

**Is Peatland Edge Woodland an Appropriate
Management Option for Afforested
Peatlands After Harvesting?**

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PhD

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“Coming back to where you started is not the same as never leaving”

Terry Pratchett, A Hat Full of Sky

General Abstract

Peatlands provide a range of ecosystem services including being a globally important carbon store. During the 20th century large areas of northern peatlands were degraded by afforestation, compromising their capacity to provide ecosystem services. There is increasing interest and investment in the restoration of afforested peatland but also debate and uncertainty as to how and when to restore peatlands. In 2014 and 2015 the Forestry Commission Scotland published new guidance for the management of afforested peatland in Scotland which included the novel proposal that some afforested peatlands should be converted to low density, low intensity, predominately native woodlands – termed Peatland Edge Woodland (PEW). This thesis investigates the concept of PEW, specifically focusing on:

- 1) How stakeholders with a professional interest in afforested peatlands and peatland restoration in Scotland have responded to the concept of PEW. The investigation shows that the concept of PEW has been interpreted differently by stakeholders of differing ideological viewpoints. Stances range from complete opposition to the concept, to identifying PEW as an innovative and useful approach to managing certain peatland areas; individuals interested in establishing PEW were found in varied stakeholder groups including public/private forestry and conservation NGOs.
- 2) How PEW impacts on carbon storage and greenhouse gas balance. The thesis explores different aspects of how PEW may impact on carbon storage and greenhouse gas balance. A PEW proxy habitat is shown to be capable of storing carbon in above-ground biomass, but much less than in conventionally restocked plantations indicating that relying on carbon sequestration in PEW trees may be ineffective. Another PEW proxy habitat showed a plot with stunted tree growth had lower overall greenhouse gas emissions relative to a treeless plot due to lowered peat surface methane emission without significantly affected carbon dioxide fluxes, while the trees themselves were only a weak source of methane.

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Declaration

I declare that this thesis is a presentation of original work and I am the sole author. This work has not previously been presented for an award at this, or any other, University. All sources are acknowledged as References.

1. General Introduction

This chapter gives background scientific, historical, and political context to the concept of Peatland Edge Woodland (PEW). The section '1.1. Background to peatland' introduces some basic background information on peatlands with a focus on peatlands in Scotland. Section '1.2. importance of peatland' overviews the global importance of peatland. Section '1.3. History of forestry on Scottish peatland' introduces the historic context which gave rise to concept of PEW. Section '1.4. Future of forestry on Scottish peatland' discusses the modern guidelines which first described the concept of PEW. Section '1.5 Impacts of the peatland forestry management options' discusses current understanding of the non-radiative forcing related impacts of PEW compared with alternative management options. Sections 1.6-1.8 aim to describe the potential impacts of PEW on radiative forcing. Section 1.6. gives background information on the natural peatland processes which affect radiative forcing, then Section 1.7 describes methods to study changes in these processes and finally Section 1.8. describes what current research indicates about the impact of PEW on radiative forcing relative to alternative afforested peatland management options. Finally, Section 1.9. outlines the overall aims and objectives for this thesis

1.1. Background to peatland

1.1.1. What is peatland?

Peat is an organic rich soil (e.g. >60% organic) which forms where the rate of decomposition of plant material has been exceeded by plant primary production resulting in the accumulation of partially-decayed plant material (Clymo, 1978). Peat most frequently accumulates in waterlogged environments where anoxic, acidic, and nutrient-poor conditions inhibit decomposition. An area where peat forms the top soil layer is known as a peatland, regardless of whether the peat is still actively accumulating (Joosten and Clarke, 2002).

Precise definitions for the depth of organic soil required for an area to be considered a peatland vary. A distinction based on depth is often drawn between shallow peaty or organo-mineral soils such as peaty gleys, and deep peat or organic soils that form peatlands. Different authorities use different depth standards (Vasander and Kettunen, 2006; Morison et al., 2010).

Recent major Scottish mapping projects and policy documents have used 40, 45 and 50 cm criteria with 50 cm being chosen in the most recent Forestry Commission deep peat management practice guide (Forestry Commission Scotland, 2015). This small discrepancy does not result in substantial differences in the areas considered deep peat (Morison et al., 2010; Bruneau and Johnson, 2014). Historically, significantly different peat depth criteria have been used in Scotland such as 100 cm, which was the old standard for the Forestry Commission (Patterson and Anderson, 2000). From a regulatory perspective an area classed as deep peat may receive substantially more protection than shallow peat and therefore substantial changes to the depth criteria may have large management and environmental impacts. In this thesis the term peatland will typically be used to refer to an area with >50 cm depth of peat.

Peatlands are found on approximately 3% of the Earth's surface and have a global but uneven distribution (Limpens et al., 2008). Temperate and (sub-)Arctic peatlands are the most expansive but there are also substantial areas of tropical peatlands (figure 1.1) which are relatively poorly understood and charted (Xu et al., 2018; Murdiyarso et al., 2019). Global peat biomass is significant because the partially decayed plant material in peat contains carbon, making peatland a substantial carbon store. Boreal and subarctic peatlands alone are estimated to store between 270-370 Pg C (1 Pg=10¹⁵ g) in their peat (Turunen et al., 2002), equivalent to over a third of the carbon present in the atmosphere and 15-30% of total global soil carbon (Limpens *et al.*, 2008).

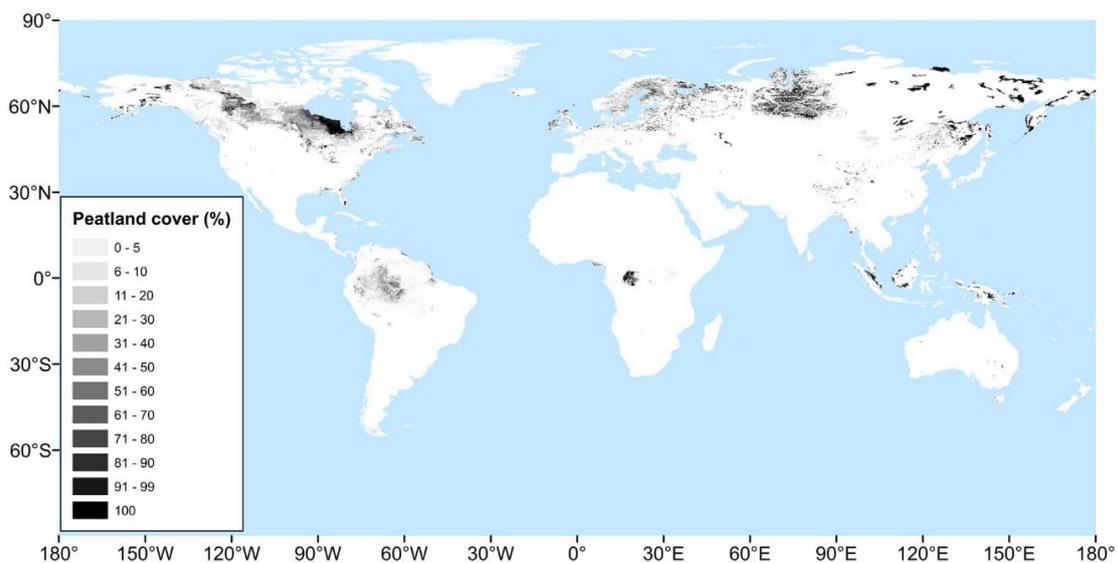


Figure 1.1: A map showing global distribution of peatland. Image produced by Xu et al. (2018).

1.1.2. Active peatland

Peatlands that are still actively accumulating peat or have the vegetation capable of forming peat are termed active peatlands or mires. These will typically have a high water table and the presence of particular peat forming species such as *Sphagnum* species and *Eriophorum* species. Active peatlands are often considered to have two layers: an acrotelm at the surface and beneath this the catotelm (figure 1.2) (Ingram, 1978). In the acrotelm, water saturation varies seasonally, hydraulic conductivity is relatively high, aerobic conditions are more prevalent and subsequently rates of decay will be higher. This is also the layer which includes the living plants that are actively accumulating biomass. For a three-layer model, the term mesotelm is used to further distinguish a lower part of the acrotelm to account for significant chemical differences within the layer (Clymo and Bryant, 2008; Lin et al., 2014). According to this classification the acrotelm is the zone comprising the top 10-20cm which is predominately oxalic; while the mesotelm zone below it is predominately anoxic but periodically oxalic due to water table fluctuations (Clymo and Bryant, 2008). The mesotelm will also be significantly less porous than the upper acrotelm (Clymo and Bryant, 2008). Beneath the acrotelm and mesotelm lies the catotelm, which begins below the minimum height of the water table. The catotelm has a higher bulk density and lower hydraulic conductivity than the layers above, it is predominately anaerobic so rates of decay are low. Underlying the catotelm will be a mineral substrate or bedrock that the peatland originally formed on. The depth of peat and its low permeability effectively insulate the ombrotrophic upper layers of the peatland from the influence of nutrient enriching ground waters from these sub-peat layers.

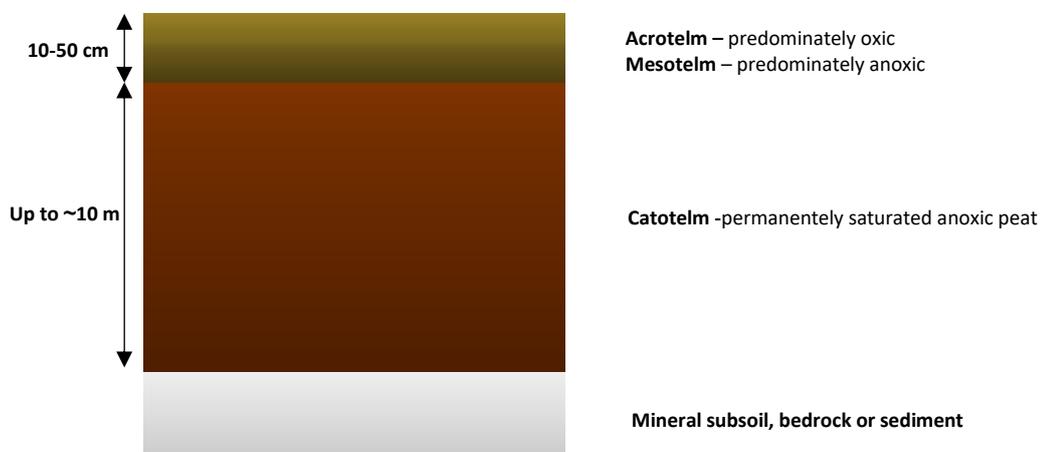


Figure 1.2: Showing the layers through a peat profile.

Living plants may grow rapidly in the acrotelm but only a small fraction of this biomass will form peat. Peat accumulation is usually considered to have occurred when partially-decayed material passes from the acrotelm/mesotelm to the catotelm where rate of decay will be greatly reduced (Lindsay, 2010). Active peatlands can vary in the speed they form according to a range of factors, but in the UK a typical accumulation rate would be ~0.5-1 mm a year (Lindsay, 2010; IUCN, 2017). Residence time of organic matter in the acrotelm before transfer to catotelm is difficult to quantify but Lindsay (2010a) estimates it to be about 80-100 years.

1.1.3. Natural history of peatland in Scotland

The first peatlands started to form in Scotland during the early Holocene with periods of further peatland initiation and expansion through time (Tallis, 1998; Gallego-Sala et al., 2016). Deep peatlands are now an important part of the Scottish landscape, covering approximately 23% of the land area, or approximately 1.8 million hectares (Smith et al., 2009; Lindsay et al., 2014a; Scottish Natural Heritage, 2015) (figure 1.3). Shallow peat/organo-mineral soils cover approximately 42% Scottish land area making the total area of organic soil in Scotland approximately 65% (Waldron et al., 2011) (figure 1.3).

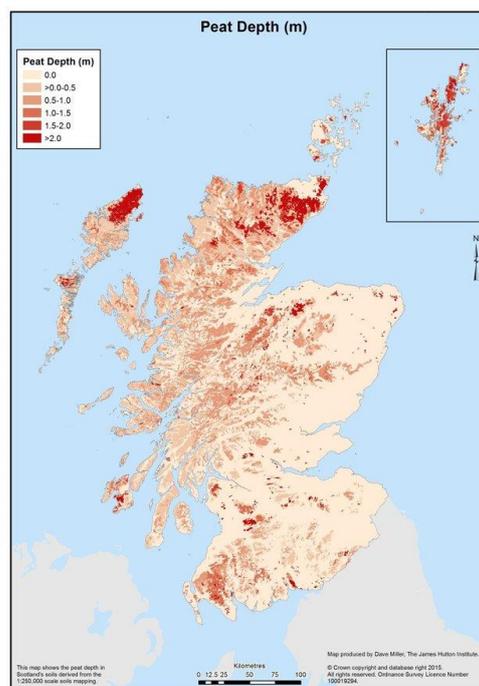


Figure 1.3: Map of estimated peat depth in Scotland, taken from Waldron et al. (2011), fig. 3.

