapa mires form homogenous lawn like fens. While further north the strings and flarks that are associated with aapa mires are predominant (Raeymaekers, 1999). In winter the flooded pools become freeze over. As the ice expands horizontally it pushes the strings upwards giving the aapa mires there characteristic shape (Seppä, 2002).

Palsa mires

Further north again the influence of freezing temperatures and frost action becomes increasingly more important. In these northern expanses minerotrophic aapa mires give way to ombrotrophic palsa mires (Charman, 2002; Raeymaekers, 1999). Palsa mires are the northernmost type of boreal mire complexes in Fennoscandinavia and are found between the latitudes 69-70 ° N (Fig. 5, 8a) (Pakarinen, 1995).They occur in continental climates in northern Fennoscandinavia, Siberia and northern Canada near the transition from forest to tundra (Pakarinen, 1995).

The defining feature of palsa mires is the 'palsa mound'. The palsa mound forms in areas where the insulating snow cover in winter is thin. Under these conditions a frozen core of peat or silt with thinner layer of ice and small ice crystals develops (Raeymaekers, 1999, Seppä, 2002). If the summer warmth is unable to melt the frozen core then it will bulge above the level of the surrounding soil forming a palsa mound (Raeymaekers, 1999, Seppä, 2002). Through successive winters the frozen peat core continues to grow as the growing mound means that the thin snow cover becomes thinner every year. Palsa mound can reach a height of 3-4 m. The mound increases to the point where the peat layer at the top of the mound dries up and disintegrates. The palsa mound collapses to form a ring (Raeymaekers, 1999). The vegetation of the palsa mires resembles the plant communities of the flarks of the northern aapa mires. Typical species of the wet surfaces include *Sphagnum lindbergii*, *Carex vesicaria*, *C. rotundata*, and *C. rostrata*. The vegetation of the palsa mound are suited to drier conditions and include species such as *Betula nana*, *Empetrum nigrum*, *Rubus chamaemorus*, lichens and, on the lower slopes, *Sphagnum fuscum* (Seppä, 2002).

Section 2

The effect of pristine peatlands in North West Europe on Climate change

Climate Change and the Greenhouse gases

Globally climatic changes have been observed worldwide over the last few decades. Over the course of the 20th century the global average surface temperature and the ocean heat content have increased. The global average sea level has risen and snow cover and ice extent have decreased (IPCC, 2001; IPCC, 2007). These changes are very likely the result of global warming which is likely to have occurred as a direct result of the anthropogenic release of greenhouse gases (Malhi & Wright, 2004; IPCC, 2007). Greenhouse gases play an important role in preventing heat from escaping from the Earth's surface. Any change in the atmospheric concentration of greenhouse gases will have a strong impact on global climate; without greenhouse gases, scientists estimate that the average temperature on Earth would be ca. 30 °C cooler (IPCC, 2007).

Three long lived greenhouse gases namely carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are the most important drivers of climate change (Forster et al., 2007). The most important of these is CO_2. Its global atmospheric concentration has increased by 36 % from a pre-industrial value of about 280 ppm to 379 ppm in 2005 (Forster et al., 2007). The concentrations of the other important greenhouse gases in the atmosphere namely methane (CH_4) and nitrous oxide (N_2O) have been increasing since the onset of the industrial revolution (Forster et al., 2007). This increase is believed to be as a direct result of the burning of fossil fuels and land use change (IPCC, 2007; Lund et al. 2009). It is believed that the pervasive changes in the earth's atmosphere are to blame for the significant warming of both land and ocean that has occurred over the last 50 years (IPCC, 2007). If current trends continue the concentration of CO_2, CH_4 and N_2O in the atmosphere is predicted to continue to increase with the concentration of CO_2 predicted to surpass 450 ppm by 2100 (IPCC, 2007). The concentration will be even greater if the carbon sink in terrestrial ecosystems continues to be depleted by land use change (Clark, 2004; Lal, et al., 2012). An increase in the atmospheric concentration of CO_2 will cause a positive feedback on global climate change (Betts et al., 2004; Friedlingstein et al., 2010; Lewis, 2006).

Pristine peatlands and the Greenhouse gases

For the purposes of both this review the definition of 'pristine' used by Worrall et al (2011) will be used. "Pristine will be used to define an area in which there is no management at the time during, or preceding, the study that could affect the peat. Pristine does not mean that the site has been unaffected by external factors such as climate change or atmospheric deposition". Peat forming ecosystems remove CO_2 from the atmosphere through the process of photosynthesis. The partially decomposed remains of the plants are stored in the form of peat due to the restricted aerobic decay caused by permanent waterlogging. If undisturbed this carbon sink may sequester carbon and store it for periods of up to thousands of years (Blodau, 2002; Gorman 1991). Peatlands in North West Europe are sinks for CO_2 and have been removing it from the atmosphere for the last 10,000 years since the Last Ice Age (Byrne et al. 2004; Mäkilä & Saarnisto, 2008). While the peat component of peatlands is the most important long term carbon sink carbon is also stored in biomass, litter, mineral subsoil layer and water (Parish, et al., 2008). This ability to store large amounts of carbon over long periods of time makes peatlands the most important long term carbon sink in the terrestrial biosphere and they are second only to the ocean in terms of the most important carbon sinks on earth (Holden, 2005; Parish, et al., 2008). When all of the carbon pools within peatlands are considered they contain disproportionally more organic carbon than the other terrestrial ecosystems. Covering a mere 3% of the world's terrestrial surface, peatlands contain 550 Gigatonnes (Gt) of carbon. This is equal to 30% of all soil carbon, as much carbon as all terrestrial biomass, and two times the carbon sink of all forests in the world (Holden, 2005; Limpens et al., 2008; Parish, et al., 2008). The ability of peatlands to store carbon for long periods of time means they are capable of slowing the rate of climate change. Peatlands are much more efficient at storing carbon than other terrestrial ecosystems because even the most productive non-peatland ecosystems reach a point where carbon capture plateaus and the total amount of carbon stored in the soil and vegetation levels off. In peatlands carbon may continue to grow for thousands of years as the peat deposit accumulate (Baird et al., 2011, Holden, 2005). The peatlands of the northern hemisphere alone store between 250 - 450 Gt of carbon, which is equivalent to about 75% of the preindustrial mass of C stored in the atmosphere (Moore et al., 2002; Strack, 2008).

Peatlands however are responsible for producing all three of the most important long lived greenhouse gases CO_2, CH_4, N_2O (Roulet et al., 2000; Sirin & Laine, 2007). The same wet and anoxic conditions that lead to the slow decomposition and so the sequestration of CO_2 also cause peatlands to be significant emitters of the potent greenhouse gases CH_4 and in some cases also of nitrous oxide N_2O (Byrne et al. 2004). Wetlands are the largest source of atmospheric CH_4 surpassing all anthropogenic emissions while two thirds of N_2O emissions to the atmosphere come from soils (Byrne et al. 2004; Smith et al., 2003; Strack et al. 2008). Peatlands have played an important role in global CO_2 and CH_4 atmospheric concentrations throughout the entire Quaternary period. Peatlands have acted as either a mediator or as a positive feedback for the atmospheric change through their expansion and contraction during interglacial periods (Sirin & Laine, 2007; Zicheng, et al., 2009). Although the absolute quantities of CH_4 and N_2O being emitted from peatlands are small in comparison with the sequestration of CO_2, they are far more effective greenhouse gases as they are better at absorbing infrared radiation and thus have a greater global warming potential than CO_2 (IPCC, 2007; Smyth et al., 2003). Greenhouse gases transmit visible light but absorb strongly infrared and near-infrared light; they thereby trap heat in the troposphere and cause global warming (IPCC, 2007; Soloman et al. 2007). The ability of a greenhouse gases to move the earths energy balance away from its normal state through positive warming of the troposphere is often evaluated in terms of its radiative forcing (Forster et al.; 2007). The different GHG have different radiative forcing's and different warming potentials for example 1 kg of CH_4 has a warming potential 23 times greater than 1 kg of CO_2, over a 100-year period, while the warming potential of 1kg of N_2O is nearly 300 times greater over the same period (Smyth et al., 2003). However a further complicating issue in quantifying the GWP of different GHG is that all gases have a different lifetime in the atmosphere and so the effects of all three gases depend on the time horizon under consideration (Joosten & Clarke, 2002). The GWP methodology is one method that is used to relate the radiative forcing of different GHG over a specified time horizon. Emissions of CH_4 or N_2O can then be converted into a CO_2 equivalent emission and there effect on climate then can be compared (Sirin & Laine, 2007).

Carbon cycling in pristine peatlands

In order to determine whether a peatland is acting as a net carbon source or sink the balance between the uptake of CO_2 form the atmosphere through photosynthesis and the loss of C back to the atmosphere must be calculated. The balance between all of the inputs and outputs is known as the net ecosystem exchange (NEE) of CO_2 (Strack et al., 2008; Worrall et al., 2010; Worrall et al., 2011). Peatland ecosystems assimilate CO_2 through the process of photosynthesis. A portion of this fixed CO_2 is returned to the atmosphere through autotrophic respiration. A further portion of the CO_2 fixed through photosynthesis is released through heterotrophic respiration during the decomposition of plant litter. Under normal conditions the anaerobic, cool and nutrient poor conditions of peatland ecosystems result in low decomposition rates. As a result of this the C fixed through photosynthesis exceeds that lost through autotrophic and heterotrophic respiration and the build-up of C in the form of the organic peat (Blodau, 2002; Limpens et al., 2008; Lund, 2009). Deeper down in the anoxic catotelm anaerobic metabolic processes such as denitrification, sulphate reduction and methanogenesis take place. These processes result in the release of N_2/N_2O, H_2S and CH_4. The general deficiency in nutrients in mire ecosystems means that nitrate and sulphate concentrations are low (Blodau, 2002; Minkkinen, et al., 2008).

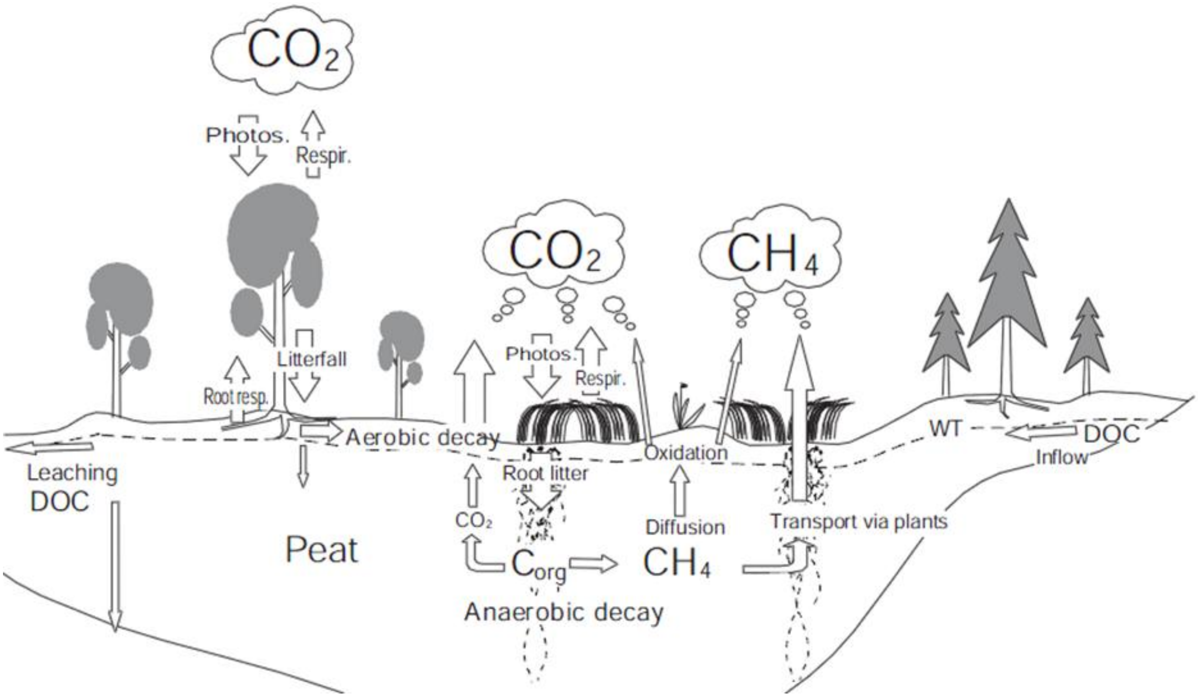


Fig. 6 The carbon cycle in pristine mire ecosystems (Minkkinen, et al., 2008)

Another important gaseous loss of C to the atmosphere is CH_4 however it is often ignored in C budget studies because it represents a relatively small percentage (<10%) of the total C budget of peatlands (Worrall et al., 2010; Worrall et al., 2011). Methane is the dominant product of the anaerobic processes in most peatlands (Lund, 2009). The depth of the water table, temperature, pH, substrate availability and quality and site specific microtopographic variations have been identified as the most important environmental controls of CH_4 production in peatlands (Joabsson et al., 1999; Lund, 2009). CH_4 has a low solubility in water (23–40 mgl^{-1} at 0 – 20°C), which means that CH_4 produced in waterlogged anoxic environments can escape through the sediment through a number of pathways (Joabsson et al., 1999; Lund, 2009). The CH_4 gas may diffuse slowly upwards through the peat giving methanotrophic bacteria the opportunity to oxidise CH_4 to CO_2 (Lund, 2009). Changes in the temperature or changes in the atmospheric pressure or a decrease in pore-water caused by fluctuations in the water table in deep peat may cause the CH_4 to come out of solution. CH_4 bubbles may form in air pockets in the peat or where the diffusion of CH_4 is blocked beneath a layer of woody peat or ice (Baird, et al., 2004; Tokida, et al., 2007). In Northern peatlands such as the aapa mires CH_4 may become trapped beneath layers of ice and may be released in large quantities during the spring thaw (11% of the annual total) (Tokida, et al., 2007). Under these conditions large volumes of CH_4 (>40 g CH4 m^{-2}) may build up within the peat and may be released to the atmosphere in events lasting from minutes to hours (Baird, et al., 2004). These CH_4 bubbles move through the peat too quickly for oxidation to take place. Another important pathway of CH_4 to the atmosphere is through vascular plants. Under anoxic conditions wetland plants may transport O_2 through their stems roots and rhizomes via specialised tissue called aerenchyma. CH_4 may move through this specialised vascular tissue to the atmosphere and thereby avoid oxidation in the acrotelm (Joabsson, et al., 1999; Lund, 2009). The transport of O_2 and CH_4 through plants however has a twofold effect on net CH_4 emissions in peatlands. On the one hand the release of O_2 into the anoxic peat layers inhibits the production of CH_4 by the methanogenic bacteria. It is thought that the oxidation of CH_4 by Methanotrophic bacteria may also be stimulated by O_2 release in the rhizosphere. On the other hand, however the net effect of the facilitated transport of CH_4 to the atmosphere is that the net flux of CH_4 to the atmosphere is increased (Joabsson, et al., 1999).