Peatlands can broadly be categorised as ombrotrophic or minerotrophic. Ombrotrophic peatlands receive minerals almost solely from precipitation and are consequently more acidic and mineral poor than minerotrophic peatlands, which are fed by streams, overland flow or groundwater. Ombrotrophic peatlands can also be called bogs and minerotrophic peatland fens, however in common usage these terms are often misused (Lindsay et al., 2014a). Over 90% of Scottish peatlands are ombrotrophic (i.e. bogs) (Patterson and Anderson, 2000; Bruneau and Johnson, 2014; Lindsay et al., 2014a) and these peatlands are the focus of this thesis.

In Scotland, ombrotrophic peatlands are split into two types: 1) Lowland raised bogs; naturally domed peatlands which form on low-lying flat ground or in depressions filled by accumulated fen peat. The peat will be deepest in the centre of the bog and usually between 3-10m thick (Patterson and Anderson, 2000). Raised bogs are often naturally fringed with a minerotrophic lagg fen where the peat thins at the transition between the bog and the surrounding mineral soils. Lagg fens may consist of open habitat or wet woodland. 2) Blanket bog; the predominant peatland type in blanket mires, where the landscape is covered in a continuous peatland that can cover flat ground as well as slopes of up to 10-25°. In Scotland it is typically 0.5-3 metres deep, but can reach depths of up to 8 meters (Patterson and Anderson, 2000; Scottish Natural Heritage, 2020). A landscape dominated by blanket bogs may also contain areas with minerotrophic influence, such as around streams. The topography of blanket bogs is much more closely related to the underlying mineral landform morphology than that of raised bogs.

A majority of all peatland in Scotland is blanket bog (Artz et al., 2012a; Bruneau and Johnson, 2014). In particular the counties of Caithness and Sutherland are covered in approximately 400,000 hectares of blanket bog – an area collectively known as the Flow Country (figure 1.4). This is the largest area of blanket bog in Europe (Lindsay et al., 1988). Scotland contains 74% of the UK's and approximately 10% of the world's blanket bog area (Waldron et al., 2011).

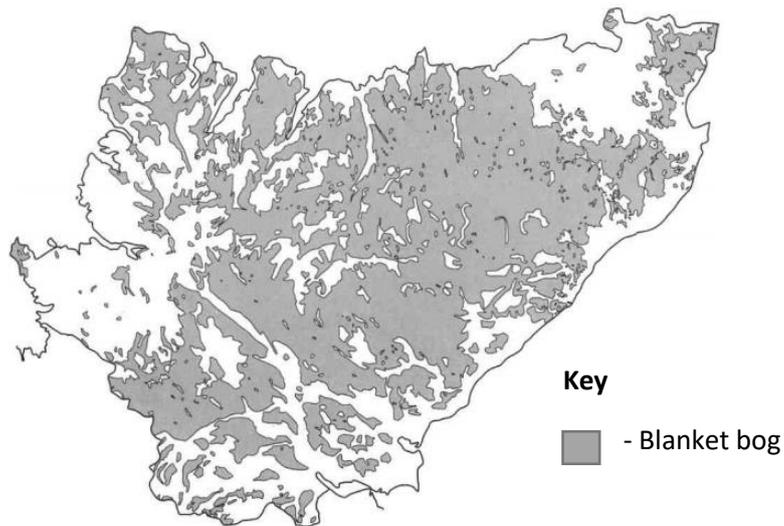


Figure 1.4: A map showing the total area of blanket bog in Caithness and Sutherland, taken from Lindsay (1988), fig. 27.

1.1.4. Bog woodland in Scotland

Most bogs in Scotland are typically thought of as naturally open habitats (Payne and Jessop, 2018). When trees are observed growing on Scottish bogs it is generally assumed that these are the product of artificial planting or progressive invasion of trees facilitated by artificial degradation of natural peatland conditions (Joint Nature Conservation Committee, 2007). However there are a small number of active bogs in Scotland which appear to have persistent tree cover, termed bog woodland (MacKenzie and Worrell, 1995). Despite its rarity in Scotland, bog woodland is relatively common on European and other temperate and boreal peatlands (Wells, 2001). Furthermore, bog woodland is internationally recognised as an important habitat and is listed under the European Commission’s Habitat Directive Annex 1. Two categories of Annex 1 bog woodland have been identified in Scotland; ‘*Sphagnum* birch woods’ and ‘Scots pine bog woods’ (MacKenzie and Worrell, 1995), with the latter being more predominant (Joint Nature Conservation Committee, 2008).

It is unclear to what extent the rarity of bog woodland in Scotland is natural. Some argue its rarity is a natural consequence of the wetter oceanic conditions found in Scotland compared to continental areas, which often experience lower summer water tables (Schouten et al., 1992). This is consistent with the fact that bog woodland occurs predominantly in the eastern Highlands

of Scotland (Moir et al., 2010), see figure 1.5. However, some argue that Scotland does not have significantly different climate conditions to some European countries where bog woodland is abundant (Halley, 2017). Instead, the rarity of bog woodland might in part also reflect differences in anthropogenic pressures which have resulted in the loss of bog woodland in Scotland, for example grazing pressures and the clearance of pine forests which could have acted as seed sources (MacKenzie and Worrell, 1995).

Scottish pollen and macrofossil evidence shows that some bogs were wooded in the early stages of formation (MacKenzie and Worrell, 1995; Tallis, 1998). However after peat depth became sufficient to create ombrotrophic bogs, woodland became uncommon, supporting the notion that Scottish bogs are not naturally wooded (Gear and Huntley, 1991; Moir et al., 2010). However, there is clear evidence that woodland expansion onto bogs and subsequent return to open bog is part of the history of many Scottish bogs. For example there was an expansion of pine bog woodland into northern Scotland starting in ~3200 BC, which had receded by 2900 BC; this has been attributed to climate change creating a drier climate (Moir et al., 2010). The fact that historical expansion of bog woodlands occurred under climates that are drier than the present climate suggests that even if Scottish peatlands are currently naturally treeless, there is an ecological continuity between open bogs and bog woodland.

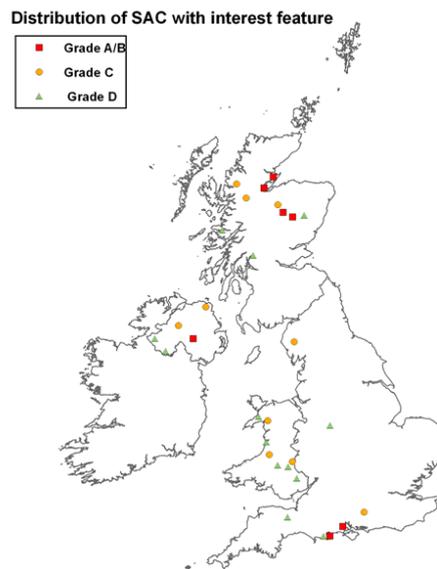


Figure 1.5: Map of distribution of bog wood in the United Kingdom. Grades correspond to quality of the habitats relative to SSSI notification standards (A being highest and D being lowest), produced by JNCC (<https://sac.jncc.gov.uk/habitat/H91D0/>).

1.2. Importance of peatland

1.2.1. Ecosystem services and nature-based solutions

Ecosystem services are the ways in which humanity directly or indirectly benefits from ecosystems (Millennium Ecosystem Assessment, 2005). Nature-based solutions are those that aim to actively protect, sustainably manage or restore natural or modified ecosystems in order to address societal challenges (Walters et al., 2016). The importance of the ecosystem services generated by peatland, and therefore the role they can play in nature-based solutions, are now broadly acknowledged in Scotland and by the international community (Hiraishi et al., 2013; United Nations, 2019a). This has led to substantial efforts and investment at a global scale to preserve and restore peatlands (Bonn et al., 2014; Alisjahbana and Busch, 2017). The following section provides an overview of these services under the four categories defined by the Millennium Ecosystem Assessment. These categories are: supporting, regulating, provisioning and cultural, and can overlap and be interdependent (Millennium Ecosystem Assessment, 2005).

Supporting

Supporting services are those that are necessary to produce other services (Millennium Ecosystem Assessment, 2005). These are usually long-term processes or cycles. For peatlands these include primary production and soil formation that initially created the habitat and could allow future expansion. Other peatland cycles like the slow rate of nutrient cycling are important for sustaining peatland conditions and its specialist flora and fauna such as *Sphagnum* mosses and *Eriophorum* species which in turn can support other ecosystem services.

Regulating

Peatland vegetation can sequester carbon dioxide (CO₂) from the atmosphere, a proportion of which can then be stored below ground in the peat for thousands of years. This store is substantial enough that the historic growth of peatlands is thought to have significantly cooled global climate since the last ice age (Yu et al., 2010). Smith et al. (2009) estimated that Scottish peatlands contain 1620 Mt of carbon below ground, compared to only 114 Mt in all of Great Britain's vegetation (Milne and Brown, 1997). In addition to long term carbon storage some active

peatlands also have a capacity to sequester more CO₂ from the atmosphere so they can continue to function as carbon sinks (Artz et al., 2012a).

Peatlands also regulate the purity and rate of release of water into watercourses. Degraded peatlands, such as those with artificial drains, exposed peat or those which have lost *Sphagnum* spp cover, can release water into watercourses more rapidly than those in good conditions (Bain et al., 2011) and affect its quality and thereby the ecology of surface waters. More rapid release of water can increase the severity of floods downstream and provide a more irregular water supply (Bain et al., 2011).

Cultural

Peatlands are an iconic landscape which support a specialised flora and fauna as well as being an important habitat for many ground nesting bird species. As peatlands are often found in areas with harsh climatic conditions and are generally hard to cultivate, they are often the parts of the landscape that are most preserved from human interference – thus providing spaces for recreations and connection with nature.

The low rates of decay typical of peatlands also preserve a wealth of archaeological artefacts and paleo-ecological indicators that can be insightful for understanding an area's natural and cultural history and prehistory.

Peatlands have a special cultural importance in Scotland where peatlands are a major component of the land area, providing a space seen as valuable by some and of little value by others but viewed by many Scottish people as part of their national identity (Martin-Ortega et al., 2017).

Provisioning

Clean water can be sustainably provided by peatlands with immediate financial implications. Peatlands are a significant component of many of the catchments that provide drinking water with suggestions that they may affect up to 70% of the UK's drinking water provision (Marsden and Ebmeier, 2012). Degraded peatlands can release substantial quantities of organic matter into watercourses which can discolour water and potentially react with chlorine during water treatment to produce carcinogenic trichloromethane (Armstrong et al., 2010; Bain et al., 2011; Lindsay et al., 2014b). Peat erosion can also result in the release of heavy metals in contaminated

catchments (Rothwell et al., 2005), requiring water companies to invest in more complex water treatment (Armstrong et al., 2010; Bain et al., 2011; Lindsay et al., 2014b). Clean water also supports the provision of river-caught salmon by providing more hospitable watercourses (Hendry and Cragg-Hine, 2003).

Peatlands have poor soils for most conventional land uses, despite which they are used for agriculture and forestry. Forestry and intensive agriculture often require substantial modification of the peatland to be viable so these practices are usually unsustainable; however there is a growing move to develop paludiculture and paludiforestry which aims to rewet peatlands used for farming or forestry while enabling lower intensity commercial land uses, for example growing water tolerant crops or tree species (Cris et al., 2014)

Direct products of peatlands such as peat and *Sphagnum* can be harvested from peatlands. *Sphagnum* collected from bogs has value in the horticultural industry, predominately as a potting medium. Peat can be used as a domestic fuel, fuel for power stations, to make growing media (e.g. potting compost) or in food smoking. Peat smoking is especially important for some of Scotland's most iconic and lucrative products including smoked salmon and smoked malt used in whisky production – an industry worth approximately £4 billion a year in Scotland (The Scotch Whisky Association, 2017). Direct harvesting of peatland products is only sustainable at a very small scale; large-scale exploitation of peat and *Sphagnum* is highly damaging and results in the decline of other peatland services.

1.2.2. Threats to peatlands

Peatlands in Scotland and internationally are under threat of being degraded by human modification such as drainage and nutrient enrichment. These modification scan lower water table depths, threaten peat-forming vegetation, and accelerate decomposition. When oxidative decay of the peat exceeds its accumulation, the result is progressive loss of peat. In Scotland approximately 90% of lowland raised bogs and 70% of the blanket bog areas are considered to be in a degraded state (Artz et al., 2013; Scottish Natural Heritage, 2015). Human activities that directly threaten peatlands include controlled burning, overgrazing, forestry, agriculture, peat cutting and commercial peat extraction (Scottish Natural Heritage, 2015).

Climate change in Scotland is predicted to increase average temperatures and increase seasonality in rainfall with winters being wetter and summers becoming drier (Werritty and

Sugden, 2012; Ferretto et al., 2019). These climatic changes, especially dryer summers, are predicted to make many areas in Scotland currently dominated by peatlands climatically unsuitable for these habitats, risking elevated CO₂ (Ferretto et al., 2019) and Dissolved Organic Carbon (DOC) emissions (Ferretto et al., 2021).

Wildfires in degraded peatlands can burn with greater severity than in near-natural or restored peatlands (Turetsky et al., 2015; Andersen et al., 2021). Human degradation of peatlands, artificial fire sources and climate change all make peatlands more vulnerable to fire. Fires have complex impacts depending on the geographic context of the fire, the time since the fire, and the severity of the burn, but can result in an increase in greenhouse gas emissions and ecological damage (Gray et al., 2021). Fires also have direct short term impacts on people, such as destroying property and threat to life, but in the long term can also cause substantial human mortality and morbidity through air pollution, with some estimating over 100,000 deaths annually (Johnston et al., 2012; Uda et al., 2019). The tropical peatlands of South East Asia and the degraded but now largely disused peatlands in European Russia are especially notable for the fires they've caused, but also for the subsequent restoration investment they are receiving to prevent fires in the future (Cris et al., 2014; Wetlands International Russia, 2020).

This thesis focuses on managing peatlands that have been used for forestry, which has been established on ~17% of deep peat in Scotland (Vanguelova et al., 2018). It is important to consider the multiple threats that could affect an area of peatland and the interaction between them. Future management of afforested peatlands needs to account for how resilient an area will be to threats under current and future conditions.

1.3. History of forestry on Scottish peatland

Historically peatland was often seen as wasteland, its soil being too wet and nutrient poor for conventional land uses. In the 1940s, Forestry Commission managers J. Alan and B. Macdonald (1945) described peatland as 'wasteland', 'derelict' and '...the wide desert where no life is found' in reference to a poem by Thomas Hood (Alan and Macdoñald, 1945). This perspective helped motivate attempts to utilise peatland for forestry, a practice that could create rural employment and improve national timber security from land which had limited direct financial value (Alan and Macdoñald, 1945; Zehetmayer, 1954).

Following the Second World War, technological advances such as the Cuthbertson plough and improved tractors began to make afforestation of peatland tenable on a large scale. It was now possible to plough peatland, creating drier ridges on which trees could be planted (figure 1.6), while the furrows collected water and connected to larger drains that took water off the site (Zehetmayer, 1954; Paterson and Mason, 1999). Continued advances in technology and planting practices furthered the expansion of forestry on deep peat. Lodgepole pine (*Pinus contorta*) was widely planted as it could tolerate the poor growing conditions better. A later innovation was to plant lodgepole pine and Sitka spruce (*Picea sitchensis*) together as the lodgepole pine was theorised to be a good nurse species that could facilitate the establishment of the more economically desirable Sitka spruce (Stroud et al., 2015).

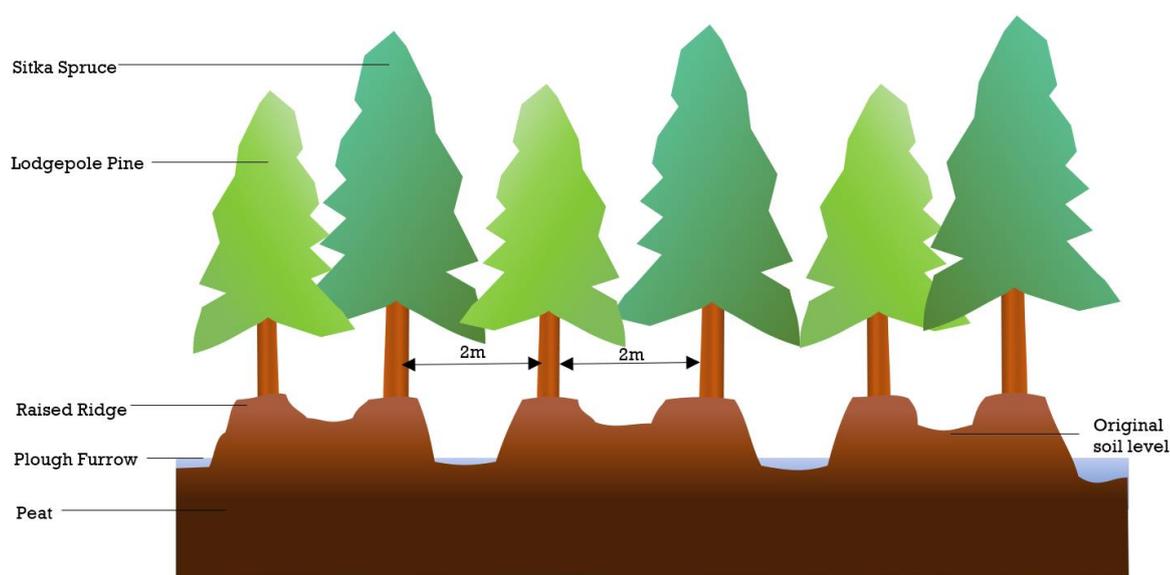


Figure 1.6: A representation of a cross-section through the profile of a plantation on deep peat. The example arrangement depicted would have been created through use of a double mouldboard plough every four metres, with trees planted on the ridges that would be produced at every two metres. This approach is quite typical of existing peatland plantations.

Expansion of forestry in Scotland was actively incentivised by the government through planting grants and tax concessions, and supported by government forestry research and advice (Lindsay et al., 2014c). These incentives were predominantly established to address the dwindling UK timber resources, create a strategic reserve of timber, reduce reliance on imported timber and create jobs in rural areas. The combination of technological advances and incentivisation led to

greatly increased tree planting across Scotland in general (figure 1.7a) which has raised forest cover in Scotland from 5% in 1919 to 19% in 2021 (Woodland Expansion Advisory Group, 2012; Forest Research, 2021). This forest expansion has been dominated by monocultures of non-native conifers; of the approximately 1,400,000 hectares of woodland in Scotland (Forestry Commission, 2011) approximately 1,000,000 hectares (70%) are productive non-native conifer plantation (Woodland Expansion Advisory Group, 2012). The rate at which new areas are afforested in Scotland did decrease from the 1980s onwards but the last decade has seen some recovery in the rate of afforestation in Scotland (figure 1.7b).

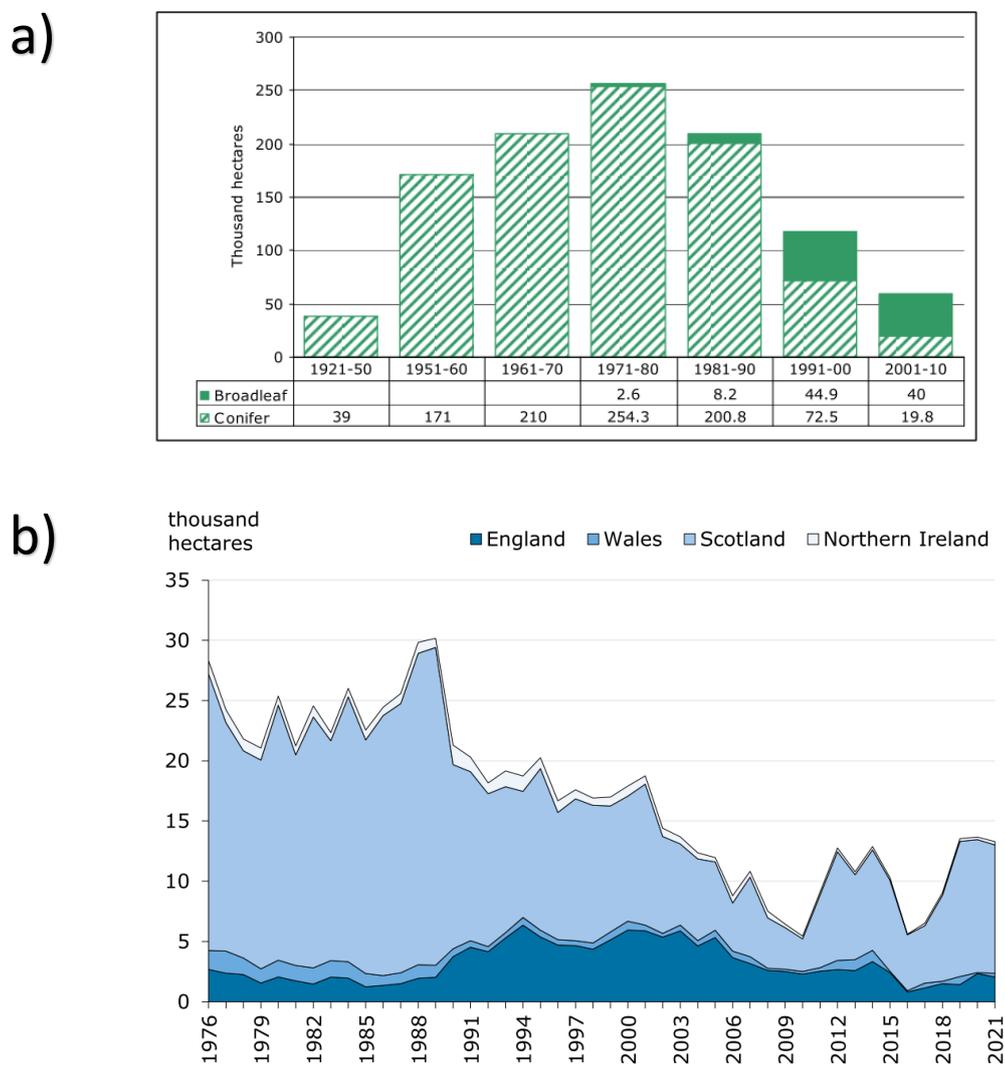


Figure 1.7: a) The average rates of woodland expansion since 1921, taken from the Woodland Expansion Advisory Group (2012). b) Although average rates of woodland expansion have fallen since the 1970s/1980s there has been some recovery of afforestation rates within the last decade in Scotland, taken from the Provisional Woodland Statistics 2021 Edition (Forest Research, 2021).

The low price of land associated with the uplands and peatlands across Scotland meant these areas were subject to a large proportion of 20th century forest expansion. By the 1970s levels of financial assistance combined with cheap land prices meant that forestry plantations could be profitably planted on peatland in some parts of Scotland with little consideration to the long-term viability or productivity of the plantation (Stroud et al., 2015). Furthermore, at this time a majority of the Scottish peatland in the best condition did not have any legal protection, leaving it vulnerable to forestry. A crisis point was reached in the early 1980s, when private forestry companies started establishing plantations across the Flow Country at a greatly increased rate (figure 1.8). In particular Fountain Forestry Ltd. Was especially active in promoting the afforestation of large areas of the Flow Country. At its peak it is thought that 67,000 hectares of the Flow Country, accounting for 17% of its land area, was thought to be planted or approved for planting (Lindsay et al., 1988).



Figure 1.8: A graph showing the rapid expansion of forestry in Caithness and Sutherland and in particular the expansion of private forestry in the 1980s (modified from Stroud et al. (2015)).

The scale and speed of the forestry expansion in the Flow Country prompted a corresponding increase in research and campaigning on its effects by organisations such as the RSPB and the Nature Conservancy Council. These culminated in high profile publications such as ‘Forestry in the Flows of Caithness and Sutherland’ (Bainbridge et al., 1987), ‘Birds, bogs and

forestry' (Stroud et al., 1987) and 'The Flow Country' (Lindsay et al., 1988). This research conclusively demonstrated the significance of bogs for biodiversity, and emphasised the threats posed thereto by forestry.

It was shown that forestry destroyed the original habitats and that the resultant mature closed-canopy plantations provided few biodiversity benefits (Stroud et al., 1987). Furthermore, there were concerns over biodiversity declines in surrounding open bog due to edge effects and habitat fragmentation (Stroud et al., 1987; Lindsay et al., 1988).

Based on research, campaigning employed emotive arguments highlighting the uniqueness of the Flow Country – sparking still-ongoing discussion about whether it should be designated as a UNESCO World Heritage Site (Stroud et al., 2015). Campaigners also exposed and publicised the extent to which this forestry was subsidised by the 'public purse' often to the significant profit of wealthy (and sometimes famous) individuals. Coincidentally to the campaigning, a national audit of the Forestry Commission also identified that the extend of subsidies available to private sector forestry represented a poor investment of public money (National Audit Office, 1986). In combination with this audit, the campaigning was highly successful; Nigel Lawson's 1988 budget abolished the tax concessions on forestry. Without incentives the expansion of forestry across the Flow Country, and other peatland supporting only very poor growing conditions, halted.

Before the legislative changes there was a move to designate large areas of the Flow Country as Sites of Special Scientific Interest (SSSI) to protect against the expansion of forestry plantations. This process was continued after the legislative changes, ultimately resulting in approximately 160,000 hectares of land designated as SSSIs (Stroud et al., 2015). Approximately 140,000 hectares of this was additionally designated as Special Protection Areas and a Ramsar site in 1999, and, later, a Special Area of Conservation under the Habitat Directive in 2005 (Stroud et al., 2015). These Flow Country sites constitute almost two thirds of the 221,000 hectares of Scottish peatland now classified as Special Areas of Conservation (Scottish Natural Heritage, 2015)

The Flow Country had been the focus of campaigning against afforestation of peatland in the 1980s, but the changes in legislation and awareness helped protect peatland across Scotland. In fact some argue that debate over the Flow Country was responsible for the shift in focus to more environmentally sensitive, multi-purpose forestry across the UK's forest industry (Warren, 2008). Multi-purpose forestry considers a wider range of forest uses in its planning and management, with greater sensitivities to the opinions of the public.