An important pathway for C to leave peatlands and one that is often not considered when C balances are being studied is through the water outflow from peatlands. Peatland streams are often supersaturated with CO_2 and CH_4 (Billett & Moore, 2008; Dinsmore et al., 2010). These gases come out of solution relatively quickly and are returned to the atmosphere (Worrall et al. 2011). In an ombrotrophic bog catchment in in Canada it was observed that the loss of CO_2 via evasion is significantly more important than loss by runoff (Billett & Moore, 2008). In a study on an ombrotrophic peatland in Scotland (2007-2008) it was found to be a net sink for GHGs (0.035 gCO_2 eq^{-2} ha^{-1} yr^{-1}) and C (0.007 g C ha^{-2} yr^{-1}). The GHGs from the stream surface returned 12% of CO_2 equivalents captured by NEE to the atmosphere while the often studied terrestrial emissions of CH_4 and N_2O combined returned only 4% (Dinsmore et al., 2010).

Apart from the gaseous loss of C to the atmosphere as CO_2 and CH_4 significant C can be lost through the fluvial system as dissolved organic carbon (DOC), dissolved inorganic carbon (DIC) and particulate organic carbon (POC) (Dawson et al., 2004; Limpens et al., 2008; Strack et al., 2008; Worrall et al., 2011). DOC is commonly studied because of its connection to water quality (Armstrong et al., 2010; Wallage et al., 2006; Wilson et al., 2011). DOC along with DIC and POC are not GHG themselves they are however significant as they are an important component of the C balance of peatlands (Limpens et al. 2008; Strack et al., 2008). DOC as it is believed to be the dominant component of the C aquatic flux (Strack et al., 2008; Worrall et al., 2011).

DOC is formed as a result of the incomplete decomposition of organic matter. It is composed of a mixture of the remains of plants, animal and microorganisms and covers a continuous size spectrum ranging from free monomers, fulvic and humic acids to aggregates which form larger particles (Dawson et al., 2004; Strack et al., 2008; Wallage et al., 2006) . Saturated peatlands with lower decomposition rates therefore produce greater concentrations of DOC. The DOC concentration and fluvial flux is strongly related to the size of the carbon pool in the soil and % peat cover along with other catchment variables such as discharge, precipitation, slope and catchment size (Dawson et al., 2004).

Aquatic C losses from peatlands usually occur as a result of peatland saturation. Losses therefore happen sporadically and when they do occur they are sudden due to heavy rain or snow melt. This is especially true for ombrotrophic peatlands where water movement is severally restricted in the lower peat layers (Limpens et al., 2008). The DOC export from fifteen Swedish peatland streams was measure at 2-10 g C m^{-2} yr^{-1} while DIC was just 0.2 -2 g C m^{-2} yr^{-1} (Limpens et al., 2008). The aquatic export from temperate and boreal peatlands ranges from 1-50 g DOC $m^{-2}yr^{-1}$ which represent ca. 10% of their the total C release (Limpens et al., 2008). In a study on the C content of a Scottish upland peat stream the total C lost from the entire catchment totalled 54,140 kg C yr^{-1}. The loss of CO_2 from the stream surface accounted for 34% of the total C loss while the DOC accounted for 57% of the loss (Hope et al., 2001). Peatland streams are therefore an important pathway for catchment greenhouse gas (GHG) and carbon (C) losses (Dinsmore et al., 2010).

The total estimated loss of DOC from northern peatlands may be 20 tonnes C km^{-2} yr^{-1}. To account for the total aquatic C flux the quantities of C leaving the peatland in the form of DOC, POC, DIC, dissolved CO_2 and CH_4 and CO_2/CH_4 degassing from the stream surface should be considered (Worrall et al. 2011). However while accounting for the C NEE of peatlands may give an indication of whether they are acting as a net carbon source or sink quantifying the impact of the various fluvial flux components from peatlands on climate change is extremely difficult. In order to understand the significance of fluvial flux in terms of radiative forcing and climate change the amount of C that ends up in the atmosphere has to be known (Worrall et al. 2011). The gaseous CO_2 and CH_4 component are likely to come out of solution relatively quickly and so return to the atmosphere quickly. The ultimate fate of DOC and POC is harder to discern. Reductions in DOC flux have been observed downstream from peatlands but whether this reduction is the result of loss to the atmosphere or the result of flocculation and precipitation of organic carbon remains uncertain (Worrall et al. 2011). It is possible that a portion of the C flux of peatland streams may be stored within the aquatic environment in the river, floodplain and estuary or within marine sediments (Worrall et al. 2009; Worrall et al. 2011). Then again it may be the case that the vast majority if POC and DOC flux is oxidised and so returns vast quantities of C to the atmosphere and in doing so it may decreases the GHG sink potential of peatlands (Worrall et al. 2011).

Nitrogen cycling in pristine peatlands

Nitrous oxide is a potent greenhouse gas and also causes destruction of stratospheric ozone (Joosten & Clarke, 2002; Smith et al., 2003). Soils are globally a major source of N_2O (Lund, 2009; Smith et al., 2003). N_2O is produced through denitrification when nitrate (NO_3–) in the soil is broken down in anoxic conditions by bacteria (Natural England, 2010; Smith et al. 2003). In peatlands and principally in ombrotrophic acidic peat, the supply of NO_3– and nutrients in general is very low (Drosler et al., 2008; Limpens, et al., 2006). NO_3–is supplied to peatlands through nitrification of NH_4 +, by atmospheric deposition, or through the fixation of N (Smith et al., 2003). Nitrification is hampered in peatlands by the cold, anaerobic and acidic conditions. NO_3 – is intercepted by microorganisms, sphagnum and other mosses, and vascular plants supplied by deposition or nitrification before it can pass into the anaerobic catotelm or microsites where denitrification takes place (Limpens, et al., 2006). Denitrification and N_2O emissions directly from pristine ombrotrophic bogs and indirectly through drainage through streams are low with rates ranging from 0 to 0.4 g N m^{-2} $year^{-1}$ (Joosten & Clarke, 2002; Limpens, et al., 2006; Natural England, 2010). Sometimes ombrotrophic mires may even consume N_2O due to the reduction of N_2O to dinitrogen (N_2) under conditions of severe oxygen deficiency (Joosten & Clarke, 2002). In mixed mires or fens, where conditions are minerotrophic the supply of inorganic N is greater and the higher pH does not inhibit nitrification of NH4+, the amount of N lost through denitrification may be higher. Likewise, water table drawdown may enhance nitrification and subsequent denitrification by stimulating N mineralization (Drosler et al., 2008; Limpins et al., 2006). None the less pristine natural mires are considered to be neutral with respect to N_2O (Byrne et al., 2004; Drosler, et al., 2008; Worrall et al. 2011). N_2O emissions from pristine peatlands in NW Europe therefore play a minimal role in the climate system (Dinsmore et al., 2010; Drosler et al., 2008; Strack et al. 2008).

Are pristine peatlands North West Europe net GHG sinks or sources?

It is known that peatlands have been removing and sequestering atmospheric CO_2 for thousands of years. Peatlands are also however important emitters of CH_4 and in some cases also of nitrous oxide N_2O both of which have a greater global warming potential than CO_2 (Byrne et al. 2004; Smyth et al., 2003). This raises the question are pristine peatlands in Ireland, Britain and Fennoscandinavia net sources or sinks of GHG. Sweden and Finland have the greatest expanse of pristine peatlands in the EU. Minor areas are found in Norway, Ireland, Scotland, England, Germany and Poland (Byrne et al., 2004).

Pristine peatlands can switch between being a net GHG sink and sources form year to year depending on variations in annual climatic conditions such as temperature, rainfall and the level of the water table (Dinsmore et al., 2010; Worrall et al., 2010). The position of the water table determines the oxic-anoxic ratio of the peat profile and the temperature influences the rate and extent of the various biogeochemical processes that determine the GHG flux (Sirin & Laine, 2007). In Fennoscandinavia Carbon gas fluxes of the boreal peatlands can vary depending on factors such as the timing of the spring thaw and the geographical location of the site (Maljanen et al., 2010).

A study by Lund (2009) in a temperate climate blanket bog in southern Sweden found that it was a small sink for atmospheric CO_2 (-8.7 ± 12.2 g C m^{-2} yr^{-1}) over three years and a small source for CH_4 (3.1 ± 1.5 g C m^{-2} yr^{-1}) over two years while N_2O was found to be negligible. It was found that over a time scale of a few hundred years the decrease in the C sink and the decrease in the CH_4 production may cause a negative feedback to climate change sink CH_4 is a stronger GHG than CO_2 (Lund, 2009).

In a recent review of GHG balances in the Nordic countries Maljanen (et al., 2010) found that the literature shows that pristine boreal peatlands may act as net sources or sinks for CO_2 and CH_4 depending on the weather conditions and that the different peatland ecosystems differ in their sensitivity towards climatic variability. The review included CO_2, CH_4 and N_2O fluxes for Swedish minerotrophic and ombrotrophic sites, Swedish subarctic mires and subarctic palsa mires, fens in southern Finland and minerotrophic peatlands in northern Finland. The studies under review showed that the pristine ombrotrophic and minerotrophic peatlands have most often been sinks for CO_2 with ombrotrophic peatlands accumulating more peat than minerotrophic ones (Maljanen et al., 2010). The general trend from the Fennoscandinavian peatlands was that the GWP (CO_2 eq) of the three GHG over a 100 year period was positive for minerotrophic peatlands and ombrotrophic peatlands due to the high radiative forcing of CH_4 emissions. The N_2O levels from the GHG study's on both ombrotrophic and minerotrophic peatlands recorded low levels of N_2O. N_2O emissions from pristine undrained peatlands are negligible as active peat forming systems usually have saturated soils and peats. The water table level determined the N_2O emissions from peatlands. Emissions may be high with a low water level and high oxidation of the peat profile but water saturated peat may even consume atmospheric N_2O in the absence of O_2 (Maljanen et al., 2010).

A similar study carried out by Drewer et al (2010) compared the GHG flux of CO_2, CH_4 and N_2O from a low lying Auchencorth Moss an ombrotrophic peat bog in Scotland to that of a minerotrophic fen in northern Finland over a 100 year time period. It was calculated that the Scottish bog was a GHG sink of −0.0321, −0.0490 and −0.0321 CO_2 eq g ha^{-2} $year^{-1}$ in 2006, 2007 and 2008. CO_2 was found to be the most important GHG. CH_4 was found to be the dominant GHG in the Finnish fen which caused the site to be a net GHG source of +0.0485 and +0.0431 g CO_2 eq ha^{-2} $year^{-1}$ in 2006 and 2007. The fen was found to be gaining nitrogen while the bog was losing nitrogen. The GWP of the bog was negative while that of the fen was positive over the investigated time period (Drewer et al., 2010).

Importantly however when the GWP of the bog and fen were investigated over a 500 rather than a 100 year period it was found that the bog remained a GHG sink due to the fact CO_2 was the dominant GHG. Interestingly using the 500 year time horizon to calculate the GWP of the Finnish fen where CH_4 was the dominant GHG reduced the GHG source strength of the site to about a quarter for 2006 and reduced it to only about a tenth in 2007. This is the result of the different radiative efficiencies and the shorter residence time of CH_4 in the atmosphere compared to CO_2 (Drewer et al., 2010). N2O was found to be negligible in the GWP calculations. The use of even longer time periods which would reflect the lifespan of the wetlands better on the scale of thousands of years it is highly likely that the peatlands have had a cooling effect on climate (Drewer et al., 2010).

In another review of six studies examining the GHG fluxes of mixed aapa mires, minerotrophic fens, and ombrotrophic fens in Sweden and Finland and nutrient rich fens in Britain it was found that most wetlands in their natural state are sinks of CO_2 and carbon and sources of CH_4 over a 100 year time span (Byrne et al., 2004). Once again the emissions of N_2O were in general negligible for the pristine mires. The different peatlands were found to vary between a small sink and a moderate source of GHG to the atmosphere. The CH_4 were once again found to be the determining factor in whether a peatland was ultimately a sink or a source. Once again the choice of the 100 year time span meant that the GWP of the peatlands was heavily influence by the CH_4 emissions (Byrne et al., 2004).

All of the aforementioned studies and many others have focused on CO_2 and CH_4 fluxes in peatlands and have failed to take into consideration the fluvial C losses in their C flux and GHG calculations (Byrne et al., 2004; Christensen et al., 2007; Drewer et al., 2010; Worrall et al., 2003). Two recent studies on at Moor house in Northern England and on Auchencorth Moss a Scottish ombrotrophic bog that was previously studied by Drewer et al (2010) have completed comprehensive carbon budgets and GHG budgets for the time horizon of 100 years which included fluvial losses (Dinsmore ta al., 2010; Worrell et al., 2009). The study by Worrall et al (2009) was carried out over a 13 year period (1993- 2005) on an ombrotrophic peatland in Northern England showed that the total carbon balance varied between a net sink of −2 to −9.1 Mg C/ha^{-1}/yr^{-1}. The uptake of carbon by primary productivity was the most important part of the carbon budget (−17.8 Mg C ha^{-1} yr^{-1} and the second largest component was the loss of DOC from the peat profile (+3.9 Mg C ha^{-1} yr^{-1}). Direct exchanges of C with the atmosphere average −8.9 Mg C ha^{-1}yr^{-1} in the catchment. The study concluded that should the result of this study be extrapolated across all British peatlands it would result in a carbon balance of −1.2 Tg C/ yr (±0.4 Pg C/yr) (Worrell et al. 2009).

A study at Auchencorth Moss which also considered both terrestrial and fluvial fluxes of GHG found Auchencorth Moss also appeared to act as a significant net sink for GHGs in with a mean of 352 x 10^{-4} gCO_2 eq ha^{-1} yr^{-1}. Once again the dominant flux component was NEE (421 x 10^{-4} gCO_2 Eq ha^{-1} yr^{-1}). The combined terrestrial emissions of CH_4 and N_2O combined equated to only approximately 4% of NEE uptake while evasion from the stream channel equated to ca. 12% of the CO_2 equivalents captured via NEE.

The general consensus seems to be that over short time spans pristine peatlands in NW Europe are having a warming effect on the atmosphere and that over longer time spans they are having a cooling effect. This is as a result of the different accumulated radiative forcing of CH_4 and CO_2 over a given time horizon and their differing life times in the atmosphere (Sirin & Laine, 2007; Strack et al., 2008). CH_4 has a higher radiative efficiency per unit mass than CO_2; however CH_4 has a shorter atmospheric lifetime than CO_2, after 65 years in the atmosphere the remaining CO_2 in the atmosphere will generate a stronger instantaneous radiative forcing than the remaining CH_4. If time periods relevant of millennia are considered then CO_2 that is removed from the atmosphere and sequestered in peat is prevented from warming the atmosphere. CH_4 emissions cause a quick strong warming while CO_2 uptake causes a slow cooling from. This means that over long tie spans pristine peatlands cause a net cooling on the atmosphere (Sirin & Laine, 2007; Strack et al., 2008). Given that the peatlands on Ireland, Britain and Fennoscandinavia are thousands of years old and the turnover time of C within an ecosystem may be millennia there is no justification to consider GHG balances over short time periods such as 100 years (Strack et al., 2008). It has been estimated that in the last 10,000 years since the last Ice Age the atmospheric carbon sequestered in peats has served to reduce global temperatures by about 1.5–2 °C (Holden, 2005).