From the end of the 1980s onwards there has been little planting on new areas of non-afforested deep peat (currently defined as 50 cm by forestry legislation) and a strong presumption against this is stated in Forestry Commission practice guides (Patterson and Anderson, 2000; Forestry Commission Scotland, 2014). Even as early as 1990 the Forest Nature Conservation Guidelines stated 'They [wetlands] are important ecologically and should not be planted' (Forestry Commission, 1990). However, the question of what to do with sites that have already been planted is more problematic. During the 1990s there was some forest-to-bog restoration, including in the RSPB's Forsinard reserve from 1995. However, at this time the large-scale restoration of afforested bogs was not advocated by the Forestry Commission. The Forests and Peatland Habitats Guideline Note (Patterson and Anderson, 2000) describes full restoration of existing forestry plantations as "special cases". There was not considered to be enough evidence of the benefits delivered by bog restoration to justify this course.

In the 2000s there has been an increasing awareness of the value of carbon uptake and storage by natural habitats. For example the Scottish Forestry Policy (Forestry Commission Scotland, 2006) and the Scottish Government's Rationale For Woodland Expansion (Forestry Commission Scotland, 2009) put a large emphasis on forest expansion for climate change mitigation. Natural habitats and their conservation are increasingly viewed in terms of ecosystem services, natural capital and their potential to contribute to nature-based solutions. For peatland, this period has led to increasing focus on 'services' provided, especially carbon storage. Forestry Commission policy has reflected this shift, with older guidance emphasising the importance of peatland for biodiversity (Patterson and Anderson, 2000) and more recent guidance discussing peatland primarily in terms of carbon storage (Forestry Commission, 2014; Forestry Commission Scotland, 2015). Viewing peatlands in terms of 'natural capital' can justify financial investment in peatlands for economic reasons. Peatland restoration is now commonly discussed as a cost effective strategy both in political (Bain et al., 2011; Scottish Natural Heritage, 2015) and scientific literature (Moxey and Moran, 2014).

The increased awareness of the ecosystem services provided by peatlands, as well as improved scientific knowledge of the effects of peatland restoration, has resulted in significantly increased government funding for peatland restoration. In 2012 the Scottish Government established the Peatland Action restoration programme, administered by Scottish Natural Heritage (SNH). Peatland Action has funded restoration projects on over 25,000 hectares of peatland and in 2020 was granted £250 million over 10 years with the target of restoring 250,000 hectares (Scottish Government, 2020). There is also a range of other governmental funding for peatland restoration such as the Scottish Rural Development Programme (SRDP), Agri-

Environment Climate Scheme (AECS) and equivalent schemes in other nations of the UK. The EU LIFE Program and Heritage Lottery Fund have been other important sources of funding for many peatlands restoration projects.

There is also interest in developing public and private investment in peatland restoration by facilitating carbon off-setting through Voluntary Carbon Markets (VCM)(Bonn et al., 2014). The IUCN has recently developed the Peatland Code for the UK which is a voluntary standard for peatland restoration projects that can be followed with the intention of assuring potential investors of the benefits of the project (IUCN, 2017).

The International Union for the Conservation of Nature (IUCN) Peatland Program has recently proposed the ambitious UK wide target of ensuring that 2 million hectares of peatland are in good condition, under restoration agreements or being sustainably managed by 2040 (IUCN, 2018a). The fact that this is seen as a plausible target illustrates how much attitudes have changed.

Climate change is being treated with an increasing sense of urgency and severity. The Paris Agreement adopted at COP21 in 2015 was a globally ratified agreement to limit global temperatures to below a 2°C or preferably 1.5°C increase from pre-industrial levels. The COP26 summit in 2021 reiterated intention to limit warming to 1.5°C with 153 countries putting forward new emission reduction targets (or Nationally Determined Contributions (NDCs)) for 2030. The UK has proposed a 68% reduction in its national greenhouse gas emissions from a 1990 baseline by 2030 and to have net zero emissions by 2050. In order to achieve these targets, large changes will be required over a relatively short time frame across different sectors of the economy, including land management. This developing broader political context will continue to exert pressure on how peatlands are viewed and managed.

1.4. Future of forestry on Scottish peatland

1.4.1. Decision making framework

Although new deep peatland areas are no longer commercially afforested, many of the existing deep peat forestry plantations are now reaching maturity as they come to the end of their first rotation. This means that decisions must be made about their future management. Deciding future land uses for afforested sites is a complicated issue as there are a lot of uncertainties as to the impact of different management options and different stakeholders have different priorities. Forestry Commission Scotland (FSC)¹ released two documents to help land managers determine how afforested bogs should be managed after a first rotations, these are;

- Forestry Commission Scotland (2014) *Forestry on Peatland Habitats: Supplementary Guidance to Support the FC Forests and Peatland Habitats Guideline Note (2000)* – herein referred to as ‘The Supplementary Guidance’
- Forestry Commission Scotland (2015) *Practice Guide: Deciding Future Management Options for Afforested Deep Peatland* – herein referred to as ‘The Practice Guide’

These documents offer three options for the future management of a plantation. Two of these were extant, standard options: forest-to-bog restoration and conventional restocking with commercial species. The third option is to create a novel habitat termed ‘Peatland Edge Woodland’ (PEW). The guidelines propose PEW as a low density, low intensity predominantly native woodland – a new and controversial suggestion (RSPB, 2014).

The Practice Guide lays out a decision-making framework for deciding the future management of a peatland following felling (figure 1.9). Under its guidelines, some afforested peatland sites will be judged as a high priority for forest-to-bog restoration on ecological grounds, in which cases restoration will take precedence. The qualifying criteria for this are:

- If the plantation is on a habitat designated as a qualifying feature of a Natura site (i.e. Special Area of Conservation (SAC) and/or Special Protection Area (SPA)), Ramsar Site, Site of Special Scientific Interest (SSSI) or National Nature Reserve (NNRs) or if the site supports a habitat designated as a qualifying feature in the UK Biodiversity Action Plan.

¹ FSC has subsequently been restructured and renamed, the equivalent organisation is now called Scottish Forestry.

- If restocking the plantation is likely to adversely affect the hydrology of adjacent Annex 1 peatland habitats (as defined in the EU Habitats Directive) or a habitat associated with such habitats.
- If the removal of the plantation will prevent the significant net release of greenhouse gases.

If the site is not a priority for restoration, and conventional restocking using minimal cultivation and fertilisation will be able to support a reasonable yield of timber then conventional restocking should be favoured. The guidelines use a criterion of a general yield class 8 for Sitka spruce as the minimum productivity required to be eligible for restocking. General yield class is the maximum mean annual increment of cumulative timber volume for an even aged stand of a given tree species measured in cubic metres per hectare per year ($\text{m}^3 \text{ha}^{-1} \text{yr}^{-1}$) (Mathews et al., 2016). Yield class 8 of Sitka spruce therefore means that at its maximum growth rate, a one hectare stand of Sitka spruce would be expected to increase its timber volume by 8 cubic meters per year.

If a site is not suitable for conventional restocking but the Ecological Site Classification (ESC) data suggests the site could support >20% canopy cover then this site could be restored to PEW. If it could not support this canopy cover the guidelines recommend that the site should undergo forest-to-bog restoration.

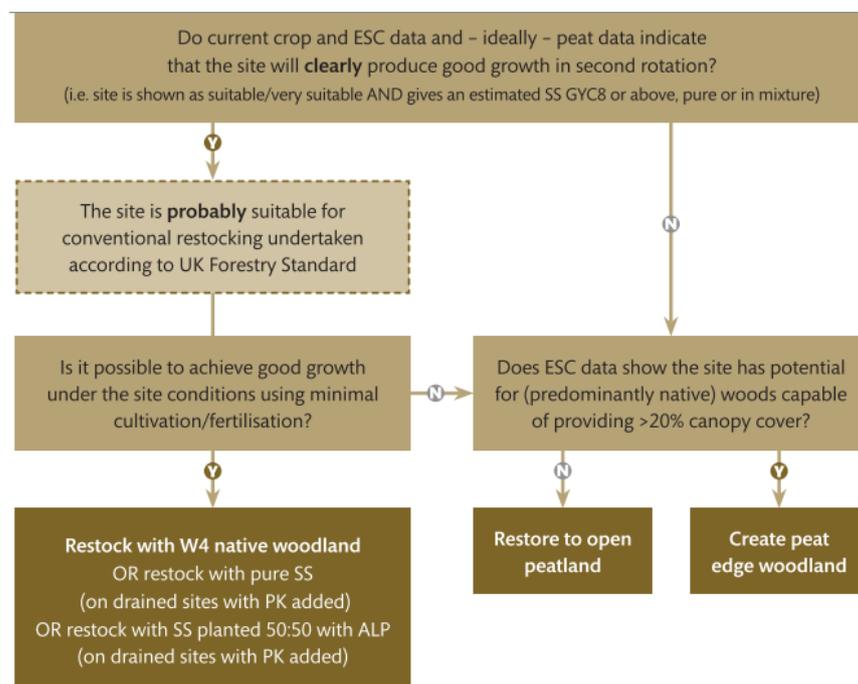


Figure 1.9: A flowchart aiding decision making for deep peat sites which are not designated as a priority for restoration. Taken from Forestry Commission (2015).

The Practice Guide pertains to areas of deep peat (defined as >50 cm deep), so shallow peatlands are not protected by this guidance (unless associated with deep peat areas). Consequently, new areas of shallow peat are still being afforested and existing plantations on shallow peat are readily restocked. The lack of protection of shallow peats is currently a contentious issue (IUCN, 2020). Research has highlighted that high productivity plantations established on shallow peat with low disturbance techniques will probably result in net carbon uptake due to carbon sequestration in the trees compensating for losses from the soil (Vanguelova et al., 2018) but also that the short term impacts of afforestation do result in significant losses of soil carbon that may take two rotations to be recovered (Vanguelova et al., 2019). With a rotation of Sitka spruce typically lasting between 35-45 years in the UK (Moore et al., 2012) it is likely that in many situations net zero emissions would be achieved after the UK national net zero by 2050 target and therefore be countering progress towards this target. New guidance on afforestation of shallow peats has been released which advocates for less intensive ground preparation techniques and emphasises the need for proper peat depth assessment (Scottish Forestry, 2021). Many peatland areas are highly heterogenous, comprising of a complex matrix of areas that are classed as shallow and deep peat. The new guidance highlights that woodland may not be appropriate in areas which are predominantly shallow peat but have a widely distributed matrix of deep peat areas (Scottish Forestry, 2021).

1.4.2. Management options

This section describes the techniques employed for the three management options in greater detail.

Second rotation forestry

Deep peat plantations were originally established with high disturbance techniques including extensive ploughing, draining, application of fertiliser and often herbicide to reduce competition from heather. Forest practices on deep peat have changed since the plantations were first established, to reduce the amount of disturbance caused to the peat while still encouraging the growth of the trees. As a prerequisite for deciding to conventionally restock an area the site is assessed for whether it would support good growth in a second rotation with the use of only minimal cultivation or fertiliser. This can be determined through the Ecological Site Classification

(ESC) decision support system which assesses which tree species are ecologically suited to a site without it being intensively modified. The Practice Guide now prohibits ploughing on deep peat, instead the maximum disturbance that is permitted is planting on mounds with a maximum size of 50 x 50 x 30 cm with a maximum drain density of 250m/ha. This has the additional benefit of encouraging better root spread instead of roots preferentially spreading along plough ridges (Paterson and Mason, 1999). Phosphate and Potassium fertiliser can be applied during restocking but only to get the trees established, not on an ongoing basis to boost their growth, and nitrogen fertiliser is prohibited by the Practice Guide. Species composition options for second rotation plantations are similar to the historical ones; either pure Sitka spruce or a 50:50 ratio of Sitka spruce and lodgepole pine (figure 1.10a) or other nurse species. Mixtures are encouraged for nutritionally poor sites since they can produce a final crop mainly of Sitka spruce without requiring ongoing fertiliser applications. There is also the additional option of commercially planted W4 downy birch (*Betula pubescens*) native wet woodland (figure 1.10b).



Figure 1.10: Different models of conventional restocking. a) Restocked by planting lodgepole pine and Sitka spruce mix, utilising pre-existing plough furrows and drains but without additional ground preparation. b) Restocked by allowing natural birch regeneration while actively maintaining pre-existing drains.

Forest-to-bog restoration

Forest-to-bog restoration techniques in Scotland have been developed experimentally since the earliest restoration projects in the 1990s and continue to be developed. Restoration practice drew on national and international experience of peatland restoration in other contexts such as those for restoration of cutover peat or drained moorland (Wheeler and Shaw, 1995).

The felling of trees and raising the water table (or 'rewetting') are the two fundamental components of most forest-to-bog restoration projects (Anderson and Peace, 2017). Trees prevent the recovery of bog vegetation by drying the peat and shading the ground through canopy cover and litter fall which also provides a nutrient-enriching leaf/needle litter (Stroud et al., 1987; Limpens et al., 2014). Trees are sometimes removed as part of forestry operations before restoration, but this can not necessarily be done profitably if the trees have grown poorly or are being removed before maturity. Historically it was speculated that managing the site to raise the water table might be sufficient to cause trees to die and topple, thus saving expense of their felling. However, trials of this approach have been shown to be ineffective (Anderson and Peace, 2017). Trees therefore have to be felled. A secondary problem is controlling the regeneration of the original conifer crop species (lodgepole pine and Sitka spruce) or preventing colonisation by birch after felling. If left unchecked at some sites regeneration would have the potential to establish closed canopy woodland (Andersen et al., 2017).

The cheapest felling option is 'felling to waste' or 'felling to recycle', meaning that after felling the trees their trunks and brash are left on site. Felling waste can inhibit vegetation recovery (Anderson, 2010), potentially as a result of shading or fertilising effects, and brash mats are also associated with higher rates of tree regeneration (Anderson, 2010). More intensive and expensive felling options are increasingly used, such as whole tree removal, brash mat removal and on-site tree mulching (figure 1.11a). These may accelerate the recovery of the bog vegetation and reduce potential for the trees to regenerate, thus potentially saving money in the future (Anderson, 2010; Andersen et al., 2017). To date most studies on the impacts of forest-to-bog restoration on in-situ gas fluxes and biodiversity have focused on the older, simpler restoration techniques so the relative effect of the different methods is poorly known. The additional disturbance caused during more intensive restoration work may, for example, have more extreme effects on stream water quality and aqueous losses of carbon in the short term (Howson et al., 2021a). However there remains uncertainty over the relative impact that different methods have on the in-situ decomposition of woody material and its downstream effects in the long term (Andersen et al., 2017; Gaffney et al., 2020; Howson et al., 2021a).

a)



b)



c)



Figure 1.11: Forest-to-bog restoration techniques. a) Recent mulching. b) Peat dams in a main drain. c) An area shortly after stump flipping.

Tree removal may help with raising the water table, but it is also important to ameliorate the effects of the original ground preparation. The larger drains will typically be blocked by building dams with peat (figure 1.11b) (Lunt et al., 2010). Drain blocking can be very effective, especially on flat sites (Anderson, 2001). Further rewetting can be achieved through blocking or filling in the old plough furrows as well, although this incurs additional costs. As an alternative to discrete dams, furrows can be filled by stump flipping or ground smoothing (figure 1.11c). In this approach tree stumps are pushed into the furrows to block them (Andersen et al., 2017). This reprofiling encourages rapid and more even rewetting, helping bog vegetation re-establish and reducing the amount of tree/scrub regeneration on dry patches (Scottish Power Renewables, 2015). Reprofiling also has aesthetic benefits as it softens the straight lines of the furrows.

Plough furrows can often suffer from peat cracking where vertical shrinkage cracks formed along furrows during prolonged droughts and remained after the droughts ended, allowing water to pass under dams. Severe peat cracking will also connect under plough ridges, allowing horizontal movement of water between furrows. Peat cracking can be remediated by 'backfill trenching', i.e. digging to below the maximum depth of the cracking and infilling the trench with intact, non-porous peat (Anderson, 2016; Artz et al., 2018).

Peatland Edge Woodland

Peatland Edge Woodland is the third management option suggested for afforested deep peat sites. It is envisaged as a relatively open, low intensity and predominantly native woodland by the Practice Guide and the Supplementary Guidance. The express aims of PEW are to avoid a net loss of carbon while providing biodiversity and landscape benefits. The guidelines discuss it as combining features of both peatland and woodland habitats to deliver these benefits. This implies at least some recovery of the peatland habitat towards its natural state, such as a rise in water table.

The details of how PEW should be created or managed are not precisely defined in the guides. In fact, they state that the advice will need updating based on the experience of practitioners who attempt to create the first PEWs. Instead of a strict specification, the guides define the objectives for PEW and offer advice as to how these might be achieved.

In the guidelines PEW is required to have at least 20% canopy cover. This cut-off would have been chosen for pragmatic rather than functional reasons as the Forestry Commission defines woodlands as areas with more than 20% canopy cover. The guidelines give a typical

specification of PEW as being 50% planted and 50% open with at least 500 trees per hectare. The supplementary guidance (2014) also stipulates that trees should be planted in groups with each tree in a group no more than 1.5m apart, although this stipulation is not repeated in the Practice Guide (2015). This would be a very high density of mature trees but allows for expected high mortality and stunted tree growth.

The guidelines state that sites can be stocked by natural regeneration if this produces 'acceptable results' in terms of stocking density and species mix at reasonable timescales, but the site may require planting to create the desired woodland. The guidelines stipulate that if planting is carried out it should be done to mimic natural spacing and with only the minimum cultivation needed to ensure satisfactory establishment and little or no artificial drainage. Little is known about how much carbon is lost from peat as a result of disturbance during planting and this would vary depending on which practices were used (Morison et al., 2010) but it is likely that planting trees would result in initially greater carbon losses than establishing trees through regeneration.

The guidelines do not give any strict guidance on the species composition of PEW other than that planting should be with native species in their natural range. Establishing the species composition of PEW will be particularly important for determining the character of the PEW and its impact on ecosystem services. Aspects that would be influenced would include the survival rate of the trees, the biodiversity the PEW supports, the impact the trees have on the peat they are growing on and the extent to which the trees will regenerate on and outside of the site.

The guidelines also state that 'an element' of non-native regeneration could be acceptable on sites that border other commercially planted blocks if this helps to achieve a positive carbon balance and does not threaten the growth of native trees. These criteria could lead to a range of interpretations of how much non-native tree cover is acceptable, an issue raised in criticism of the concept (RSPB, 2014). The guidelines do state that non-native regeneration should be controlled between the boundary of PEW and open peatland or native woodland to avoid trees of the non-native species encroaching on these habitats.

The guidelines state that the creation of priority native woodland habitats as per Annex 1 of the Habitats Directive is desirable. Birch or pine bog woodlands are the only obviously suitable Habitat Directive habitats to create, however bog woodlands are characterised by the stable combination of tree species with bog vegetation and whether it could be possible to create this on such disturbed sites is not known (Patterson and Anderson, 2000; Anderson and Harding, 2002). Other wet woodlands such as W4 *Betula pubescens* – *Molinia caerulea* are UK biodiversity action plan priority habitats and may also represent suitable habitats. These habitats can be

similar to birch bog woodland but their ground flora will not necessarily be dominated by typical bog species such as *Sphagnum* species (Joint Nature Conservation Committee, 2018)

The guides state that PEW sites should be created with little or no artificial drainage. The Practice Guide also says that planting should be done in such a way as to minimise disturbance and proposes hand turfing or hinge mounding with no drains as the most appropriate. It is unclear what should happen to pre-existing drains and plough furrows. The Supplementary Guidance mentions that PEW could be achieved for example by minimal cultivation and 'retaining little or no artificial drainage' but is open to question whether this means that pre-existing drains dug for establishment of the first rotation forest would be blocked. Drain blocking and other rewetting restoration practices such as reprofiling, stump flipping and furrow blocking are processes that could substantially increase the expense of PEW creation. Costs vary widely but a recent report found a mean projected cost of a forest-to-bog restoration project in Scotland of £3003.92 per ha (Glenk et al., 2022); if a PEW project involved rewetting the site to the standard of a forest-to-bog restoration project then the costs would be similar but would additionally include any tree planting costs. Drain blocking and other rewetting practices would promote recovery of the water table, which is important for reducing carbon loss from the below-ground carbon store, however it may also reduce tree growth and regeneration which would reduce the amount of carbon sequestration in above-ground biomass.

The guidelines on PEW do not specifically discuss fertiliser application. However, they do state that only the minimum cultivation necessary to ensure satisfactory establishment should be used. More generally, the guidelines also preclude the use of nitrogen fertiliser on peatlands. The guidelines therefore seem to support either no fertiliser application or minimal use of fertiliser such as phosphate and potassium fertiliser during establishment.

The guidelines suggest controlling deer populations to less than 5 per square kilometre and removing invasive shrub species. Both these management methods should support the growth of native trees.

Peatland Edge Woodland is described as a deep peat management option (defined as >50 cm) and there is no explicit guidance that its location in an area of peatland has to reflect peat depth other than this minimum depth criteria. However, the inclusion of the wood 'edge' in its name, as well as the description of it being suitable for areas of peatland intermediate between those suitable for restocking and open-bog-restoration, implies that PEW may be established in areas reflecting natural ecotones defined by physical characteristics such as peat depth. If PEW is seen as suitable for sites with relatively shallow deep peats it seems likely that some PEW habitats

would be established across the boundaries between deep and shallow peats (which can be complexly interlocked). Although if established on shallow peat these woodlands would not by definition technically be PEW these habitats could otherwise be identical except that, in very general terms, the shallower the peat becomes the more vigorous the tree growth would likely be. Shallow peat areas could therefore potentially be important parts of some PEWs.

1.5. Impacts of the peatland forestry management options

The main focus of the peatland forestry guidelines is in terms of the effects the options will have on greenhouse gas balance and this will be discussed in the subsequent sections. However there are a wide range of other considerations in determining the future management of afforested peatland sites, some of which are acknowledged in the guidelines (biodiversity, landscape aesthetics, hydrological impacts and timber security), along with those highlighted by other land managers such as stability and sustainability (RSPB, 2014). This section explores the impacts the three management options might have on these other considerations.

1.5.1. Biodiversity

Although peatlands have relatively low species richness, they support a specialised flora and fauna. Forest-to-bog restoration does facilitate the recovery of this diversity with some taxonomic groups being able to recover quickly (Pravia et al., 2020) but recovery times can be substantial and in many cases longer than any existing restoration project (Creevy et al., 2017; Hancock et al., 2018). Forest-to-bog restoration also removes a number of edge effects associated with plantations, for example declines in breeding wader population density (Calladine et al., 2014; Wilson et al., 2014) and tree expansion onto adjacent open bog (Manzano, 2012). In contrast conventional restocking would perpetuate edge effects on surround habitats. There is debate over the biodiversity values of non-native conifer plantations with conventional wisdom indicating that they have limited value but some research highlights that well established non-native plantations can provide habitats for varied taxa (Quine and Humphrey, 2010)

The biodiversity value of PEW is hard to predict and would strongly depend on the composition and structure of the woodland. Peatland Edge Woodland could potentially

perpetuate some of the damaging edge effects similar to commercial forestry such as providing shelter for predators of wader species. It is unclear to what extent specialised open bog flora and fauna could exist in a PEW. Peatland Edge Woodland could potentially be created to resemble low density forest edge or fringe woodland habitats which are a very rare habitat which support a number of rare plant and animal species such as juniper, downy willow, argent and sable moth, black grouse, nightjars and pine marten (Pickett, 2004; Galloway and Southern Ayrshire Biosphere, 2015; Forestry and Land Scotland, 2020).

1.5.2. Impact on surrounding hydrology

Commercial forestry plantations can have a drying effect on adjacent peatland (Shotbolt et al., 1998). Peatland Edge Woodland should not have the same density of trees or the same intensity of drainage (Forestry Commission Scotland, 2015) so it would be logical to assume it would not cause as extreme an edge effect. However forest-to-bog restoration would probably reduce the edge effect most since this strategy is most focused on rewetting the site.

Afforestation of peatland and forest-to-bog restoration can affect the water quality and flood risk of down-stream water courses. Evidence of the impact of the three peatland management options on flooding is limited and can be contradictory as there is a complex interaction between different factors (Anderson, 2001; Allott et al., 2019) with drain blocking inhibiting water loss from a peatland but in contrast drained peat has a greater capacity to store water in dry pore space (Van Seters and Price, 2001) and woodland cover can also decrease peak flow rate. Howson et al. (2021) find that at a forest-to-bog restoration site on blanket bog does not have an increased peak flow rate but a raised bog site does, the extent of this effect reduces with time since restoration. Peatland forestry is associated with declines in water quality such as water acidification (Helliwell et al., 2014). Forest-to-bog restoration can improve downstream water quality at least in the long term (Gaffney et al., 2018) while restocking would logically be likely to prolong declined water quality. The lower density of PEW may mean any negative effects of PEW would be reduced, also PEW may consist predominantly of broadleaf species which are not typically associated with acidification.

1.5.3. Aesthetic and landscape impacts

Woodlands, even remote conifer plantations, exist in a social context and may be used recreationally. The felling of a forest and its conversion to open bog or PEW is an obvious change in a landscape and community opinions need to be considered. Even isolated sites where recreational use is limited have cultural issues associated with them as there is still a social interest in the state of the Scottish landscape in general even if a majority do not physically see it (Martin-Ortega et al., 2017). However, it can be difficult to predict or assess what the most culturally sensitive management options are. Opinions are sometimes split over how the Scottish landscape should be managed, leading to disputes between those advocating the protection of open habitats and those advocating woodland expansion (Keane, 2017; Mountaineering Scotland, 2017; Mountaineering Scotland and The Scottish Gamekeepers Association, 2017).

1.5.4. Timber provision and employment

Woodlands are important as a source of timber. The UK's demand for timber is much greater than its production; in 2019 it was the second biggest net importer of timber products globally (Forest Research, 2021). Global demand for timber is predicted to as much as triple by 2050, making this level of importation a concern (World Wildlife Fund, 2012). In recent years the Scottish Government has set a series of ambitious woodland expansion targets such as increasing Scotland's forest cover to 21% by 2032 (Scottish Government, 2019). Peatland restoration actively decreases the forest area in Scotland and so runs counter to these targets and reduces Scotland's potential to produce timber. However, there is some debate over the point at which it becomes economically justifiable to restock deep peat sites; the guidelines' minimum requirement for general yield class 8 for Sitka spruce is a relatively low yield so these plantations may not make a substantial contribution to timber security. Peatland Edge Woodland will not generate as high a yields as conventional restocking and it is unclear from the guidelines to what extent harvesting timber from PEW might be incorporated into its management, for example firewood. Utilising economic opportunities from woodland primarily intended for its conservation value features in modern policy advice (Woodland Expansion Advisory Group, 2012).