Section 3

The effect of agriculture and forestry on the GHG flux of NW European peatlands

The current state of degradation of peatlands in NW Europe

Across Europe the foremost causes of peatland degradation and destruction have been the conversion of peatlands for agricultural use (50%), for forestry (30%) and peat extraction (10%). Other issues such as urbanisation, inundation, erosion, burning and atmospheric deposition have played a lesser but significant role in the degreadation of peatlands (Bain et al. 2011; Byrne et al., 2004; Moore, 2002;). While in more recent times the construction of wind farms and communication masts has become an issue on upland bogs (Bain et al., 2011; Renou-Wilson & Farrell, 2009).

Out of a total mire and peatland area of 617,000 km^2 in Europe, 52 % has been lost during the last century (Byrne et al., 2004). In the British Isles over 90% of the original area of raised bogs has been modified damaged or destroyed (Raeymaekers, 1999). In Britain only 3.3% and in Scotland perhaps 6% of the original raised bog peat remains in a natural state (Barkham, 1993). Most of the peatlands in Britain are degraded to the point where they no longer form peat: 16% are severely eroded, 10% are used for forestry, and 11% are affected by past peat extraction and 40% have been converted for agricultural use (Bain et al. 2011). Of the 775,000 ha of raised bog originally found in the Republic of Ireland, 86% has been lost (Raeymaekers, 1999).The total area of active raised bog has decreased by over 35% from 1995-2005. It is estimated that between 2% and 4% of this active area continues to be lost every year since then (Bullock et al., 2012; Renou-Wilson, et al., 2011). In Ireland it is thought that only 10% of the original raised bog and 28% of the original blanket peatland resource are now in a good enough condition to be considered of conservation value. (Renou Wilson et al. 2011).

In Finland extensive drainage of mires has occurred over the 20th century (Seppä, 2002). In southern and central Finland > 25% of the original pristine mires exists today. Of the original mire area more than half (~ 55%) in Finland and 15% in Sweden have been lost due to draining for forestry (Byrne et al., 2004; Maljanen et al., 2010; Vasander et al. 2003). In global terms Finland is second only to Russia in having the largest area of original peatlands that have been converted for other uses over the last century (Byrne et al., 2004). In Finland, Sweden and Norway about 10% of peatlands have been converted to agricultural uses. In Finland and Sweden 0.6% and 0.1% of the total peatland area is subject to peat extraction (Maljanen et al., 2010).

The effect of drainage on the GHG balance of peatlands

This section will focus on the two main drivers of peatland degradation i.e. the conversion of peatland for agriculture and forestry and the effects that these land uses have on the GHG flux of peatlands. Drainage is at the heart of of the three major forms of human disturbance to peatlands and drastically alters their GHG dynamics (Maljanen et al., 2010). It has been common practice during the 20th century particularly in Britain, Ireland and Finland where the draining of peatlands for agriculture and forestry was actively encouraged (Holden et al., 2004; Ramchunder et al., 2009). As we know hydrology is essential in the formation, functioning and ecology of intact mire systems. The economic utilization of bogs and fens for agriculture, forestry or peat extraction is dependent on controlling the hydrology of the system (Charman, 2002). This process usually involves some form of drainage to lower the water table and to aerate the soil so that cultivated plants can grow or peat can be extracted (MacDonald & Yin, 2001; Oleszczuk, et al., 2008; Westman & Laiho, 2003).

There are differences in technical approach between the drainage of peatlands for agriculture and drainage for forestry. In forestry the ditches are open and secondary ditches connected to the main ditches are excavated to achieve the desired water table level. The changes in the biogeochemical processes resulting from drainage are similar for both agricultural and forestry and may therefore be described together (Maljanen et al., 2010). Drains facilitate the drying of the upper peat layers by providing a route for increased surface runoff through the ditches. The effect of drainage is initially only felt in the upper peat layers due to the very low hydraulic conductivity in the catotelm. The rate at which the catotelm dries will depend on the depth and spacing of the drain system the peat composition and the amount of precipitation (Charman, 2002).

The drying of the catotelm drastically changes the peat morphology. The peat may begin to shrink and crack. As the water table is lowered the upper peat may collapse causing the bulk density in the upper 40 cm to increase by up to 63% within a few years of drainage. This results from the fact that peat is usually 90% water by mass and 300% by volume (Holden, 2004). The dry surface increases capillary action and causes the dehydration of the subsurface layers (Holden, 2004). This change in the peat structure may lead the peat to collapse and resulting in the loss of significant amounts of carbon through the fluvial system (Holden, 2004).

Vegetation succession whether intentional through the planting of crops or tree species will increase the evapotranspiration from the peats surface and so accelerate the drying of the peat and the development of shrinking cracks (Charman, 2002; Holden, 2004). New species invade the peat surface they may be purposely introduced species or wild invasive species. They replace the sphagnum mosses and sedge species which are responsible for peat formation. The new species have higher rates of evapotranspiration and accelerate the rate at which the peat dries. In peatland plantations canopy closure leads to the interception of rainfall and thus greater evaporation (Holden, 2004).

Drainage seriously alters the physical and hydraulic characteristic of peatlands and inevitably the biogeochemical processes that are responsible for the net fluxes of CO_2, CH_4 and N_2O (Holden, 2005; Oleszczuk, et al., 2008). As discussed in the previous section the sequestration of CO_2 in peat soils is as a result of the waterlogged anoxic conditions. The drainage of the peat results in a lowering of the water table and a deepening of the oxic zone thus increasing the air-filled porosity of the peat. This in turn affects microbial processes and thus decomposition rates (Holden, 2004; Worrell et al., 2011). With more available oxygen in the peat aerobic decomposition is accelerated and peat mineralisation takes place. This inevitably increases CO_2 emissions from the peatland (Charman, 2002; Oleszczuk, et al., 2008; Worrell et al., 2011). The rate of decomposition in the upper aerated zone is typically thousands of times greater than those in the deeper anoxic layers. A lowering of the water table therefore can greatly increase CO_2 emissions and switch the peatland from a C sink to a source (Gorman, 1991; Holden et al., 2004; Worrell et al., 2011). Peatlands are dense in carbon if just 2 mm of peat were oxidized annually; then this would result in the liberation of ca. 1.6 billion tonnes of CO_2 into the atmosphere. This is equivalent to 8% of current fossil fuel release (Holden, 2005). Globally 500,000 km^2 of drained peatlands release as much as 2 Gtons of CO_2 annually (Joosten, 2009).

Just as drainage of peatlands turns them from a CO_2 sink to a source it generally switches peatlands to minor sources or often net sinks of CH_4 (Maljanen et al., 2004). The increased aeration in the peat soils reduces the size of the anoxic peat layer and in doing so reduces the production of CH_4 by the anaerobic methanogenic bacteria providing a partial counterbalance to the increase in CO_2 emissions. The increase in the depth of the aerated zone also increases the oxidation of CH_4 by methanotrophic bacteria to CO_2 thus increasing CO_2 emissions and lowering CH_4 emissions (Holden, 2005).In this way the depth of the water table largely determines the CH_4 flux rates. In general drainage in boreal peatlands has been found to change the CH_4 fluxes less in ombrotrophic than in minerotrophic peatlands. This is because the water table is less sensitive to drainage in ombrotrophic sites (Maljanen et al., 2010). On ombrotrophic sites drainage decreases CH_4 emissions on average by half, while many of the minerotrophic peatlands with high CH_4 emissions show CH_4 uptake after drainage. The calculated annual mean of CH_4 emissions from peatlands drained for forestry in Finland Sweden, Norway, Denmark and Iceland are 26 x 10^{-7} CO_2 eq kg ha^{-2} yr^{-1} (n = 9) for ombrotrophic and 12 x 10^{-7} CO_2 eq kg ha^{-1} yr^{-1} (n = 29), for minerotrophic peatlands (Maljanen et al., 2010). High CH_4 emissions may occur in the drainage ditches (Huttunen et al., 2003). In drained peatlands drainage ditches create a new anoxic zone which will be similar to the conditions of undrained mires. Under these circumstances CH_4 emissions may persist and even be enhanced compared to undrained mires (Schrier-Uijl, 2010). In ombrotrophic peatlands CH_4 ebullition from the ditches may be of the same order as in undrained peatlands. In minerotrophic peatlands the high emissions of CH_4 may occur due to the eutrophic water in the ditches. The emissions may be large enough to completely counteract the reducing impact of drainage (Minkkinen, et al., 2008). It is known from research on agricultural fen meadows in the Netherlands that ditches and bordering edges can be responsible for between 60% and 70% of farms total terrestrial CH_4 emissions (Schrier-Uijl, 2010). It has been estimated that during the summer months in a drained peatland plantation in Finland that the CH_4 emissions from the ditches were 4.5% of the total CH_4 emissions (Minkkinen, et al., 1997). CH_4 emissions from ditches may be assumed to be higher than those quoted as unknown quantities of dissolved CH_4 will be transport outside the peatland and emitted to the atmosphere (Minkkinen, et al., 2008).

Contradictory results have been found for the concentrations of DOC in peatlands catchments following drainage (Holden et al., 2004; Holden et al., 2005; Worrell et al. 2011); DOC release has been observed to increase immediately after drainage (Minkkinen et al., 2008). The severe erosion of peat due to peat decomposition has caused the loss of peat through the drainage network (Holden et al., 2004; Holden et al., 2005).In Britain severally eroded peatland fluvial losses of POC may exceed 1×10^{-5} CO_2 eq kg ha^{-1} yr^{-1} while the fluvial loss of gaseous CO_2 may amount to 2×10^{-7} to 13×10^{-7} CO_2 eq kg ha^{-2} yr^{-1} (Limpens et al., 2008). In the long term the increase in organic C leaching may be small (ca. 10%) or it may even decrease due to the reduction in groundwater flow caused by the ditches (Minkkinen et al., 2008).

N_2O emissions from peatlands are regulated by aerobic nitrification and anaerobic denitrification processes which again are controlled by the nutrient and oxygen status of the site (Huttenen et al., 2003; Pearson et al., 2012).The drainage of peatlands increases the availability of oxygen and increases the decomposition of organic matter which has an abundance of carbohydrates (Oleszczuk, et al., 2008). The oxygenation of the upper peat layers also enhances the mineralization of nutrients, particularly the carbon-bound nitrogen and sulphur and the organically bound phosphorus (Holden, 2004). These nutrients intensify the processes of nitrification and denitrification stimulating the production of N_2O (Maljanen et al., 2010; Oleszczuk, et al., 2008). Soil C: N ratio is known to be strongly related to seasonal N_2O emissions from peat soils (Pearson et al., 2012). Emissions of N_2O from drained peatlands are therefore greater for minerotrophic, nutrient-rich peatlands due to the greater availability of nutrients and because their pH encourages nitrate formation (Maljanen, et al., 2010; Huttenen et al., 2003; Pearson et al., 2012). Drainage of ombrotrophic peatlands may indeed have very little effect on N_2O fluxes (Huttenen et al., 2003). This difference in N_2O fluxes between ombrotrophic and minerotrophic peatlands drained for forestry has been observed in the Nordic countries were the mean annual emissions were 31×10^{-8} CO_2 eq kg ha^{-1} yr^{-1} (n = 11) and 14×10^{-6} CO_2 eq kg ha^{-2} yr^{-1} (n = 31) respectively (Maljanen et al., 2010).

The saturated microenvironment around the ditches can act as sinks for atmospheric N_2O

(Huttenen et al., 2003). The agricultural management of the drained peat will also influence N_2O emissions. Cultivated plants grown on drained peatlands are known to increase the amount of organic compounds in the soil by releasing carbohydrates, amines, amides and amino acids from their roots. Nitrate produced by the mineralisation of humus in the upper layers of drained soils moves into deeper layers of the peat and is reduced to N_2O. Drained peatlands are highly significant sources of N_2O with annual fluxes varying between 6.2 – 173.6 x 10^2 M CO_2 eq kg ha^{-1} yr^{-1} (Oleszczuk, et al., 2008). The application of fertilizers or manure to drained soils at high concentrations (480 kg N ha^{-1}) can increase the N_2O emissions significantly (Beetz et al., 2012; Oleszczuk, et al., 2008). N_2O losses from unfertilized grassland on peat soils are particularly potent with one study finding a 2.5–13.5 times more N_2O being emitted from the peat soils than from the sand and clay soils (Velthor & Onema, 1995). The N_2O emissions from drained peatlands are particularly relevant for the GHG balance for countries like Finland with has drained vast areas of peatland. For example 25% of the anthropogenic N_2O emissions in Finland are due to drainage of organic soils for agriculture despite the fact that they cover less than 10% of the total arable land area (Maljanen et al., 2010).

The effect of agricultural management peatlands on GHG emissions

The intensity of agricultural use on peatlands can vary greatly from the comparatively low impact land use of grazing on upland peatlands to the conversion of peat to nothing more than a substrate for grass or arable crops (Joosten & Clarke, 2002). Different management regimes will have different effect on the hydrology and biogeochemical processes of peat soil. The intensity of the management regime and the nature of the peatland will influence the GHG flux (Schrier-Uijl, 2010).

Annual mean GHG balances including net CH_4, N_2O and CO_2 emissions for drained peatlands in Fennoscandinavia over a 100 year time horizon are 2.26 x 10^{-4} CO_2 eq kg ha^{-1} yr^{-1} for grass swards, 2.28 x 10^{-4} CO_2 eq Kg^{-1} $ha^{-1}yr^{-1}$ for cereals and 3.14 x 10^{-4} CO_2 eq kg^{-1} ha^{-1} for those left fallow. Even after the cessetion of the cultivation practices, N_2O and CO_2 emissions were found to remain high (Maljanen et al., 2010). The mean net GHG emissions in abandoned agricultural peatlands is 1.58 x 10^{-4} CO_2 eq Kg^{-1} $ha^{-1}yr^{-1}$ (Maljanen et al., 2010). Ploughed arable land in Fennoscandinavia and fertilized cereal crops are greater sources of CO_2 and N_2O than grass fields but they are larger sinks for atmospheric CH_4 compared to grass fields due to their better aeration (Maljanen et al., 2010). In Western Finland for example the GHG efflux from two sites both under barley and grass were compared. The barley crop had higher net CO_2 emissions (830 x 10^{-6} CO_2 eq Kg^{-1} ha^{-1}) and N_2O emissions (026.4 x 10^{-6} CO_2 eq Kg^{-1} ha^{-1}) than those under grass (39.5 x 10^{-6} CO_2 eq Kg^{-1} ha^{-1} and 85 x 10^{-6} CO_2 eq Kg^{-1} ha^{-1}). The mean CH_4 uptake rate from the two farming practices was 21 x 10^{-8} CO_2 eq Kg^{-1} ha^{-2} and from bare soils 12 x 10^{-8} CO_2 eq Kg^{-1} ha^{-1} (Maljanen et al., 2004). The higher emissions associated with lands drained for crops are probably related to the process of tillage. Tillage causes the acceleration of peat mineralization compared to low intensity grassland management due to the intensive aeration of the peat. Arable farming may leave peat exposed and susceptible to water and wind erosion which may cause additional loss of POC (Parish et al., 2008). The N fertilization associated with the production of arable crops on peatlands will also result in greater N_2O emissions (Maljanen et al., 2010).