A traditional argument for commercial forestry is its role in creating rural employment, often in areas that are especially deprived. In 2015 there were an estimated 19,555 people

employed in jobs attributable to Scottish forestry and timber processing (CJC Consulting et al., 2015). There is also employment associated with peatland restoration and PEW creation and monitoring although the author is unaware of any work quantifying the relative employment benefits of these options.

1.5.5. Stability and sustainability

At an early stage of the research the author came across many people with a professional interest in afforested peatlands who have concerns about the sustainability of one or more of the three future afforested peatland management options.

Forest-to-bog restoration aims to restore natural processes that will maintain the habitat – this would make it a sustainable option. Experiments and trials have shown that given enough time and investment many peatlands can be set on a trajectory towards a near-natural open state (Hermans et al., 2019). Peatland restoration practices are advancing and there are now sustainable solutions for some previously problematic features for peatland restoration, such as peat cracking (Anderson, 2016).

Conventional restocking of a site keeps the site under intensive human management and active decisions will have to continue to be made as to what the best use for the site is at the end of each rotation. It is possible that the longer a site is used for forestry, the more difficult forest-to-bog restoration will become. Depending on the management of a site it is also possible that the productivity of a site will decline through successive rotations but research has shown that this is not necessarily an unavoidable consequence of forestry and can usually be avoided through good management (Evans, 2000; Lim et al., 2020; Garrett et al., 2021).

The sustainability and stability of PEW is very debatable. Various stakeholders have raised concerns that excessive tree regeneration and growth will convert the site to closed canopy woodland, and some are concerned that tree death rate will exceed regeneration, resulting in recovery of open bog. Naturally occurring bog woodland exists in an equilibrium (Anderson and Harding, 2002; Barsoum et al., 2005). Woodland establishing on degraded peatland sites often results in the formation of closed canopy woodland associated with the loss of the original bog habitat through the drying and shading out the original vegetation (Lindsay et al., 1988). In this context a positive feedback can exist whereby trees progressively dry out the bog and so facilitate further invasion (Limpens et al., 2014). Stable bog woodlands might be sustained by critical

thresholds of disturbance determining whether tree growth on bogs results in a stable open canopy woodland, succession to closed woodland or reversion to open bog (Anderson and Harding, 2002; Eppinga et al., 2009). Bog woodlands can have highly variable stem densities varying between 400-5000 stems per hectare; even at high densities an open canopy can be maintained by smaller stunted trees. For comparison the closed canopy of a commercial forestry plantation typically has 2500 trees per hectare (Anderson and Harding, 2002). As such, tree growth rate rather than tree density may be more important in determining whether a bog woodland is at risk of becoming a closed canopy woodland. It is very unclear how much ongoing management an area of PEW might require or whether there is potential for PEW to develop a stable equilibrium between closed canopy woodland and open bog.

Climate change may affect the future resilience of the management options and is an important consideration when considering how appropriate the different management options are and what the likely outcomes of each option maybe. Scotland is predicted to have warmer annual temperatures with drier summers under projected climate change conditions (Werritty and Sugden, 2012). These changes are predicted to result in greater carbon loss from peatland (Ferretto et al., 2019, 2021). Changing climatic conditions are making many areas less suitable for open peatlands; instead areas which were previously stable peatlands may shift to being forest as an alternative stable state, and as a result forest-to-bog restoration may become more challenging (van der Velde et al., 2021). Exactly how resilient peatlands may be to climate change is unclear (Page and Baird, 2016), however peatlands in good condition may be more resilient to climate change than those in poor condition (e.g. those with artificial drainage) (Bain et al., 2011).

Increased fire frequency is a major concern in relation to climate change. Restored peatlands have, in some situations, been shown to recover resistance and resilience to perturbation such as wild fire (Blier-Langdeau et al., 2021). The PEW guidance indicates that PEW sites may not be as well restored as forest-to-bog restoration areas (e.g. the drainage may not be fully remediated) and so PEW sites may be more susceptible to climate change related degradation such as wild fires. A study in Canada identified that areas which had been more intensively drained and had subsequently experienced an increase in tree growth were significantly more badly burned (i.e. peat burned to a greater depth) (Wilkinson et al., 2018).

Trees growing on Scottish peatlands as part of PEW would generally be expected to grow more vigorously under predicted future climate conditions (Yu et al., 2021). This might increase carbon sequestration in the trees, but might also mean that even if stable PEW habitats could be created under current conditions, they may become unstable due to climate change and be more

likely to become dense closed canopy woodlands. Conventionally restocked sites may grow better under new climate conditions (ClimateXChange, 2016), potentially making restocking sites more desirable, although the sites may then become harder to restore to open habitats should it be attempted in the future.

1.6. Mechanics of greenhouse gas flux from peatland

The three future management options – PEW, Conventional Restocking and Forest-to-Bog Restoration – are all described primarily in terms of their impact on climate forcing. Peatlands are a significant component in regulating the earth's climate through being sources or sinks for major greenhouse gases; carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) (Leifeld and Menichetti, 2018). Different greenhouse gases have different potentials to cause global warming, and different metrics have been created to quantify a comparison between different gases. The most commonly used of these is Global Warming Potential (GWP) which considers the lifetime of a gas in the atmosphere with its radiative characteristics to give a comparative figure in CO₂ equivalents (per molecule/per mole) (Lashof and Ahuja, 1990; Rodhe, 1990). GWP is calculated for specific time periods (e.g. 100 years); gases with a longer lifetime will have higher GWP if considered over a longer time period and there is some debate over what is the most relevant timescale to calculate this over (Sarofim and Giordano, 2018). Some researchers highlight the weaknesses of the GWP approach and have proposed more complex measures to compare the importance of different greenhouse gases (Neubauer and Megonigal, 2015). Different approaches to comparing the warming effect of greenhouse gases will have different strengths and weaknesses and this should be considered when using one (IPCC, 2013).

In this and the following three sections different aspects of climate forcing and peatlands are reviewed. This section (1.6) reviews the natural processes that underpin greenhouse gas fluxes from peatlands. The next section (1.7) covers methods to study the climate forcing potential of peatlands. The subsequent section (1.8) reviews how the three future management options for deep peat forestry sites affect the climate forcing potential of a site.

1.6.1. Mechanics of greenhouse gas fluxes from peatland surface

Carbon Dioxide (CO₂)

Peatlands in good condition are generally weak carbon sinks (Drewer et al., 2010; Bain et al., 2011). As plants on peatland photosynthesise, they take up CO₂ and although peatland vegetation has relatively low primary productivity, the low rates of decomposition can result in carbon net accumulation. However, there is some debate over how effective contemporary peatlands are at accumulating peat, some peatlands may have reached a natural equilibrium with equal rates of decomposition and accumulation – this is supported by theory of peatland formation and growth (Clymo, 1984).

The slow rate of decomposition on peatland in good condition is due to the waterlogged, acidic and anoxic conditions found close to the peat surface (Limpens et al., 2008). Decay is inhibited further by the phenolic compounds and uronic acids produced by *Sphagnum* (Verhoeven and Liefveld, 1997). Carbon dioxide will be released from the oxidative decay of plant tissue above the ground surface or in the acrotelm, but increasingly anoxic conditions will inhibit this as plant material becomes more deeply buried. When peatlands are degraded, for example by draining, water table depths can be lowered, resulting in a greater rate of oxidative decay. If this rate exceeds primary productivity then the peat will be a net source of CO₂ (Limpens et al., 2008).

Methane (CH₄)

In contrast to CO₂, anoxic conditions facilitate the production of CH₄. Per molecule CH₄ has a greater GWP than CO₂ – 28-34 times greater over a 100 year timescale (Myhre et al., 2013). CH₄ though, is typically released at a much lower rate than CO₂ and has a much shorter atmospheric lifetime (Myhre et al., 2013). There is much debate about what timescale the warming potential of CH₄ should be considered over, with many arguing that in the case of peatland a timescale of 500 years is more relevant (Artz et al., 2012b). At longer timescale the warming potential of CH₄ would be less. This debate influences the relative importance of CH₄ when assessing the climate forcing impact of different management options.

CH₄ is produced in the anoxic catotelm by methanogenesis, a metabolic pathway which has evolved a specialist group of archaea bacteria known as methanogens. In aerobic conditions a taxonomically diverse group of bacteria can break down CH₄ by methanotrophy. These bacteria are endosymbiotic with *Sphagnum spp.* with the CO₂ released by the bacteria promoting

photosynthesis in the *Sphagnum* (Raghoebarsing et al., 2005). *Sphagnum* removal from the peat surface has been found to cause a 5 fold increase in CH₄ emissions (Kip et al., 2010).

CH₄ can be released into the atmosphere by several mechanisms including; molecular diffusion from point of production upward through the water, ebullition where CH₄ rises quickly up from the catotelm through the acrotelm in bubbles and via conduits formed in the tissues of vascular plants such as the aerenchyma of many Cyperaceae (sedge) species, e.g. *Eriophorum vaginatum* (Frenzel and Rudolph, 1998; Marinier et al., 2004; Lai, 2009). Ebullition and plant mediated transport allow the rapid release of CH₄, decreasing the potential for it to be broken down before it is released (Frenzel and Rudolph, 1998).

Nitrous oxide (N₂O)

Per molecule, N₂O has a greater GWP than either CO₂ or CH₄ at between 265-298 CO₂ equivalents over 100 years (Myhre et al., 2013). Ombrotrophic peatlands in good condition are usually only weak sources or even weak sinks for N₂O. In Scotland, N₂O emissions are usually considered to be insignificant if the area has not had a history of nitrogen fertiliser application and not in the central belt where significant levels of atmospheric nitrogen deposition may have occurred (Artz et al., 2012b). N₂O fluxes are less studied and less predictable so there is more uncertainty around its relative importance (Worrall et al., 2010; Liimatainen et al., 2018).

N₂O fluxes are underpinned by various and not necessarily understood mechanisms (Butterbach-Bahl et al., 2013). In peatlands N₂O can be produced anaerobically by denitrification (respiration with NO₃⁻ as electron acceptor) or Dissimilatory Nitrate Reduction to Ammonium (DNRA) and aerobically by nitrification (oxidation of NH₄⁺ to NO₃⁻) (Machacova et al., 2013). Denitrification can also convert N₂O in the atmosphere to N₂ which may drive the weak N₂O sink dynamics of some peatlands (Hatano, 2016). Nitrogen cycle dynamics mean peatlands that have been drained and/or had nitrogen fertiliser applied are usually stronger source of N₂O (Martikainen et al., 1995; Salm et al., 2012)

Summary of greenhouse gas dynamics

Under wetter conditions, anoxic conditions will occur closer to the peat surface, while under dryer conditions aerobic conditions will occur to a deeper depth. Given the conditions in which different greenhouse gases are produced, a dynamic exists whereby under wetter conditions peatlands will

either be a sink or weaker source for CO₂ and N₂O but a strong source for CH₄. In dryer conditions peatlands will likely be a stronger source for CO₂ and N₂O but a weaker sink of CH₄.

1.6.2 Mechanics of greenhouse gas fluxes from trees growing on peatland

Carbon dioxide (CO₂)

Photosynthesis fixes CO₂ during tree primary production. This will remain fixed in the tree for varying periods of time. Some of the carbon will be respired almost immediately (either by tree tissue or organisms feeding on it), producing CO₂ which may then be released back into the atmosphere or may be transported around the tree and fixed again. Other fixed carbon may be transported to other parts of the tree and/or converted to complex storage/structural molecules which may remain resident in the tree for years. Carbon fixed by a tree may also contribute to soil carbon, either through loss from the tree roots (e.g. exudates, root litter production and sloughing-off of root cells) or above ground (e.g. leaf/needle/twig/cone/budscale drop). Once part of the soil the carbon may be stored or released as described in the previous section. When the tree dies its biomass is partly incorporated into the soil and partly respired by decomposers. In commercial forests the tree may be harvested. In this case when the carbon stored in the harvested wood is released would depend on the usages of the wood products; certain products such as firewood or paper may be broken down in the short term while carbon in structural wood may remain fixed for 100s of years.

Trees can also act as conduits for CO₂, transporting it from below-ground, where it may be produced by oxidative decay of the peat, to the canopy where a proportion may be released into the atmosphere (Bloemen et al., 2013; Anné, 2014). This process can continue after the death of the tree (Carmichael et al., 2018).

Methane (CH₄)

Trees significantly contribute to ecosystem CH₄ flux. This significance has only recently been appreciated and so its relative importance to woodland CH₄ cycles remains uncertain (Yamulki, 2017). CH₄ fluxes direct from woodland soils generally make up the largest proportion of net ecosystem fluxes however trees have been found to contribute up to 18% of woodland emissions

(Machacova et al., 2016). On a global scale trees may account for as much as 10% of anaerobically derived CH₄ emissions (Rice et al., 2010).