In Britain the GHG flux from drainage may be expected to be significant as more than half of the agricultural land is on drained land (Holden, 2004). A range of estimates of GHG flux from UK peatlands has been calculated by Natural England (2010) based on the IPCC peatland emission factors and the Durham Model. They reveal that undamaged peatlands in the UK are a carbon sink of -0.14 x 10^3 CO_2 eq kg yr^{-1}. As would be expected peatlands under agricultural management regimes were found to be significant sources of GHG. Cultivated and temporary grasslands were estimated to be net sources of GHG with a GWP of 3.11 x $10^3$$CO_2$ eq kg yr^{-1} which was the highest figure for any management regime in the study. It was almost double the total combined emissions caused by the extraction of peat, rotationally burnt peatlands, afforested peatlands and drained peatlands. The management regime with the second biggest GHG emissions is improved grassland on peat soils (1.06 x 10^3 CO_2 eq kg yr^{-1}). The positive GWP of grassland on drained peat are important as most commonly agricultural use of peatlands in Europe is as meadows and pasture for the grazing of cattle and sheep (Joosten & Clarke, 2002). Under grassland management drained bogs in the boreal and temperate zones lose ca. 2.5 x 10^3 of C kg ha^{-1} yr^{-1} while fens lose ca. 3.5 x $10^3$of C kg ha^{-1} yr^{-1} (Joosten and Clarke, 2002).

Grazing on upland peatlands is a common practice in Ireland and Britain. 85% of British upland peats are subject to some form of grazing (Worrell et al., 2011). Overgrazing on peatlands may cause the direct erosion of peat as a result of the trampling of the peat surface by the animal hooves or indirect erosion by leaving the peat soils bare and vulnerable to erosion by wind and water. Either process may lead the loss of carbon (Parish et al., 2008). However it is interesting to note that overgrazed peatlands have been found in some studies to be a net sink for -0.01 x 10^3 CO_2 eq kg yr^{-1} of GHG (Natural England, 2010; Worrall et al., 2011). The results from a study at Moor House in the North of England which investigated the annual fluxes of CO_2, CH_4 and DOC from a peatland being grazed by sheep also found that peatlands being grazed by sheep were GHG sinks. They found that grazing increased rates of respiration and photosynthesis relative to ungrazed plots making grazed plots a net sink for CO_2 (Ward et al., 2007; Worrell et al., 2011). These studies however do not take into account the CH_4 produced by sheep via fermentation or that while the litter produced in ungrazed plots is more recalcitrant than sheep effluent and therefore the sheep is converting C into a form that is more readily decomposed and lost from the environment (Worrell et al., 2007).

Greenhouse gas fluxes (CH_4, N_2O and CO_2) at a converted fen meadow were compared with those in a virgin fen in Finland over a two year period (Nykanen et al 1995). The field site was drained and converted to pasture. The natural fen was found to be a net sink for of atmospheric CO_2 (600 kg CO_2 ha^{-1} yr^{-1}), while it was a source of CH_4 (0.2- 0.4 x 10^{-2} CO_2 eq kg ha^{-1} yr^{-1}). However if the CH_4 produced by the fermentation of cows grazing on the field was added to the low CH_4 emission from the managed meadow then the farm had a GWP of CH_4 at 0.3 x 10^{x4} CO_2 eq kg ha^{-1} yr^{-1}. The methane associated with the cattle compensated for the natural soil methane (Nykanen et al 1995). The CH_4 emissions associated with the everyday running of the farm if included in this study would presumably increase the CH_4 emissions considerably. Agriculture on peat soil, excluding CO_2 emissions from machinery, cattle and energy production for fertilizers, caused a total GWP of 2.6 x 10^{x4} CO_2 eq kg ha^{-1} yr^{-1} with the increased N_2O emissions accounting for 10% of this figure. The results show that C loss from peat due to the intensity of the management regime is responsible for most of the atmospheric impact caused by agricultural utilisation of boreal peatlands (Nykanen et al 1995).

Table 1 is taken from Worrell et al (2011) and gives the greenhouse gas flux emissions from England's peat soils under a range of managements (CO_2 eq ha^{-1} yr^{-1}). It demonstrates the negative RF of peatlands compared to many common agricultural management regimes on peatland soils.

	Blanket Bog / Raised Bog (tonnes CO_2 eq ha^{-1} yr^{-1})	Fen Peatlands (deep - tonnes CO_2 eq ha^{-1} yr^{-1})
Cultivated & temporary grass	22.42	26.17
Improved grassland	8.68	20.58
Extracted	4.87	1.57
Rotationally burnt	2.56	
Afforested	2.49	2.49
Restored	2.78	4.2
Bare peat	6.00	
Overgrazed	0.1	
Pristine peatland	-4.11	4.2

Table 1. The greenhouse gas flux emissions from England's peat soils under a range of managements (CO_2 eq ha^{-1} yr^{-1}). (Worrell et al., 2011)

One thorough study which took place in the Netherlands compared the GHG emissions of CO_2, CH_4 and N_2O on two adjacent fen meadow areas which were subject to intensive and extensive agricultural management (Schrier-Uijl, 2010). While the study was not carried out in the British Isles or Fennoscandinavia the fen meadow habitats are comparable. Unlike many other studies which have dealt with the topic the full range of emissions originating from agricultural activities were accounted for. This included such as emissions due to the grass harvest and the CH_4 efflux from cow fermentation. The emissions from the drainage system were also measured as these can be a significant source of CH_4 that often goes unaccounted for. It was found that both of the meadows acted as net terrestrial GHG sources. The intensively managed area had emissions of 1.4 x 10^{-4} CO_2 eq kg ha^{-1} yr^{-1} and the extensively managed area had emissions of 1.0 CO_2 eq kg $ha^{-1}yr^{-1}$. The addition of the farm-based CO_2 and CH_4 emissions however increased the GHG source strength to 2.7 x 10^{-4} CO_2 kg eq $ha^{-1}yr^{-1}$ for the intensively managed farm and 2.1 x $10^{-4}CO_2$ eq kg $m^{-2}yr^{-1}$ for the extensively managed farm. The higher GHG emissions associated with the intensively managed grassland on peat compared to the extensively managed farm are mainly as a result of the higher N_2O and CH_4 emissions associated with the former. The drainage system was found to significantly contribute to the regional emissions of CO_2 and CH_4 due to the anoxic and eutrophic conditions (Schrier-Uijl, 2010).

The effect of peatland forestry on GHG emissions

As has been mentioned there has been extensive drainage of peatlands for forestation in Fennoscandinavia. Here nutrient rich mires have been drained and planted predominantly with species such as Norway spruce (Picea abies), Black spruce (Picea mariana), Tamarack (Larix laricina), and pubescent birch (Betula pubescens) (Joosten & Clarke, 2002; Moore, 2002). On ombrotrophic sites Scots pine (Pinus sylvestris) is the dominant species. In the British Isles were treeless mires are the norm drainage and afforestation with lodgepole pine (Pinus contorta) and Sitka spruce (Picea sitchensis) has been common particularly in many upland sites. In recent times as the conservation value of peatlands has become more appreciated there has been a cessation in the drainage of pristine peatland and instead forestry has focused on maintaining currently drained and forested sites (Moore, 2002).

The drainage of peatlands for forestry just as for agriculture lowers the water table and increases CO_2 emission and N_2O emissions while significantly reducing CH_4 emissions (Cannell et al., 1993). Where forestry differs from agricultural practices on drained peat is the gains in C storage associated with increased above and below ground litter input from trees and shrubs and the obvious gain in C storage in the biomass of the trees and vegetation (Cannell et al., 1993). Many studies have shown that unlike many agricultural practices the C balance on drained peatlands may actually increase due to the increased productivity of the forest stand (Minkkinen et al., 2008). In the early years of drainage the mineralisation of peat caused by the lowering of the water table causes plantations to be sources of CO_2 (Hargreaves et al. 2003). In Finnish and Swedish peatlands the seasonal CO_2

emissions increase after drainage. The relationship between the average water table level and CO_2 efflux is linear (Von Arnold et al., 2005).

The succession from peat forming vegetation to tree species changes the nature of the litter being deposited onto the peat. The tree litter is less recalcitrant than its predecessor and so there is an increase in the labile pool of soil organic matter and an increase in heterotrophic CO_2 efflux. Autotrophic respiration from the roots increases contributing usually 10–50% of total soil respiration in forestry-drained peatlands (Minkkinen et al., 2008). Four to eight years after the planting of the trees the peatland turns into a CO_2 sink. The C gains associated with the increased productivity of the recolonized ground vegetation combined with the increase in the biomass and litter stores cancels out the losses from the peat carbon store (Hargreaves et al. 2003; Joosten & Clarke, 2002). The litter deposited in the soil decomposes and emits CO_2 (Minkkinen et al. 2008). The C stores in ground vegetation biomass may either increase or decrease but are generally eclipsed by variations in the tree stand (Minkkinen et al.,2008). It has been shown that peatland forests can be either a source or sink for CH_4 depending on the drainage status, but upland forests most often are sinks for atmospheric CH_4 (Maljanen et al., 2010). As in the process of agricultural drainage CH_4 emissions may decrease by up to 50% after drainage (Minkkinen, 1999; Minkkinen et al., 2002).

Drainage for forestry has been shown to increase N_2O emissions significantly at nutrient-rich peatland sites where the pH is high enough for nitrate formation through nitrification (Minkkinen., 2008; Minkkinen et al., 2002; Von Arnold et al., 2005).

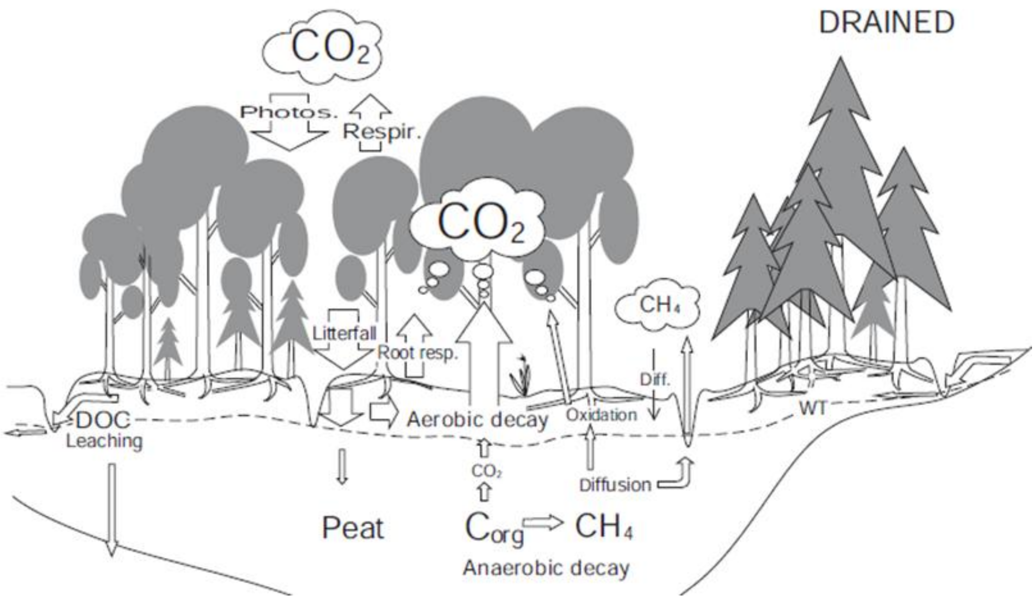


Fig. 7 The carbon cycle in peatlands drained for forestry (Minkkinen, et al., 2008)

Clear felling alters the stand density as well as the site hydrology and thus also the oxygen status of the soil which effects the emissions of CO_2, CH_4 and N_2O (Maljanen et al., 2010;

Worrell et al., 2011). Increased N_2O emissions and decreased CH_4 uptake rates have been observed following clear cutting (Minkkinen et al., 2008; Huttunen et al., 2003). There is an obvious loss of C following clear felling. The decrease in primary production and the drastic changes in light and moisture conditions for the ground vegetation greatly reduces production after felling. During this period the CO_2 lost from the peat is not compensated by primary production (Minkkinen et al., 2008).

The C stores increase or decrease due to afforestation, depending on the nutrient level of the peatland and climatic conditions such as the temperature (Parish et al., 2008). Positive C balances may be achieved on nutrient poor sites. This is thought to be the result of a number of contributing factors. The trees produce more roots on poor sites in order to increase their nutrient uptake. The ombrotrophic sites are typically dominated by Scots Pine (Pinus silvestris) which allows a lot of light through to the ground layer. The production of mosses in the ground layer is therefore superior. The paucity of available nutrients inhibits high decomposition rates. Poor drainage is typical on nutrient poor sites which maintain a shallow aerobic peat layer. The leaf litter of the Scots Pine is more recalcitrant than the leaf litter of broad leaf species which may be found on minerotrophic sites. The high production rates combined with the poor decomposition rates compared to minerotrophic sites results in the observed disparity between site types (Minkkinen et al., 2008).

In a study in Finland where the lowering of the water table was minimal the rate of C sequestration into peat has increased from a value of 2.2 Tg a^{-1} on a pristine state in 1900 to 3.6 Tg a^{-1} (Minkkinen et al., 2002). The C store in the tree biomass increased from 60 - 170 Tg during the 20th century while CH_4 emissions have decreased from an estimated 1.0 - 0.5 Tg CH_4 C a^{-1}, while those of N_2O have increased from 0.0003 to 0.005 Tg N_2O N a^{-1}. The negative RF caused by afforestation is caused by increases in CO_2 sequestration in peat (-0.5 mW m^{-2}), tree stands and wood products (-0.8 mW m^{-2}), decreases in CH_4 emissions from peat to the atmosphere (-1.6 mW m^{-2}), and only a small increase in N_2O emissions (+0.1 mW m^{-2}) (Minkkinen et al., 2002). While the accumulation in the peat layer is significant this study based its carbon balance on the assumption that ditches have a zero C balance. However it is known that ditches are a significant source of CH_4 and may have emissions levels that are akin to those of undrained peatlands (Minkinnen, 1999). For example in a study in Canada it was that the emissions of CH_4 from the ditches were significant at <5 to >400 mg CH_4 $m^{-2}d^{-1}$but the typical ditch spacing used in forested bogs (40 m) resulted in a net increase in CH_4 emissions (Roulet & Moore, 1995). The increse in the peat and tree stand C sequestration and the decrease in CH_4 emissions following drainage and afforestation greatly overshadowed the large increase in N_2O emissions causing the plantation to have a negative radiative forcing compared to the undrained peatland over at least the next 100 years (Minkkinen et al., 2002).

In Finland it has it has been suggested that forestry on drained peatlands has significantly decreased the GHG emissions and radiative forcing of peatlands (Minkinnen, 1999). On such study in Southern Sweden compared an undrained peatland with three drained and forested peatlands (Von Arnold et al., 2005). The CO_2 emissions from the drained sites were found to be significantly higher (0.9–1.9 x 10^{-4} kg ha^{-1} yr^{-1}) than at the undrained mire site (0.8 - 1.2 x 10^{-4} kg ha^{-1} y^{-1}). CH_4 emissions were significantly lower for the drained sites (0.0 - 1.6 x 10^{-7} kg ha^{-1} yr^{-1}, compared to 10.6 - 12.2 x 10^{-7} kg ha^{-1} yr^{-1}), while N_2O emissions were significantly lower from the undrained site than from the drained sites (20 - 30 x 10^{-9} kg ha^{-1} yr^{-1}, compared to 30 - 90 x 10^{-9} kg ha^{-1} yr^{-1}). The drained sites were still net sinks, taking up 0.2 – 0.8 x $10^{-4}$$CO_2$ eq kg ha^{-1} yr^{-1}. The undrained mire was a GHG source of 0.5 x 10^{-4} CO_2 eq kg ha^{-1} yr^{-1}. A Significant observation however that is this study was based on the net emissions of GHG over 30 to 50 years. As such the GWP of the peatlands were heavily influenced by the initial C gain associated with the growing trees and with the strong RF of CH_4 emissions (Von Arnold et al., 2005).

In a peatland in Scotland the Carbon balance of an undisturbed peatland was also compared to a newly drained and afforested peatland. The pristine peatland accumulated ~0.25 t C $ha^{-1}a^{-1}$. The drained peatland emitted 2 - 4 t C $ha^{-1}a^{-1}$ but became a sink of ~3 t C $ha^{-1}a^{-1}$ 4 – 8 years after afforestation (Hargreaves et al., 2003). Despite this accumulation in above ground biomass however the peat continued to decomposing at a rate of ~1 t C $ha^{-1}a^{-1}$. The slow rate of peat decomposition means that afforestation will cause the plantation to be a net carbon sink in tree, litter, forest soil and products for 90–190 years (Hargreaves et al., 2003).

Byrne et al. (2004) calculated the GWP of different management regimes on European peatlands in both temperate and boreal zones. The study revealed that that ombrotrophic bogs had a GWP of -10.5 CO_2 eq kg ha^{-1} yr^{-1} over the next 100 years (CO_2 = 0.19 CO_2 eq kg $ha^{-1}yr^{-1}$ (n=8), CH_4 = 234.15 CO_2 eq kg $ha^{-1}yr^{-1}$(n=14), N_2O = 12.4 CO_2 eq kg $ha^{-1}yr^{-1}$(n=7)). It was found that drained and afforested minerotrophic fens had a GWP of 4.2 CO_2 kg eq ha^{-1} yr^{-1} (CO_2= -0.2 CO_2 eq kg $ha^{-1}yr^{-1}$ (n=4), CH_4 = -1.05 CO_2 eq kg $ha^{-1}yr^{-1}$ (n=13), N_2O = 12.4 CO_2 eq kg $ha^{-1}yr^{-1}$ (n=20)). A 100 year time period is used to calculate the GWP for afforested sites as this is the normal time scale which the IPCC uses to calculate GWP. Given the high productivity of forests compared to peatlands the CO_2-fixation rates of the plantations are significantly higher than those of undisturbed peatland vegetation. The high C sequestration rates will cause the forests to become slight net sinks for CO_2 if the fixation rate is greater than the carbon C lost through the decomposition of the peat (Byrne et al., 2004; Parish et al., 2008). However this is when the timescale under consideration becomes very important as the biomass and litter stores of forests in temperate and boreal regions usually reach C equilibrium after 100 years. As the mineralisation of the peat continues the cumulative carbon losses will dominate the GHG balance and exceed the C gained due to forestry if a long enough time scale is considered (Cannell et al. 1993; Hargreaves et al., 2003; Joosten & Clarke, 2002; Parish et al., 2008). The wood is normally harvested and the biomass is removed long before even this equilibrium is reached as forestry rotations are usually 60-

100 years (Minkkinen et al., 2008; Parish et al., 2008). Undisturbed peatlands may in comparison continue to accumulate carbon over millennia (Byrne et al., 2004).

The carbon storage capacity of wood products has been used to increase the C sequestration capacity of forested peatlands. It is important to note that wood products from peatlands or drained peatlands are typically short lived products like paper, furniture or construction materials which may only last between 57-92 years after which point the C they contain will be decomposed and returned to the atmosphere (Byrne et al., 2004; Hargreaves et al., 2003). Another point that is important is that in order for a full GHG budget to be made for forestry on peatlands the emissions associated with the management, felling, transport and finally the treatment and manufacturing of the wood should be considered. These emissions may be vast. The total GHG emissions associated with the forestry operation in the Pacific Northwest for example have been calculated to be 1.6 Mg CO_2 eq ha^{-1} per 100 m^3 (Sonne, 2006).

The analysis of the GHG dynamics of plantations on peatlands over short time spans is myopic. Over time spans of a century the increases in peat and tree stand C sequestration and indeed the decrease in CH_4 emissions from peatland after drainage will cause afforestation to have a negative radiative forcing on global warming (Minkkinen, 1999). If short time scales are under consideration the greater radiative forcing of CH_4 and N_2O compared to CO_2 will overshadow the GHG balance of the peatlands and the long term C sequestration ecosystem service will go underappreciated as has been the case with many studies on the GHG dynamics of pristine peatlands (Byrne et al., 2004; Hargreaves et al., 2003; Minkkinen, 1999; Von Arnold et al., 2005). The transformation of peatland to plantation may be irreversible. Timber extraction not only damages the peatland vegetation but also the peat soils (Moore, 2002, Minikkinen et al., 2008). If considerable damage is done to the peatland it may become impossible to recreate the C and GHG sink function of certain peatlands (Worrell et al., 2011). The C sequestration capacity and the ability of certain peatlands to lower RF over millennia may therefore be lost forever.

The effect of peatland restoration on GHG emissions and C sequestration

The value of peatlands and the ecosystem services are known to include biodiversity, agricultural value, forestry, water quality enhancement, flood water retention, recreation, archaeological resource and landscape enhancement (Barkham, 1993, Renou-Wilson, et al., 2011). Peatland restoration is a relatively new development with the first upland peat drain being dammed in Britain in the late 1980's (Ramchunder et al., 2009). In Sweden 30 % of the peat cut area covering 5000 ha was targeted for restoration by 2010. In Ireland the predicted cessation of industrial peat cutting on many peatlands will leave ca. 30,000 ha where restoration practices may be applied over the next 20-30 years (Höper, 2008). In the past nature conservation has been the greatest driver of peatland restoration. In more recent times the carbon sequestration service of peatlands has gained more attention and has been the excuse for 62% of restoration projects (Lunt et al., 2010). Certain types of management of peatland are unsustainable (Parish et al., 2008). It may be the case that the GHG emissions resulting from the management and degradation of peatlands can only be significantly reduced if the agriculture or forestry regime that is causing the emissions is reduced in intensity or completely abandoned (Röder & Osterburg, 2012; Schrier-Uijl, 2010). The aim of peatland restoration is to recreate a sustainable, active peat forming ecosystem that is as close to a pristine condition as possible (Komulainen, et al., 1999; Moore, 2002). If drastic alterations have been made to the peat morphology, composition and hydrology then recreating an active peatland may be impossible (Höper, 2008). The degree to which the peatland has been altered from its original state will determine the possibility that it can be quickly restored to a near pristine condition. Peatlands that have been afforested or converted for agriculture and have only seen minor drainage and fertilization have a greater chance of being successfully restored to a pristine condition compared to areas that have been excavated (Höper, 2008). Restoring the hydrological system is essential. The first step is restoring the water table nearer to the surface by damming the drainage system. Revegetation is essential in order to recreate an active peat forming ecosystem (Lunt et al., 2010; Ramchunder et al., 2009). The revegetation of the peat surface dramatically reduces surface erosion and its associated C loss, in particular the loss of POC (Lunt et al., 2010; Moore, 2002). Decreasing erosion is an important step in preserving the existing C stocks. This in itself should be a high priority in relation to climate change (Lunt et al., 2010). The rewetting of a managed peatland causes a significant reduction in the CO_2 emissions from decomposition of the peat due to the recreation of anaerobic conditions in the peat (Moore, 2002). Rapid vegetation recovery after rewetting is more likely in peatlands drained for forestry as mire plants may recolonize from nearby or may have persisted within some forest where drainage was less severe (Vasander et al., 2003)

Restoration generally reduces the N_2O emissions from drained ombrotrophic raised and blanket bogs (Joosten & Clarke, 2002). As pristine peatlands are N_2O neutral any return to this state will reduce N_2O emission in the long run (NaturalEngland, 2010). The rewetting of a peatland will recreate the anoxic conditions in the peat that are associated with CH_4 production (Joosten & Clarke, 2002; NaturalEngland, 2010). The scale of the increase of CH_4 is vital as the increased CH_4 emissions from the rewetted peat may cancel out the prevented losses of CO_2. High CH_4 emission may occur when peatlands with a lot of standing vegetation are reflooded (Sirin & Laine, 2008). The removal of vegetation and the careful control of the water table should help to prevent this. Recolonization by *Sphagnum* and other mosses and sedge species will recreate the acrotelm. Methane-oxidizing bacteria live in and on

Sphagnum and serve as a methane filter and limit methane emissions (Dedysh, 2009; NaturalEngland, 2010; Kip, et al., 2011).

In the case of restoring forested peatlands the release of the emissions associated with clear-felling will result in the loss of the large stocks of C as well as increased N_2O emissions decreased uptake of CH_4. It is important to note that these emissions would ultimately be in the rereleased when the timber is being harvested and the above ground C stocks will inevitably be rereleased to the atmosphere anyway once the wood products are decomposed (NaturalEngland, 2010). Initial carbon sequestration by the restored peat surface will be well below that of the felled trees. In the long run however restoration will recreate the long-term carbon sequestration capacity of pristine peatlands. Over time spans of greater than 150 years peatland restoration more likely than not will deliver more greenhouse gas benefits than afforestation (NaturalEngland, 2010).

It is possible that given enough time all peatlands with the exception of those that have been drastically modified will return to an active peat forming state after restoration. Where exceptions do occur, short term negative impacts on water quality and an increase in methane emissions may arise (Lunt et al., 2010). There is however cause for optimism as a number of case studies have shown restoration of the C sequestration capacity of degraded peatlands and a switch to a negative GWP in the long term.

In southern Finland Komulainen, et al (1999) found that the restoration of a minerotrophic fen site which had been drained 43 years previously returned it to a carbon sink again comparable to that of a pristine fen. Restoration involved clear cutting the trees and blocking the drainage ditches. Blocking the drains was effective in raising the water table ca. 25 cm. The vegetation responded quickly to the change and the cover of cottongrass (*Eriophorum vaginatum*) increased significantly. Restoration was found to decrease the CO_2 efflux from the soil surface. Even after one year the seasonal CO_2-C balance of the fen site began to compensate for the lost carbon fixation capacity of the former tree stand. The carbon balances at the restored fen had already approached those of pristine mires during the first years of restoration. The seasonal CO_2-C balances at the rewetted fen site varied from 162 - 283 g m^{-2}. These are encouraging results from the point of view of restoring the C sequestration capacity of afforested fen sites in Fennoscandinavia. However this study vitally failed to account for the rise in CH_4 due to rewetting or to calculate the GWP of the restored fen (Komulainen, et al., 1999).

In his review paper on the relationship between climate change and peatlands in the UK Lindsay (2010) uses the model created by Colls (2006) in his unpublished PhD to study the carbon fluxes associated with the restoration of previously afforested blanket bog. Fifteen years prior to restoration of the blanket bog it had been plantation of lodgepole pine and Sitka spruce. The reworked model of Colls (2006) uses 5 previously observed rates of carbon accumulation following rewetting (1×10^4 CO_2 kg $ha^{-1}yr^{-1}$ - 5×10^2 CO_2 eq kg $ha^{-1}yr^{-1}$). Using the same model as Colls (2006) but applying a more realistic decomposition model that partitions decomposition in the surface, acrotelm and catotelm it was found that under all scenarios except the slowest peat accumulation rate the process of restoration results in net gains in CO_2 carbon from the very start at year one. This reworked model indicates that using this slowest rate of carbon accumulation (0.1×10^2 CO_2 eq $ha^{-1}yr^{-1}$), net carbon gain over decomposition losses is achieved after ca. 42 years. Balanced against these gains are the losses of carbon as CO_2 from decomposition and the release of CH_4 from the bog surface following rewetting. The CH_4 GWP used by Colls (2006) in this model was 3×10^2 CO_2 eq $ha^{-1}yr^{-1}$ over a 100 year time span. The highest carbon accumulation rate of 5×10^2 CO_2 eq $ha^{-1}yr^{-1}$ was found to be greater than the GWP of the restored peatland over a 100 year time period. While the 4×10^2 CO_2 eq kg $ha^{-1}yr^{-1}$ accumulation rate was marginally less that GWP over the same time span. Over a 500 year time period all but the two lowest accumulation rates of 2×10^2 CO_2 eq kg $ha^{-1}yr^{-1}$ and 1×10^2 CO_2 eq $ha^{-1}yr^{-1}$ were below the GWP caused by the CH_4 emissions. That implies that the three highest carbon accumulation rates for the restored peatland have a cooling effect on the climate despite the CH_4 emissions. The 2×10^2 CO_2 eq $ha^{-1}yr^{-1}$ carbon accumulation rate was marginally below the GWP over the 500 year time scale over a longer time scale; in the region of millennia it would undoubtedly greatly exceed it. Recolonization by *Sphagnum* will also decrease the levels of CH_4 emissions used in this model even more thereby increasing the cooling capacity of the restored peatland with time.