There are four known mechanisms by which trees can affect CH₄ flux, all of which could theoretically apply to trees growing on peatlands:

1) Conduits for soil derived methane

Trees can act as conduits for soil derived CH₄ which can be taken up by the roots and transported via aerenchyma, if present, (Rusch and Rennenberg, 1998) and/or the xylem (Covey and Megonigal, 2019). After transport CH₄ can subsequently be released into the atmosphere via the stem surface, lenticels and leaf stomata. Methane flux through tree conduits is usually higher when soil conditions are wetter and consequently concentration of below ground CH₄ is higher (Machacova et al., 2016).

2) Microbial production of methane on or inside tree structures,

Methanogenic bacteria which live in or on plant material can be a significant source of CH₄ (Covey et al., 2012; Warner et al., 2016). Methanogenic bacteria can be active in/on living or dead woody material (Warner et al., 2016). Methanogens require anaerobic conditions such as those in rotting heartwood, consequently this mechanism is likely to be more important in older trees with greater proportions of deadwood (Covey et al., 2012). Fungal activity can also create anaerobic microsites in predominately aerobic environments (Reith et al., 2002).

3) Production of methane in tree tissues.

Keppler *et al.* (2006) demonstrated that CH₄ can be produced directly in the plant tissues through an unknown aerobic mechanism. A detailed understanding of this mechanism is still lacking but it appears to be related to elevated production of reactive oxygen species, which are produced in response to environmental stresses (Liu et al., 2015; Covey and Megonigal, 2019). Reactive oxygen species may react with plant compounds such as pectins and result in the release of CH₄ (Keppler et al., 2008; Liu et al., 2015). It is very uncertain if aerobic CH₄ production is a significant component of global CH₄ flux, with estimates varying by orders of magnitude (Keppler et al., 2006; Bloom et al., 2010).

4) Methane uptake by trees

Trees can also act as weak CH₄ sinks. This is likely due to endophytic or epiphytic methanotrophic bacteria (Sundqvist et al., 2012). Sundqvist *et al.* (2012) found that the rate was strongly related to

Photosynthetically Active Radiation (PAR); given that PAR is also related to stomatal conductance this suggests that the bacteria live inside the leaf.

Nitrous oxide (N₂O)

There is limited knowledge of N₂O fluxes from trees growing on peatlands. However, there are a number of studies on trees growing in non-peatland wetlands, some of these find trees to be a significant source of N₂O (Machacova et al., 2013) while others find them to contribute relatively little to the whole ecosystem flux (Schindler et al., 2020).

1.6.3. Carbon loss to watercourses

Carbon flux through watercourses can also be a significant route of carbon loss from peatlands (Roulet et al., 2007; Nilsson et al., 2008; Yupi et al., 2016). Carbon can be lost as dissolved gases (CH₄ and CO₂), as Dissolved inorganic matter (DIC - inorganic ions) and organic forms such as Dissolved Organic Carbon (DOC) and Particulate Organic Carbon (POC) (Meybeck, 1993; Dawson et al., 2002). The distinction between DOC and POC is that DOC will pass through a 0.45µm filter while POC will be retained on the filter (Bonnett et al., 2011).

Dissolved Organic Carbon is produced as organic matter decomposes – it accumulates in pore water and drains out into watercourses or is washed out by surface run-off (Limpens et al., 2008). Up to 55% of DOC lost to watercourses is converted to CO₂ either chemically by photoreactivity or biologically by bacteria (Pickard et al., 2017). Dissolved Organic Carbon can be a cause of the distinct colour of 'peaty' water but DOC has varied composition and not all components are coloured (Cory et al., 2014; Temnerud et al., 2014)

1.6.4 Geophysical effects

So far in this thesis radiative forcing has only been discussed in terms of greenhouse gases, the cycling of which is underpinned by biogeochemical processes. This is consistent with the guidelines on peatland forestry and the emphasis of research on land use management (Schwaiger and Bird, 2010). Radiative forcing can however also be affected by biophysical processes such as albedo (amount of radiation reflected from a surface), roughness and

evapotranspiration (which determines cloud cover – which have high albedo) (Pongratz et al., 2010). Forests have a lower albedo than open areas (causing a warming effect) especially in boreal regions which have seasonal snow cover but also increase evaporation which has a cooling effect (Myhre and Myhre, 2003; Bala et al., 2007; Pongratz et al., 2010). When this is considered together with biogeochemical processes, major modelling studies have found in general increased tree cover has a net cooling effect in equatorial regions, net warming effect in boreal regions and limited cooling effects in temperate regions (Bala et al., 2007; Pongratz et al., 2010).

1.7. Methods for studying greenhouse gas emissions

1.7.1. Fluxes from peat

Greenhouse gas fluxes can be measured at a small scale using chamber systems where specific areas or plants are enclosed with a chamber, or at the ecosystem level, with an eddy covariance system which uses vertical wind velocity and CO₂ mixing ratios (Baldocchi, 2003). This thesis focuses on chamber methods as eddy covariance systems operate at a spatial scale too large to be viable for use on the field sites identified as suitable for the data collection. By using a chamber it is also possible to compare small-scale plots or features. Additionally, eddy-covariance systems are more expensive and there was no system available to this thesis.

There are several different chamber systems for measuring gas fluxes. These are broadly split into three categories 1) closed static chamber 2) closed dynamic chamber (non-steady state through-flow chamber) 3) open dynamic chamber (steady-state through-flow chamber) (Pumpanen et al., 2004). Open dynamic chambers allow a constant flow of ambient air through the chamber at a known rate and gas concentration, with measurements taken at the chamber outlet. Closed chambers are isolated from external air, with gas concentration measurements taken direct from the headspace of the chamber. Closed Static Chambers (CSC) involve manual sampling of gases from the headspace of a chamber with syringes over a time series, while closed dynamic chambers cycle air in the headspace of the chamber to a gas analyser which can automatically take concentration measurements up to every second.

Each chamber system has limitations. Closed systems can exhibit asymptotic fluxes where rises or falls in chamber headspace gas concentrations disrupt diffusion gradients inhibiting further fluxes out of the soil or vegetation into the headspace (Davidson et al., 2002). Closed

dynamic chambers are generally considered superior to closed static chambers if a portable gas analyser is available and suitable for use in the experimental area. This is because closed static chambers provide instantly available data while measurements are being taken, allowing unreliable measurements to be identified and retaken. Additionally the more frequent gas concentration measurements of dynamic chambers allow flux measurements to be taken over shorter time periods, reducing the likelihood that measurements will be affected asymptotic fluxes. (Heinemeyer and McNamara, 2011). Open dynamic chambers can develop pressure differences with ambient air resulting in mass transport of CO₂ out of the soil, however the risk of asymptotic fluxes is reduced, as ambient air is continuously circulated into the chamber. This makes them more suitable if chambers are going to be left closed for hours or days (Lund et al., 1999). Data for this thesis were collected with closed dynamic chambers since portable gas analysers were available and usable on the site and flux measurements were desired over short time windows. This is also the dominant approach in similar studies (e.g. Hermans et al., 2019; Mazzola et al., 2022) making the results more easily comparable.

1.7.2. Fluxes from trees

Eddy-covariance systems have been a useful way to study gas fluxes from forest ecosystems for decades (e.g. Grace et al., 1996; Yu et al., 2006; Rebane et al., 2019). However, eddy covariance measurements include the contributions of all the ecosystem components in an area – for example from soils – so the direct contributions from the trees cannot be so well understood with an eddy covariance approach.

Measuring gas fluxes directly from specific tree components is a relatively recent experimental field which is still actively being developed. Although the fundamental approaches are the same as from ground collars the collars and chambers needed to enclose all or part of a tree need to be very specialised. New designs of chambers are frequently published as more workers attempt to take measurements from a variety of trees for a variety of objectives. Chambers are most commonly attached to tree stems (e.g. Jeffrey et al., 2020; Pangala et al., 2015) but also branches (e.g. Sundqvist et al., 2012; Hakola et al., 2006) and shoots (e.g. Wallin, Skarby and Sellden, 1990). Chambers developed include rigid (Pangala et al., 2013, 2015; Jeffrey et al., 2020) and semi-rigid designs (Siegenthaler et al., 2016). Different diameters and textures of trees have necessitated variations in attachment methods, with both permanently attached (Warner et al., 2016) and removable collars (Pangala et al., 2015; Siegenthaler et al., 2016; Jeffrey

et al., 2020). Most studies have used manual chambers but there are also studies which have made use of automated systems that automatically take measurements at regular intervals (Sundqvist et al., 2012; Covey and Megonigal, 2019).

Large flux chambers have been developed with dimensions of ~1-2 metres to enclose macrophytes but these are usually herbaceous (Pangala et al., 2017) or shrubs, they have never been used to enclose entire small individuals of a tree species.

1.7.3. Carbon stock

Flux chamber techniques can be used to measure contemporary fluxes of CO₂, CH₄ and N₂O from trees and soil. However estimates of net CO₂ uptake or loss in forests and soils over long time periods can be measured through destructive biomass methods (Fearnside et al., 1993). The percentage carbon of mass in plant material or soil can be estimated through a range of lab techniques including precise analytical carbon content analysers and loss on ignition.

For peat, samples can be collected with an auger (such as a Russian corer). Samples can either be taken from the whole peat column or subsampled at representative depths (Kauffman and Donato, 2012; Chimner et al., 2014). To calculate the carbon content of a sample, its volume and dry weight must first be calculated to give a mass per unit volume of peat (dry bulk density). Bulk density can be combined with an estimate of the percentage carbon in a sample (e.g. by mass spectrometry) to give a carbon content per unit volume of peat (i.e. volumetric carbon density). Combined with the established depth of an area of a peatland this can be used to calculate the whole carbon stock.

For trees, whole trees can be felled, split into categories (stem, branches and leaves), weighed, dried and re-weighed to calculate the proportion of wet to dry mass. The percentage of carbon per dry mass can be calculated using the measured carbon concentration and this extrapolated back to the whole tree. The results of this can be generalised over larger areas by taking sample plot measurements such as DBH, height and mass and building allometric equations for the stand from all the trees (Picard et al., 2012; Velasco and Chen, 2019). More recently remote sensing techniques such as LiDAR have been applied in place of manual measurements (Hao et al., 2019).

1.7.4. Dissolved Organic Carbon

Dissolved Organic Carbon can be measured from water samples collected from dipwells, watercourses originating from peatlands or directly from peat soil pores through rhizon samplers (Bonnett et al., 2011). Dissolved Organic Carbon samples are conventionally defined as organic carbon that can pass through a 0.45µm filter with larger particles of organic carbon classed a POC. Inorganic carbon can be removed from filtered samples by acidifying the sample and purging it with nitrogen, the resulting sample should contain only non-purgeable organic carbon (NPOC). Carbon content analysers can then be used to quantify NPOC concentration which should equate to DOC concentration if the samples have been filtered.

1.8. Impact of afforested deep peat management options on climate forcing

1.8.1. Introduction

This section reviews the impact of forest-to-bog restoration, restocking and PEW on the climate forcing effects of afforested deep peat. The Forestry Commission Scotland Practice Guide argues that, correctly located, each management option has the potential to provide the best climate forcing benefits, however this has been disputed. This section explores current understanding of the relative impacts that the three strategies have on climate forcing with the uncertainties that exist. The overall climate forcing effect is complex to estimate and is determined by the combined effects of the net gas fluxes in the area, carbon losses to connected watercourses and geophysical effects, see figure 1.12. Peatlands are globally very variable, and therefore research pertaining to peatlands cannot be considered globally applicable. For example peatland forestry is relatively well researched in Fennoscandinavia, but the context of this forestry is very different to Scottish forestry, typically being characterised by the draining of naturally wooded peatlands to increase productivity of existing trees (Lohila et al., 2011). It is important to be aware of the context in which research was done: this section focuses on studies carried out in Britain and Ireland.

There is a high degree of scientific uncertainty about the effects of different management options on radiative forcing. There is a complete absence of research on PEW, so the effects of this management option are necessarily more speculative and based on inferences from related research. Second rotation peatland forestry and forest-to-bog restoration are also relatively

recent management approaches which again limits scientific enquiry into their long-term effects. The development of more advanced restocking and forest-to-bog restoration techniques such as ground smoothing are even more recent and so little is known about the relative effectiveness of different restoration methods (Hermans et al., 2019). The existence of multiple approaches to both restocking and forest-to-bog restoration and the large variation between the physical properties and management histories of different peatland areas causes difficulty for generalising results of existing studies. Equally, although there is research based on the impacts of first rotation forestry plantations there has been little opportunity to study the impacts of restocking existing plantations with lower-intervention modern methods (Morison et al., 2010). There is a clear need for more and longer-term studies monitoring the impacts of the management options (Andersen et al., 2017).

The Intergovernmental Panel on Climate Change (IPCC) has developed a system of national greenhouse gas inventories whereby the emissions of a nation including those from Land Use, Land-Use Change and Forestry (LULUCF) are estimated (Penman et al., 2006; Hiraishi et al., 2013; Buendia et al., 2019). National greenhouse gas inventories can be calculated according to 1-3 tiers with tier 1 considered the least accurate. Tier 1 uses very generalised emission factors but is also the cheapest/simplest method, while tier 3 is the most complex but should give more accurate results. A recent project to make tier 2 estimations of greenhouse gas emissions from peatlands identified a particular need for more field flux measurements in afforested peatlands to enhance the accuracy of future tier 2 and 3 estimates (Evans et al., 2017). The following sections describe the current state of knowledge of the effect of afforested peatlands on different components of climate forcing.