A study in the Netherlands measured the total field GHG emissions originating from a drained fen meadow under intensive agriculture (Schrier-Uijl, 2010). These GHG emissions were compared to those measured on a former agricultural site which had been flooded and restored to a nature reserve 15 years previously. The CO_2 and N_2O emissions from the managed site were much greater than those found in the restored site. The intensively managed site was a source for 1.4×10^{-4} CO_2 eq kg $ha^{-1}yr^{-1}$ while the restored site was found to be a GHG sink of 7×10^{-3} CO_2 eq kg $ha^{-1}yr^{-1}$. When the farm based CO_2 and CH_4 emissions associated with the day to day running of the farm were included in the GHG budget then the source strength for the managed increases to 2.7×10^{-4} CO_2 eq kg ha^{-1} yr^{-1}. That study demonstrates that the rewetting of agricultural peatland can not only decrease the emissions associated with degraded peat but that within a short space of time it is possible to switch them from net GHG sources back to sinks (Schrier-Uijl, 2010).

Although there are some notable success stories, restoration science is in its infancy and the results of restoration on the GHG balance of degraded peatlands vary greatly. Each case is different but the emissions of CO_2, CH_4 and N_2O and the C accumulation vary depending on the biome, peat type and state of degradation (Höper, 2008). Figure 8 is an illustration taken from Bain et al (2011) show the GWP of peatlands under global warming potential of peat bogs under natural, drained and rewetted states. It demonstrates the potential that rewetting a peatland by blocking drainage ditches has to reduce emissions. Even the initial increase in CH_4 emissions that occur due to rewetting are not enough to outweigh the benefits of rewetting (Baird et al. 2009).

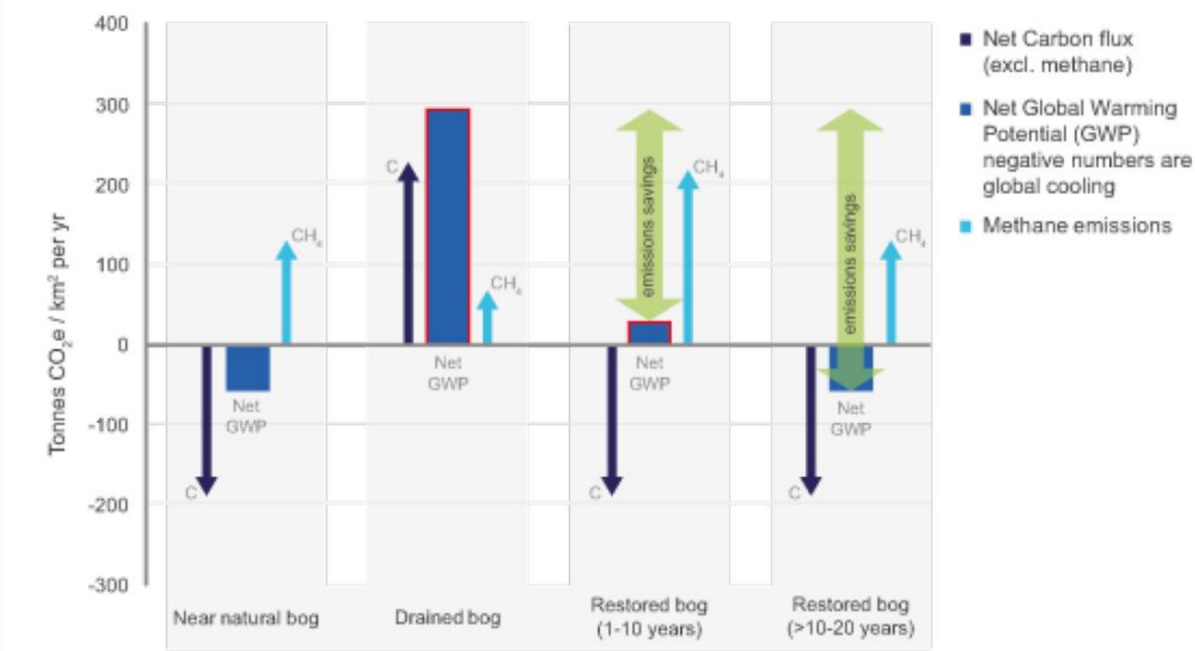


Fig. 8 The global warming potential of peat bogs under natural, drained and rewetted states (Bain et al., 2011).

The common consensus seems to be that if the extent of degradation is not too severe then it may be possible to restore the carbon sequestration of managed peatlands (Drösler, et al., 2008; Parish et al., 2008). There is therefore hope that vast areas that have been degraded for the purposes of agriculture and forestry in North Western Europe may be returned to a near pristine and slow the rate of anthropogenic warming for millennia. The future promise of peatland restoration has been recently recognised by the IUCN (Röder & Osterburg, 2012; Roulet, 2000). Starting in 2013 as part of the new Kyoto Protocol countries will be able to use the restoration of peatland under the Land Use and Land Use Change (LULUCF) sector to help them meet their national greenhouse gas targets. All peatland restoration projects undertaken since 1990 will be eligible (Bain et al., 2012).

Discussion

In general these studies show that that the drainage and degradation of pristine peatlands causes the release of considerable amounts of GHG. While the effects of the various management regimes may differ it would seem that the more intensive the management regime is the greater the GHG emissions will be. The reduction in CH_4 emissions and the initial increase in C sequestration associated with the primary productivity of trees and ground vegetation may cause a decrease in the GWP of peatlands over short time periods such as a century. Generally speaking the drainage and cultivation of peatlands significantly increases their CO_2 and N_2O emissions to the atmosphere, while it may reduce CH_4 emissions (Oleszczuk, et al., 2008). Following drainage there is a 53% probability that a peatland will become a net source of C, due to an increase in the flux of CO_2. However due to the decrease in CH_4 emissions following drainage there is a 69% probability that drainage will result in an overall improvement in the GHG budget in a peatland over a 100 year time span (Worrell et al. 2011). However as with the analysis of the GHG balance of pristine peatlands this cooling effect is only evident over short time spans of century's and over longer time spans of millennia. It is likely that given the longer turn over time of CO_2 in the atmosphere relative to CH_4 and N_2O, any loss in the carbon sink function of peatlands will lead to a net warming effect over longer time scales. Globally the 500,000 km^2 of drained peatlands release as much as 2 Gt yr^{-1} of CO_2 to the atmosphere (Bain et al., 2011). The UNFCCC has an even higher estimation of 3 Gt yr^{-1} of CO_2 from degraded peatlands. This is equivalent to over 10% of the total global anthropogenic CO_2 emissions from 1990 or 20% of the total net 2003 GHG emissions of the Annex 1 Parties to the UNFCCC (Parish et al., 2008). This puts into context just how significant degraded peatlands are in climate change. The GHG fluxes of drained peat soils depend on soil properties, ground water level and regional climate but beyond this the agricultural and forestry practices, the crops planted and the management regimes that are applied will also have a major impact (Maljanen et al. 2010; Schrier-Uijl, 2010). If all of the emissions associated with farming and the timber industry are accounted for it is likely these management regimes will be greater sources of GHG than has been predicted in many publications. The protection of peatlands is one of the most cost-effective management strategies for minimising CO_2 emissions (Pearish et al., 2008). Although in its infancy peatland restoration seems to offer a means of protecting further loss of existing peat C stores. This in its self is significant as peatland degradation is one of the most important global sources of CO_2 emissions from the Land Use and Land Use Change (LULUCF) sector (Pearish et al., 2008). Pristine peatlands where they exist should be strictly protected for their conservation value as well as their role in the carbon sequestration. The restoration of the peat forming capacity of degraded bogs should be made a key priority in the climate mitigation policies of Ireland, Britain and Fennoscandinavia. The incorporation of peatland restoration projects into the new Kyoto Protocol is one mechanism that can be used to help the countries of NW Europe meet their national greenhouse gas targets (Bain et al., 2012). The science of peatland rehabilitation needs to be refined and future research is needed to develop specific management plans.

Section 4

The effect of projected climate change on the peatlands of NW Europe

As was mentioned in Section 2 the global climatic changes have been observed worldwide over the last few decades are very likely the result of global warming which is likely to have occurred as a direct result of the anthropogenic release of greenhouse gases through the burning of fossil fuels and land use change (Malhi & Wright, 2004; IPCC, 2007). As we know peatlands are capable of producing and storing vast quantities of all three of the most important long lived greenhouse gases CO_2, CH_4, N_2O (Sirin & Laine, 2007). Peatlands have played an important role in global CO_2 and CH_4 atmospheric concentrations throughout the entire Quaternary period. Through their expansion and contraction during interglacial periods they have acted as mediator or as a positive feedback for the atmospheric change (Moore, 2002; Sirin & Laine, 2007). Peatlands through their C sequestration capacity have the potential to provide a negative radiative force and slow down the rate of global climate (Holden, 2005).

Climate is the most important determining factor on distribution of peatlands on both national and continental scales (Parviainen & Luoto, 2007). It determines their location, typology and biodiversity throughout the world (Joosten & Clarke, 2002; Parish et al., 2008). Peatlands will be one of the most vulnerable ecosystems to environmental change (IPCC, 2007). After anthropogenic land use change, climatic change together with nitrogen (N) deposition are the most significant threats to peatland ecosystem functioning (Limpens et al., 2008). Future global climate change will have substantial impacts on the distribution, functioning and biodiversity of peatlands throughout the world (Parish et al., 2008). Climate governs many of the features that determine the function that peatlands play in the carbon and greenhouse gas cycles. These strong links between climate and peatlands imply that any future change in climate will have a knock on effect on the distribution and functioning of peatland ecosystems (Parish et al., 2008). Differing regional climate change predictions mean that the influence of climate change on peatlands will vary from place to place (Schouten et al., 1992).

An area of uncertainty in future climate projections is the global climate-carbon cycle feedback. Many of the carbon sinks are predicted to be negatively influence by future climate change predictions (Lund, 2009). Coupled carbon climate models predict a positive feedback between terrestrial carbon cycles and global warming (Cox, et al., 2007, Friedlingstein, et al., 2006). Over the last five decades there has already been a decline in the efficiency of natural CO_2 sinks worldwide (Limpens et al., 2008). The three carbon pools that are thought to be most vulnerable to climate change are the carbon locked in permafrost, in wetlands/peatlands, and forests (Gruber et al., 2004; Lund, 2009). Climate change is predicted to significantly impact northern peatlands in particular (Alcamo, et al., 2007, Lund, 2009). It has been suggested that 100 Pg of the C stored in peatlands may be vulnerable to climate change and may be released into the atmosphere during the 21st century (Lund, 2009). Given their vulnerability, rarity and important role in the global C cycle and greenhouse gas flux research into the effects of predicted climate change of the peatlands of NW Europe is of foremost importance.

Future Climate Change predictions for NW Europe

The current scientific consensus is that many trends that have been observed in the global climate over the last few decades have been attributed to anthropogenic climate change. According to the IPCC Fourth Assessment Report (2007) on global climate change, there is a > 90% chance that the rise in global temperatures observed over the past 50 years is due to anthropogenic activities associated with increasing atmospheric concentrations of greenhouse gases. The average global surface temperature has increased by 0.76°C over the period 1850–1899 to 2001–2005 while global land precipitation has increased by 2% per over the past century (IPCC, 2007). Mean annual temperatures in Ireland for example has have risen by 0.74°C over the past 100 years (Sweeney et al., 2008).

A number of modelling studies have tried to predict the climate of N W Europe under various emissions scenarios. Uncertainties remain in the future projections particularly on a regional and local scale. A major factor in this is that N W Europe and the North Atlantic are particularly challenging in terms of climate system understanding. Despite the lack of detail in the future projections NW Europe is clearly seen as a as a regional climate change 'Hot Spot' (Coll, et al., 2003).

Temperature

The IPCC Fourth Assessment Report (2007) predicts Europe will warm across all seasons in both the SRES A2 and B2 emissions scenarios (A2: 2.5 - 5.5°C, B2: 1 - 4°C). Western Europe is one of the regions that will see the greatest warming in summer (JJA) temperatures. Temperatures in the north Atlantic are predicted to rise with the temperature change being greatest at higher latitudes with two regional climate models predicting a larger warming in winter than in summer in northern Europe. In Ireland by the 2050's, regional climate models predict warming for all seasons. The HadCM3 and CCC ma models propose a difference of almost 2°C in the winter season (Sweeney et al., 2008). Annual warming in the UK is predicted to lie between 1 and 5°C by the 2080's with greater warming in summer and autumn than in winter and spring (Hulme et al., 2002).

In northern Europe it is highly likely that warming will be experienced in the high latitudes in winter with the lowest winter temperatures predicted to increase more than average winter temperature (IPCC, 2007). Benstad (2005) using the multi-model IPCC AR4 created climate simulations which predict that the strongest warming is northern Europe will be in the high mountains in southern Norway, and the interior of Finland, Sweden and Norway. The least warming is projected for the British Isles due to the greater oceanity of the climate. The greater winter warming would have a knock on effect meaning a shorter snow season a change in the seasonality of river flow due to an earlier snow melt (Benstad, 2005). Using 15 global model simulations of future climate the annual mean temperature in Finland is predicted to rise by 2-5 °C by the 2050's and 2-7 °C by the 2080's (Julha et al., 2004). Models predict that there is a > 95% that the frequency of warm spells and heat waves will increase as well as an increase in maximum temperatures (IPCC, 2007). Extreme weather events such as droughts may also increase in the frequency (IPCC, 2007). Regional decreases in soil moisture are predicted (Hulme et al., 2002).

Precipitation

There is less certainty over predicted changes in precipitation in the climate models than for temperature (Parish et al., 2008). Averaged water vapour, evaporation and precipitation are projected to increase in Europe (Benstad, 2005; IPCC, 2007). There is a >90% probability that annual precipitation and extremes in precipitation will increase in northern Europe. In multitudes where most of the peatlands in NW Europe are found it is predicted that there will be an increase in winter and decreases in summer mean precipitation (Hulme et al., 2002; IPCC, 2007). Winter precipitation is predicted to rise and most models predict an increase in summer precipitation north of 55°N. This will affect Scotland and Fennoscandinavia. Precipitation in Finland is projected to increase 5–40% by the 2080's (Jylha et al., 2004). Extremes in winter precipitation will increase in magnitude and frequency In Fennoscandinavia (Christensen et al., 2001; IPCC, 2007). There is no major change predicted in dry-spell length in northern Europe (IPCC, 2007). In Ireland it is projected that by the 2050's there will be an increase in precipitation (12%) in winter and a decrease (12%) in summer (Sweeney et al., 2008).

Snow

There is a >90% chance that the duration of the snow season in Fennoscandinavia will shorten extent as is the depth of snow due to increased temperatures. In the UK snowfall will decrease as will the likelihood of snow fall (Hulme et al., 2002). There is a projected increase in winter precipitation in Fennoscandinavia (Julha, et al., 2004). This increase may counteract these decreases in snow cover and season length particularly in northern high latitudes (IPCC, 2007; Parish et al., 2008). An increase in snow depths will have a significant effect on the thermal regime at the peatland surface (IPCC, 2007; Parish et al., 2008). In southern Fennoscandinavia there may by a 50 to 100% decrease in snow depth in most of Europe by the late 21st century. Less change is projected for the North of Fennoscandinavia over the next century. By 2100 up to 90% of the upper layer of the world's permafrost is expected to melt (IPCC, 2007). This will have a huge impact on northern peatlands.

Sea Level

Rising temperatures will cause the thermal expansion of sea water as well as causing the melting of terrestrial glaciers. This is projected to raise global sea levels by between 0.28 and 0.4 m (IPCC, 2007).

Climate Change and the mid-latitude peatlands

Within NW Europe blanket bogs are mainly restricted to the hyperoceanic regions on the western Ireland, Britain and Norway. They thrive in wet and humid areas without any sustained droughts (Clarke et al., 2010a; Gallego-Sala et al., 2010; Gallego-Sala & Prentice, 2012). Precipitation has been shown to be more important climatic variable in explaining the blanket bog distribution in Fennoscandinavia (Parviainen & Luoto, 2007).The existence of these peatlands is reliable on high levels of precipitation and poor drainage to maintain a high water table (Clarke et al., 2010a). The *Sphagnum* species which are an essential component of ombrotrophic bogs rapidly suffer damage at temperatures greater than 15 °C (Gallego-Sala & Prentice, 2012). This dependence on precipitation makes ombrotrophic peatlands very vulnerable to projected climate change (Clarke et al., 2010a).

Increasing temperatures are predicted to result in water stress in the mid latitude peatlands of the British Isles and Fennoscandinavia. Such pressures on the water supply to peatlands will be further exacerbated by an increase in the frequency of extreme weather and regional decreases in soil moisture (Hulme et al., 2002; IPCC, 2007). The projected increase in heat waves, droughts and maximum temperatures as well as the projected decrease in summer precipitation in NW Europe will very likely have a detrimental influence on the functioning and distribution of ombrotrophic peatlands in NW Europe (Hulme et al., 2002; IPCC, 2007). These predicted climate changes will very likely affect the net water balance between precipitation and evapotranspiration thus altering the species composition and the balance between decomposition and primary production (Clarke et al., 2010a). The NEE of ombrotrophic bogs in summer time has been shown to be significantly influenced by precipitation (Lund et al., 2010). Bog ecosystems as they are elevated above the underlying WT and are reliant on precipitation will be especially vulnerable to drought, elevated transpiration and water stress associated with higher temperatures and more sporadic precipitation patterns (Lund et al. 2010). Drought is known to decreases photosynthesis and GPP while increasing Reco rates and therefore will reduce the C sink strength of temperate and boreal peatlands (Tarnocai, 2006). The lowering of the water table as a result of climate change will have the same effect on peatlands as the drainage (Section 2) changing them from a C sink to a source (Clarke et al., 210a).

Aurela et al (2007) using eddy covariance (EC) measurements of net ecosystem CO_2 exchange (NEE) in a boreal sedge fen in southern Finland found a clear depression in the CO_2 sequestration capacity of the peatland due to a drought in the summer of 2005. Measurements were taken during a one and a half year period covering two summers in 2004–2005. A drought period in 2005 was observed to markedly suppress the net uptake of CO_2, with a clear response to various hydrometeorological quantities like temperature, water vapour pressure deficit and WT level. The annual net sink of the fen was found to be 31 g CO_2 m^{-2} weaker in 2005 compared to 2004. It was concluded that the suppression in the fens C sink was caused by drought related to an increase in respiration and a decrease in photosynthetic rates caused by the high vapour pressure deficit (Aurela et al., 2007).

Recently a number of bioclimate envelope models (BEM) or ecological niche models have been developed to project the future distribution of peatlands in Ireland, Britain and Fennoscandinavia under different emission and climate change scenarios (Clarke et al., 2010a; Clarke et al., 2010b; Gallego-Sala, et al., 2010; Gallego-Sala & Prentice, 2012; House, et al., 2010; Jones, et al., 2006; Paviainen & Luoto, 2007) . Models are first developed that can adequately replicate the current full spectrum of climates encountered within the distribution the peatlands. Once this is achieved the future distribution of peatlands can be predicted by manipulating the climatic variables to replicate future climate scenarios (Sweeney et al., 2008).

Seven out of eight BEM models created by Clarke et al (2010a) under both high and low GHG emission scenarios showed a decline in the bioclimatic space associated with blanket peat. All 7 simulations run on a British BEM model developed by Gallego-Sala et al (2010) based on a 2.0°C warming scenario by 2050; predicted a contraction of the blanket peatland bioclimatic envelope in England and Wales, and eastern parts of Scotland. The model predicts that under a high emissions scenario by 2071–2100 the blanket peatland bioclimatic space will crash to ~84% of its 1961–1990 distribution (Gallego-Sala et al., 2010).

In Ireland regional climatic models have predicted that over the next 50 years the annual average precipitation is predicted to decrease by 10% in winter and by 10-40 % in summer. BEM model studies have projected that by 2055 the suitable Irish climatic area of fens will have declined by 40%, raised bogs by 31%, Atlantic blanket bogs by 39% and Mountain blanket bogs by 31% (Jones et al., 2006; Sweeney et al., 2008).

Under future climate projections a number of models predict that blanket bogs may retreat to high altitudes where the temperature and precipitation regimes that define their BEM will persist (Clarke et al., 2010a; Clarke et al., 2010b). Predicted precipitation deficits will be least severe in the west compared to the east of the British Isles (Clarke et al., 2010a; Clarke et al., 2010b. Models predict a retreat of the upland ecosystem envelope over lower altitude areas in the east of Britain, while it will likely persist in the high-altitude areas in the west (Clarke et al., 2010a; Clarke et al., 2010b; Gallego-Sala et al., 2010). Precipitation deficits will be most severe in the south encouraging a northward retreat of the blanket peatland bioclimatic envelope (Gallego-Sala et al., 2010). The NW Scottish Highlands are projected to be the stronghold of blanket peatland in Britain in the future (Clarke et al., 2010a; Clarke et al., 2010b; Gallego-Sala et al., 2010).

The maximum annual temperature was found to be the preeminent limiting factor in the BEM distribution of upland montane plant species in Britain (Clarke et al., 2010a). The mean annual temperature for upland areas of Britain is only 0 to 4°C lower than that of lowland areas. The mean and maximum annual temperature of these upland areas will warm as a result of climate change even over the next 50 years (Clarke et al., 2010b). The predicted warmer and drier conditions in NW Europe during the summer will likely instigated the invasion of lowland species onto upland peats (Clarke et al., 2010a). These vascular plants have higher rates of transpiration and may accelerate the desiccation of the peat surface (Clarke et al., 2010a). This will increase mineralisation and nutrient release in the peatlands further altering the environment to the detriment of acidic and wet tolerant species (Clarke et al., 2010a; Clarke et al., 2010b).

BEM models for global blanket bog distribution show only small core areas persisting within each region in the future. Vast areas particularly in Britain and Ireland are projected to lose blanket bog cover (Gallego-Sala & Prentice, 2012). An increase in precipitation in Western Europe may happen if projected changes in the North Atlantic Oscillation come to fruition. This may in some way compensate for the increase in temperature. However should summer drought also increase in frequency as is predicted then the increase in precipitation in winter will not be enough to halt the decline in the distribution of ombrotrophic peatlands in the mid-latitudes of NW Europe (Gallego-Sala et al., 2010). A warming climate and altered precipitation patterns will open up between 9-39% additional new areas for colonization in NW Europe. Norway is predicted to undergo warming in the future which will allow for the spread of blanket bog into the high latitudes (Benstad, 2005; Gallego-Sala & Prentice, 2012). The new areas however are small compared to the habitat that may be lost (Gallego-Sala & Prentice, 2012). These models do not consider the habitat that will be lost in the future as the result of rising sea levels. Inundation of coastal peatlands and the conversion of freshwater peatlands to saline marshes may be expected as a result of global warming. Some inland areas may also become susceptible to flooding. Some areas in NW Europe will however be safe from rising sea levels. Northern Sweden and Finland are still undergoing isostatic uplift since the last Ice Age and their land uplift rates of 8-10 mm pa are far in excess of the likely 3-5 mm pa of global sea-level rise (Parish et al., 2008).

Both winter precipitation as and the frequency of flooding are projected to increase in the future (Hulme et al., 2002; IPCC, 2007). Such changes in precipitation will very likely increase erosion where vegetation cover is already reduced by drainage or overgrazing (Hope et al., 2010; Parish et al., 2008). This may well lead to increased amounts of DOC, POC fine sediments and heavy metals being washed into peatland streams (Hope et al., 2010). Increased loss of DOC and POC will negatively affect the C balance of the effected peatlands in the future (Worrall et al., 2011). An increase in flooding events would also increase the CH_4emissions from peatlands (Parish et al., 2008). The relationship between the WT level and CH4 has been described in Section 2.

Climate Change and high-latitude peatlands

In the far northern latitudes the changes in temperature and precipitation are to be the most extreme on the planet (IPCC, 2007). The effect of global warming on these ecosystems and in particularly on peatlands associated with permafrost will very likely be pervasive. Changes in the time of the snowmelt and thawing may result in irreversible changes in the hydrology and carbon balance of these arctic peatlands (Nykanen et al., 2003). The projected regional variability particularly in precipitation changes coupled with the temporal and spatial variability in differing ecosystems of the high artic make future predictions of change in ecosystem functioning and GHG flux difficult. In the high-latitudes global warming may cause an earlier snowmelt resulting in changes in the hydrology and carbon dynamics of arctic peatlands (Nykanen et al., 2003). It is projected that the growing season defined as the number of days above 5°C length will increase by 20-50 days by 2100. It is thought that the productivity response to a longer growing season will be greatest in the high latitudes because the predicted increase in precipitation will mean there will be much less drought stress to limit growth compared to the mid-latitudes (Parish et al., 2008). The non-permafrost peatlands and aapa mires of Fennoscandinavia may well benefit from an earlier and extended growth season (Aurela et al., 2004; Tornocai, 2006).

In a high artic heath ecosystem in Greenland for example the NEE of CO_2 was measured over five summers. The increased growing season and the development of warmer summers resulted in the NEE of CO_2 uptake being enhanced by - 0.16 gCm^{-2} per increase in growing degree-days during the period of growth. This would suggest that a warming climate without increased precipitation may cause an increase in C sequestration in the high Artic (Groendahl, et al., 2007). Measurement of the tundra atmosphere exchanges of CO_2 and water vapour in the Canadian Low Arctic showed that the tundra was a net sink for CO_2 carbon in all years. The snow melt occurred three weeks earlier in 2006 compared to the other two years this coupled with warmer air and soil conditions resulted in better growing conditions and in a larger CO_2 uptake (Lafleur & Humphreys, 2008). This suggests a warming climate will result in increased C sequestration in some peatlands in the high artic. It is important to note however that only summer time emissions were considered and CH_4 emissions were not even considered in these studies.

A thorough eddy covariance study was carried out by Aurela et al. (2004) on an aapa mire in northern Finland over six years. Measurements of the CO_2 exchange between a subarctic fen and the atmosphere revealed that the fen is acting as of 22 g C m^{-2} yr^{-1}. Year round measurement reveal that while the wintertime efflux is a significant component of the annual CO_2 balance it is the timing of snow melt and the length of the growing season that controls the annual CO_2 balance in aapa mire. Hydrometeorological variations throughout the growing season had only a minor effect on the annual C balance. For sub artic fens like the aapa mires of Fennoscandinavia a warming climate may therefore increase their carbon pool. It was found that there was a 2.0 g C m^2 yr^{-1} difference in the annual CO_2 balance per one-day change in the snow melt date. If these calculations are correct it is thought that the aapa mires may have increased their C sink by 11 g C m^2 yr^{-1} over the last 40 years.

A three year (2001–2003) of continuous Eddy Covariance measurements by Sagerfors et al (2007) on an oligotrophic, minerotrophic mire in Sweden also found that the mire was acting as a net carbon sink of over the three years with an average net uptake of 55 ± 7 g (mean ± SD) CO_2-C m^{-2} a^{-1}. Like the findings of Aurela et al (2004) the timing of the snow melt and thus the length of the growing season was the most important single factor determining the annual carbon balance. The CO_2 uptake during the growing season surpassed the CO_2 released during the non-growing season. The growing season uptake was 92 ± 10 g CO_2 C m^{-2}, of which ca. 40% or 37 ± 5 g CO_2 C m^{-2} was lost during the nongrowing season. It was also found that the timing of the growing season was important. The timing of the initiation of growth in spring had a greater effect on the annual budget compared to an equal period of time in autumn (Sagerfors et al. 2007).

An analysis of eddy covariance measurements of 12 wetland sites spanning temperate to arctic climate zones spread across Europe and North America was carried out by Lunt et al (2010). The length of growing season period once again found to be the most important factor describing the variation in summertime Gross primary production (GPP) and also Ecosystem respiration (R eco). The prolonged growing seasons projected to occur as a result of global warming will allow for a longer period of photosynthesis and decomposition and GPP and Reco. Gains in GPP can be expected to be offset by Reco. However the increased GPP be greater than the increase in Reco therefore an increase in C sequestration may be expected. Limitation however to the positive benefits of an extended growing season will be increased Reco during the autumn and winter; light limitation in the higher latitudes and water availability in more southerly and water stressed peatlands (Lunt et al. 2010).

The literature seems to support the theory that the elevated temperatures, increased growing season, and higher atmospheric CO_2 could increase biomass production and so C sequestration (Tornocai, 2006). Water table draw down will be less severe compared to more southerly peatlands. The aapa mires of this region are characterised by hydrological buffers may alleviate the worst effect of a warming climate while allowing them to take advantage of earlier snow melts (Aurela et al., 2004). The aapa mires may therefore respond to a warming climate by increasing their productivity and may sequester more carbon in the future.

Not all ecosystems will respond in the same way to change however. This is well demonstrated from a 5-year study (1999 – 2003) carried out by Kwon et al (2006) on adjacent Alaskan wet sedge tundra and moist tussock tundra ecosystems. During the summer the eddy covariance technique was used to measure NEE CO_2 exchange of these tundra ecosystems. Over the study period the wet sedge tundra was a sink for carbon of 46.4 - 70.0 g C m^{-2} $season^{-1}$, while the moist tussock tundra was found to be C neutral or was a source 60.8 g C m^{-2} $season^{-1}$. This demonstrates how different ecosystems and their respective C balances can respond differently to the same climate. Warming and drying increased R eco in the moist tussock tundra, causing a net loss of carbon. The wetter wet sedge tundra was presumably buffered against the drying conditions due to its connection to the local hydrology. This study also failed to consider the CH_4emissions. The warmer conditions and greater organic material associated with the increased productivity would be expected to have resulted in an increase in CH_4 emission as long as the water table did not lower substantially due to drying. This would have a significant effect on the RF of the wet sedge tundra yet it is not considered in this study.

The palsa mire ecosystems are shaped by the permafrost layer within the peat. Permafrost thawing is uncertain and dependant on local controls related to vegetation composition. Therefore the effects of permafrost thawing on C sequestration and GHG efflux may be expected to vary (Limpens et al., 2008). Two different scenarios however are predicted.

(1) The melting of the permafrost may cause the drawdown of the WT leading to desiccation of the palsa surface and thermokarst erosion (Gorman, 1991). The drying associated with a lowering of the WT would result in a loss in carbon due to oxidation and a reduction in CH_4 emissions (Nykanen et al., 2003).

(2) The thawing of permafrost in certain places may cause a shift to a wetter ecosystem. This will result in greater NPP with an associated increase in C sequestration. Wetter conditions can be expected to be accompanied by an increase in the production of CH_4 (Holden, 2005; Lund, 2009). To what extent this will be counter balances by an increase CO_2 sequestration from increased productivity remains to be seen (Parish et al., 2008).

A 29 year study over nine palsa mire sites in sub-arctic Sweden has revealed that warming in the area particularly in the last decade has resulted in the thawing of the permafrost layer and a deepening of the active layer in the peat. The active layers at all sites have become thicker with rates of up to 2 cm per year occurring over the last decade. Permafrost has disappeared at 81 % of the sites. The loss of permafrost from these ecosystems which span the high latitudes of the planet will lead to changes in biogeochemical cycling, biological, geomorphological, and hydrological processes that will more than likely lead to an increase in the greenhouse gases emissions to the atmosphere and provide a positive feedback to global climate change (Akerman & Johansson, 2008)

Nykanen et al (2003) using the static chamber technique studied the annual CO_2 exchange and CH_4 fluxes from a palsa mire in the subarctic zone of Finland over two climatically different years. Many of the predicted future climate changes such as a longer growing season higher rainfall and snow fall and permafrost melt were found to increase CH_4 emissions. Increased emissions of CH_4 were observed as a result of snow accumulation on the sides of the palsa mounds sides. The snow cover insulates the soil and raises the temperature within the palsa mound. This increases the rate of anoxic decomposition and the release of CH_4. The Increasing temperatures melt the permafrost associated with palsa mires. The wet conditions resulting from the melting permafrost caused an increase in CH_4 emissions (Nykanen et al., 2003). The warmer winter temperatures within the palsa mounds associated with rising temperatures and greater snow cover was sufficient to allow decomposition of organic material to also take place during winter. Therefore increased CH_4 emissions may also be projected for the winter. The melting of the frost layer in July was found to liberate trapped CH_4 from the peat. Following this release the increasing temperatures and the new labile C associated with the vegetation causes a steady increase in methanogenesis and the release of CH_4 over the course of the summer. An earlier and longer growing season may therefore be expected to increase the CH_4 liberated from palsa ecosystems in the future (Nykanen et al., 2003). A particularly wet summer in 1999 meant caused an increase in CH_4 emissions. It was thought that the high water tables in 1999 may have caused degradation in the peat and a release in CH_4 (Nykanen et al., 2003).

A study by Johansson et al (2006) on the Swedish artic palsa mires documented the changes that have happened in this threatened peat ecosystem from 1970 – 2000. The study revealed similar trend pervasive trend of palsa mire degradation due to warming and increased emissions of CH_4. Over this twenty year time span the mean annual air temperature in the region has risen as has winter precipitation and snow cover. Increasing snowfall in winter has occurred insulating the palsa peat mounds and preventing the low winter temperatures penetrating deep into the peat. Warming temperatures have caused a thaw in the permafrost. As the upper layers thaw the water table has been drawn downward by the retreating permafrost. This has caused the surface peat of the palsa mires to become desiccated and degraded in Stordalen mire in Sweden. The effects of permafrost thawing in this study included the deepening of the active layer and soil subsidence. The thawing of the permafrost resulted in changes in the hydrology and vegetation compared to survey conducted in 1970.

Over the space of three decades the vegetation of the palsa mire switched from Sphagnum dominated hummock vegetation to wet-growing plant communities. Hummock vegetation was found to have declined by 11%. These vegetation changes in a peatland may be expected to alter the NEE, long term C and GHG balance of peatlands in two main ways: (1)

by changing the net primary production through increased GPP due to greater photosynthesis but higher autotrophic respiration. (2) The litter deposited by the vascular plants is less recalcitrant than the proceeding hummock litter. It is more degradable and will increase the C flux (Johansson et al., 2006). A switch from Sphagnum dominated communities to vascular plant communities may also be expected to increases the aerenchymal transport thereby causing an increase in the amount of CH_4 being emitted to the atmosphere (Nykanen et al., 2003).

The longer growing season, warmer temperatures and altered plant community caused an increase in GPP and the sequestration of the C but a concurrent increase in heterotrophic respiration caused emissions of GHG to increase. The changes resulted in the ecosystem increase its C sequestration by 16% but also increase CH_4 emissions with 22%. Over three decades the peatland had a 47% greater radiative forcing over a 100-year time horizon mainly as a result of increased CH_4 emissions. The changes observed in vegetation are estimated to have caused an increase in the net CO_2-C influx to the mire during the growing season by 3.8 gm^{-2} (15.5%) and the net CH_4-C efflux by 1.5 gm^{-2} (22.2%) (Johansson et al., 2006).

These results are supported by the findings of Christensen, et al (2004). Significant changes to the permafrost, vegetation and GHG emissions were documented to have occurred in the Swedish subarctic peatlands as a result of warming between the years 1970-2000. Thawing of the permafrost has resulted in the deepening of the active layer and an increase in the wetness of the peat. These warmer wetter conditions have seen over the period 1998–2002 the vegetation switch from ombrotrophic shrub communities to wet vascular graminoid dominated communities that are usually associated with minerotrophic conditions. The movement of the ecosystem towards a warmer wetter more minerotrophic ecosystem resulted in an increase in the landscape scale CH_4 emissions from 1970 to 2000 ranging between 22 and 66% (Christensen, et al., 2004).

Discussion

The varying climate projections and the diversity of peatland ecosystems make predicting the future impact of global warming on the peatlands of NW Europe extremely difficult (Johansson et al., 2006). This is especially true for peatlands of the sub-Arctic where the heterogeneous landscape with its humps and hollows is a patchwork of different ecosystems with their own unique characteristics (Charman, 2002; Seppä, 2002). Uncertainties about future climate change particularly the projected changes in regional precipitation make predictions even more difficult (IPCC, 2007). Their response to changes in temperature, precipitation, hydrology and vegetation will thus vary between ecosystems and will show regional variations. It is projected using BEM models that palsa mires will probably lose their suitable climate space in Fennoscandinavia due to warmer temperatures and higher precipitation in the future. Palsa mire will unlike other mire types will be unable to shift north as temperatures rise. This is due to the fact that the peat accumulated in the tundra mires to the north is not deep enough to allow palsa mound to form. Their restricted biological envelope makes palsa mires extremely vulnerable to a changing climate and they can be expected to become a rare ecosystem in the future (Parviainen & Luoto 2007). Aapa mires and raised bogs and blanket bogs have comparatively larger biological envelopes and they will be capable of migrating north into territories vacated by the palsa mires in the future (Parviainen & Luoto 2007).

Projected changes will have a drastic effect on the hydrology and biogeochemical processes associated with the sequestration of GHG in the peatlands on NW Europe. Some peatlands will be positively affected by the extended growing season and warmer conditions. Increased photosynthesis will lead to greater quantities of C being sequestered in these peatlands. Increased GHG emissions due to soil warming and the erosion of existing peat C stocks are likely to occur as a result of climate change over the subsequent decades. These emissions particularly the increased CH_4 efflux to the atmosphere associated with a warmer wetter environment will create a significant positive feedback to climate change (Johansson et al., 2006). For example by the by mid-21st century the annual net flux of CH_4 from Russian permafrost regions is projected to increase by 6–8 Mt. This increase in atmospheric concentrations alone may lead to a 0.012°C increase in the global temperature (Anisimov, 2007). Conditions may be similar to those in the early Holocene when maximum peat expansion and accumulation occurred in Alaska. It is thought that the northern peatland C dynamics contributed to the peak in atmospheric CH_4 and the decrease in CO_2 concentrations during this period (Zicheng, et al., 2009). Peatland ecosystems have adapted to and survived climatic changes in the past (Gallego-Sala & Prentice, 2012; Moore, 2002). Over the course of millennia peatland ecosystem may be expected to adapt to the new conditions. Northern peatlands tend to occur within a mean annual air temperature range of –12° to 5°C and a mean annual precipitation range of 200 to 1000 mm (Zicheng, et al., 2009). Future changes in precipitation and temperature will therefore create the possibility for the expansion of peatlands northward. Peatland formation will in the future like the tree line move north into territories where due to the extremely low temperatures it has been impossible for them to exist. Light limitation in the far north however may become a limiting factor of peat accumulation. Peatlands have existed in further north in the past during warm interglacial periods it is likely they will reoccupy these territories and reinitiate the sequestration of C in these regions (Koshkarova, 1995; Lozhkina, et al., 2011). The heightened values currently detected in the North Atlantic Oscillation Index, rising winter temperatures, and increased precipitation in Fennoscandinavia may encourage the spread of peatlands into some boreal forest biomes (Crawford, et al., 2003). The extended growing periods and warmer conditions will lead to increased C sequestration in some peatlands which may cause a cooling of the atmosphere over the course of millennia due to the longer residence time of CO_2 in the atmosphere compared to CH_4. Many peatlands will be resilient to climatic changes however they cannot remain in a relict state indefinitely and the *Sphagnum* species associated with peat formation and C sequestration may be replaced by vascular plants and trees. In the short term the oxidation and erosion of peat will occur over hundreds of years releasing CO_2 into the atmosphere. The extent to which this process will be offset by reductions in CH_4 and N_2O is unknown however the loss of the carbon sequestration capacity of NW European peatlands will lead to a positive radiative forcing and warming of the atmosphere over millennia (Gallego-Sala & Prentice, 2012; Hop et al., 2010; Moore, 2002). What the end result of these counteracting processes of increase sequestration in some regions while conversely there will be peat oxidation and CH_4 and N_2O emissions from other regions is impossible to quantify at this point given the amount of unknowns. The extent of these processes will be heavily influenced by the extent to which anthropogenic management of peatlands exacerbates the deterioration caused by climate change. It has been estimated that losing only 12% of Britain's peatland carbon stocks would have the same effect on global warming as the total annual UK emissions of greenhouse gases from the burning of fossil fuels, carbon loss due to management (Hope et al., 2010). As ~50 - 84 % of British blanket bogs are projected to experience climate stress under different emission scenarios by the end of the 21st century it is extremely important that management regimes which may exacerbate future warming be legislated against (Gallego-Sala et al., 2010; Gallego-Sala & Prentice, 2012; Hope et al.,2010). Reducing greenhouse gas emissions from land use and land use change and encouraging the regeneration of drained peatlands to reduce CO_2-emissions, secured C stocks and reinitiate C sequestration should be high priority's for on the international climate change mitigation (Essl, et al., 2012).

Conclusion

The peatlands of North West Europe play a significant role in the global climate. They have been acting as sinks for CO_2 since the Last Ice Age. Covering a mere 3% of the world's terrestrial surface, peatlands contain 550 Gigatonnes (Gt) of carbon making them the most important long term carbon sink in the terrestrial biosphere. This ability of peatlands to store carbon for millennia means they have a net cooling effect on the global climate. It has been estimated that in the last 10,000 years since the last Ice Age the atmospheric carbon sequestered in peats has served to reduce global temperatures by about 1.5–2 °C. Peatlands however are responsible for producing all three of the most important long lived greenhouse gases CO_2, CH_4, N_2O. The same wet and anoxic conditions that lead to the slow decomposition and so the sequestration of CO_2 also cause peatlands to be significant emitters of the potent greenhouse gases CH_4 and in some cases also of nitrous oxide N_2O. Wetlands are the largest source of atmospheric CH_4 surpassing all anthropogenic emissions while two thirds of N_2O emissions to the atmosphere come from soils. The greater radiative forcing of CH_4 and N_2O mean that over a time span of 100 years many pristine peatlands may be positive radiative force on global warming. However CO_2 has a much longer residence time in the atmosphere compared to CH_4 and N_2O and therefore over the span of millennia the sequestration of CO_2 in the form of peat means that natural peatlands have a net sinks for GHG and have a cooling effect on the global atmosphere. The drainage and degradation of pristine peatlands causes the release of considerable amounts of CO_2, DOC, POC, sediment and N_2O. Reduction in CH_4 emissions occur as the result of increased oxidation in the peat profile. This decrease in CH_4 and an initial increase in CO_2 sequestration associated with an increase in primary productivity may cause a decrease in the GWP of peatlands over short time periods such as centuries. However the inevitable loss of C stocks and the loss of the C sequestration capacity of peatlands through the minimisation and erosion of the peat profile mean that over longer time spans peatland degradation positively effects global warming. The UNFCCC estimates that 3 Gt of CO_2 may be emitted from degraded peatlands annually. This is equivalent to over 10% of the total global anthropogenic CO_2 emissions from 1990 or 20% of the total net 2003 GHG emissions of the Annex 1 Parties to the UNFCCC. It is obvious that man is causing the degradation of peatlands directly through poorly informed management decisions but also indirectly on a huge scale through the process of anthropogenic climate change.

The response to changes in temperature, precipitation, hydrology and vegetation will thus vary between ecosystems and will show regional variations. Some ecosystems such as palsa mire due to their restricted biological envelope will be extremely vulnerable to a changing climate. The melting of permafrost will lead to the release of vast amounts of CH_4 and CO_2 that will provide a positive feedback to global warming. Some peatland ecosystems may benefit from a longer growing season with increased primary production. This may have a positive effect on C sequestration although a warmer climate with more regular droughts will cause the lowering of peatland water table and the invasion of vascular plants and trees onto the peat surface. Over the short term these changes will likely see the release of vast amount of GHG that will further warm the climate. Peatlands however have adapted to climate change in the past and warmer temperature will open up new territories were plaudification may be reinitiated and regions of the far north may once again sequester C. Over the historical time frame the changes that are predicted for the climate in the future are unprecedented. It is therefore very difficult to predict how the diverse peatland ecosystems will respond to climate change in the future with any great accuracy. What is clear is that the extent of peatland degradation due to climate change may be exacerbated or ameliorated depending on how we alter our management of the peatlands of NW Europe in the future.

The rewetting and restoration of peatlands has the capacity to secure existing carbon stocks and reinitiate the C sequestration capacity of degraded peatlands. Restored peatlands therefore have the potential to ameliorate the global warming. The contribution that restored peatlands can make to global climate change has recently been recognised and peatland restoration projects have been incorporated into the new Kyoto Protocol. Peatland restoration an important mechanism for the countries of NW Europe meets their national greenhouse gas targets.

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