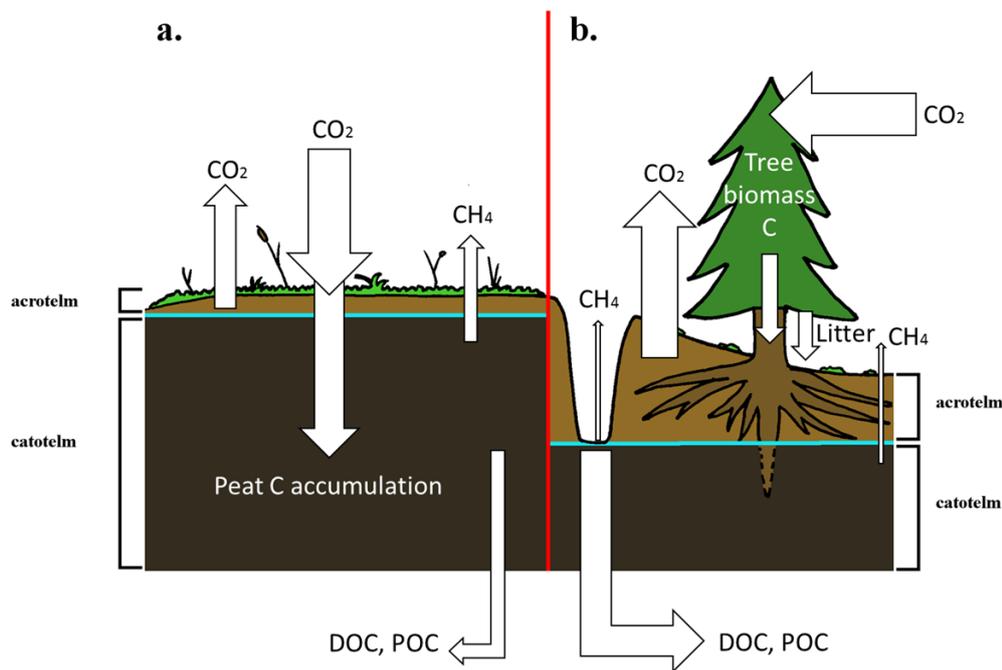


Figure 1.12: Illustration of radiative forcing effects comparing open peatland with afforested peatland, taken from Sloan et al. (2018).

1.8.2. Effect on carbon dioxide

Restocking

Determining CO₂ balance of an afforested peatland is complex. Afforested drained peatlands maintain a lower water table than areas restored to open bog (King et al., 1986) and therefore CO₂ release by oxidative decay of peat is expected to be higher under restocking (Clymo, 1983). However, carbon is transferred to the soil by the planted trees via root growth and needle drop (Jandl et al., 2007) which can increase below ground carbon storage. The primary productivity of ground flora – and therefore its capacity to sequester CO₂ will be reduced by using conventional restocking, however, growing trees are likely to sequester substantially larger amounts of CO₂ compared to the ground flora of a restored open bog.

Many studies on first rotation forestry plantations in the UK and Ireland suggest that, over the lifecycle of a deep peat forestry plantation, oxidative decay of the peat exceeds forestry-

related carbon inputs into the soil such as needle litter and root exudates (Cannell et al., 1993; Hargreaves et al., 2003; Lindsay, 2010; Morison et al., 2010; Chapman et al., 2013; Sloan et al., 2018a; Sloan, 2019; Jovani-Sancho et al., 2021). However some studies find no evidence for significant carbon net loss from afforested deep peatland soil over the life cycle of a forest (Anderson et al., 1992; Byrne and Farrell, 2005), and there is evidence that the soil of a mature forest on deep peat is a net sink for CO₂ (Hermans et al., 2022). Studies that have accounted for carbon sequestration in the timber have variously found – using carbon stock methods – evidence that this only partially offsets carbon losses from the peat (Sloan, 2019), and – using eddy covariance methods – evidence that it exceeds carbon losses from the peat, resulting in net carbon sequestration (Hargreaves et al., 2003). Some modelling work shows that trees growing on peatlands may or may not completely offset losses from the soil depending on how productive the plantations are (Morison et al., 2010; Worrall et al., 2011). However, others have questioned the validity of models which find forestry plantations to be net sinks over the course of a rotation (IUCN, 2014). Conversely, different modelling work has found that UK afforested peatlands are a substantial carbon source with net emissions of 4600 kt CO₂e yr⁻¹ (Evans et al., 2017).

A large portion of the carbon losses from peatlands are associated with the initial ground preparation and tree planting due to the disturbance these activities cause. Therefore, establishing a second rotation of forestry on deep peat already in a disturbed state is expected to result in smaller carbon losses than the first rotation, especially since trees would then be planted with lower disturbance methods. For shallow peat (<50 cm) carbon losses from the peat during the second rotation may be lowered to the extent that inputs into the upper soil layer from the trees may eventually compensate for carbon losses from the peat during both rotations (Vanguelova et al., 2019). In deep peat where carbon losses from the soil are typically higher, the inputs from the trees into the soil are less likely to completely compensate for peat carbon losses. However, when carbon sequestration in trees is accounted for, good tree growth may compensate for CO₂ losses from the peat during a second rotation. As previously discussed, the recent forestry guidance predicts that a general yield class of 8 for Sitka spruce will typically be sufficient to compensate for peat carbon losses (Morison et al., 2010; Forestry Commission Scotland, 2015; Vanguelova et al., 2018).

There is also debate about the degree to which carbon storage in deep peat, forest topsoils and trees should be seen as equivalent: carbon in peat is potentially much more stable than carbon stored in trees. Carbon stored in peat may have been stored for thousands of years and might have the potential to be stored for thousands more, whereas carbon in tree biomass, litter and wood products, depending on their use, may have a much shorter life span. Prolonged

forestry at a site will maintain a relatively constant rate of CO₂ loss from the peat carbon store but CO₂ losses from the decay of wood products derived from trees grown on the site would increase over time (Hargreaves et al., 2003). After successive rotations the combined peat and wood product CO₂ losses would exceed the CO₂ sequestered in the growth of new trees (Hargreaves et al., 2003). This leads some to conclude that restocking plantations on deep peat is unsustainable in the long term (Hermans et al., 2019).

The UK is the second largest net importer of forest products in the world (Forest Research, 2021); a reduction in productive forestry area in the UK will cause increasing reliance on imports. To fully understand the greenhouse gas balance of not restocking a plantation, there needs to be consideration of whence the foregone forest products will be supplied and what their greenhouse gas balance will be, taking into account transport into the UK. This displacement of an activity due to an environmental policy intervention is known as 'leakage' and is a complication in assessing the impact of environmental policy (Bastos Lima et al., 2019). Further complication arises in calculating carbon balance if increased timber production could replace materials that require the use of fossil fuels in their production. However, since the guidelines propose that PEW should be considered for sites deemed unsuitable for restocking, i.e. sites which generate low yields, PEW would arguably not be substantially reducing timber production.

The complexities of land management and supply chains make it difficult to assess the long-term net flux of CO₂ from afforested peatlands.

Forest-to-bog restoration

Successful forest-to-bog restoration does result in an often rapid rise in the water table depth (Worrall et al., 2007; Anderson, 2010). This should theoretically reduce oxidative losses of CO₂ and in the long term facilitate the recovery of vegetation such as *Sphagnum* that helps prevent decay. Field studies have shown that forest-to-bog restoration typically results in an initial short term increases in CO₂ emission associated with the disturbance of restoration work and the influx of dead wood that is left after tree felling such as tree roots which Hermans et al. (2019) estimates contributes up to 27% of CO₂ emissions from the peat in the first year of forest-to-bog restoration. After this initial rise CO₂ fluxes tend toward near natural levels and can become net CO₂ sinks again (Hambley et al., 2018; Hermans, 2018; Creevy et al., 2020). The aforementioned studies are all based on older, lower intensity restoration methods where trees are felled and partially or entirely removed, and drains are blocked. Newer, higher intensity methods such as

ground smoothing and mulching are less well understood, but could potentially result in faster recovery of a net greenhouse gas sink.

Peatland Edge Woodland

Peatland Edge Woodland would be expected to have intermediate properties to forest-to-bog restoration and restocking. The lower intensity of management called for by a PEW is unlikely to facilitate growth of trees which sequester carbon at a rate comparable to that of a conventional forestry plantation. Without intensive efforts to rewet, oxidative losses from the peat in a PEW would be expected to be higher than open bog. It is unclear though what net effect this will have. The author knows of two flux studies carried out in the UK on habitats that in some ways resemble the description of PEW (i.e. native woodland planted in peaty soils) (Friggens et al., 2020; Mazzola et al., 2022). However, neither study is based on a habitat which is strictly PEW as both studies look at habitats on sites which have not been historically commercially afforested, and Friggens et al. (2020) is based shallow peat. Mazzola et al. (2022) found that CO₂ fluxes from the soil surface of deep peat colonised by Scots pine regeneration were higher in close proximity to Scots pine trees growing in a bog edge woodland, this was attributed to CO₂ fluxes from tree roots, but also to increased fluxes from the decay of the peat itself. Friggens et al. (2020) found that Scots pine and downy birch planted on shallow peat heather moorland increased the rate of respiration in the soil, releasing CO₂. This loss was not compensated for in the growth of tree biomass.

1.8.3. Effect on methane

Peatlands under forestry usually have reduced CH₄ emissions from the peat surface when compared to near-natural peatlands, which has been attributed to the lower water table (Sloan et al., 2018b). Forest-to-bog restoration results in an increase in CH₄ emissions relative to the originally present forestry plantations (Hermans, 2018; Creevy et al., 2020). Creevy et al. (2020) found evidence that CH₄ emissions fell between a restoration site that was 6 years old and 17 years old. Conversely (Hermans, 2018) found a long term increase in CH₄ flux with restoration age trending towards near natural CH₄ emissions. Short term localised very large increases in CH₄ emissions have been recorded in the first year of a restoration project (Hermans, 2018).

Peatland Edge Woodland is likely to have a water table lower than near-natural peatlands and as such CH₄ emissions lower than those of a near-natural peatland would be expected from the peat surface (Järveoja et al., 2016). Mazzola et al. (2022) found that CH₄ emissions were lower close to Scots pine trees growing on deep peat and higher further away.

Forest-to-bog restoration also allows recovery of the peatland vegetation which will often increase CH₄ emissions further as *Eriophorum species* become more abundant (Kettunen, 2003). It is unclear how PEW would affect the recovery of *Eriophorum species*, but it is plausible that if PEW creates a dryer environment this may inhibit *Eriophorum species* and therefore limit the associated CH₄ emission.

An additional complication is the contribution the trees might make to CH₄ flux. There haven't been any studies of CH₄ emissions from trees under peatland forestry management. If the water table is very low then the tree might not be a significant conduit for CH₄ release if the trees are shallow rooted but lodgepole pine is capable of rooting into waterlogged layer (Coutts and Philipson, 1978) so may be able to transport CH₄ produced at depth in the anaerobic zone, Sitka spruce typically roots shallower so potentially may not be such an important conduit for CH₄, however in the absence of scientific data this is speculative. On mineral soils Sitka spruce has been recorded taking up CH₄ through an unidentified mechanism, which is speculated to be epiphytic or endophytic methanotrophic bacteria associated with the tree branches (Sundqvist et al., 2012), suggesting that Sitka spruce in commercial peatland plantations could also plausibly be a CH₄ sink. Peatland Edge Woodland may be wetter than conventional plantations and therefore the trees are more likely to act as conduits for CH₄ produced underground. Downy birch and Scots pine are both species that may form PEW and both are known to be sources of CH₄ in wet soils (Pangala et al., 2015; Machacova et al., 2016), which could increase the overall CH₄ budget of PEW.

1.8.4. Effect on nitrous oxide

The data for N₂O fluxes from afforested peatland and forest-to-bog restoration projects is variable. Artificial drainage has been associated with increased N₂O emissions (Salm et al., 2012). However clear felling peatland forestry has also been associated with increased N₂O emissions, concurrent with increased water table (Huttunen et al., 2003). Hermans (2018) found that N₂O did emissions did not vary significantly between different aged forest-to-bog restoration areas or

open and afforested peatland. On a peaty gleys soil felling of a Sitka spruce plantation increased N₂O emissions for at least 4 years after felling (Yamulki et al., 2021). Given this uncertainty it is difficult to make any general predictions about the effect of the three management options on N₂O flux.

1.8.5. Effect on Dissolved Organic Carbon

Studies on the effect of open-bog restoration on DOC production and flux from peatlands are mixed. In the UK a number of studies on the effects of forest-to-bog restoration show that in both raised and blanket bogs DOC concentrations in surface and pore water rise immediately after restoration work with longer studies finding that over a number of years DOC concentration falls, tending to near natural concentrations (Muller et al., 2015; Gaffney et al., 2018; Shah and Nisbet, 2019; Howson et al., 2021a) Gaffney et al. (2018) studies a chrono-sequence of similar blanket bog sites where forest-to-bog restoration work involved felling the trees either to waste or felling and extracting the main stems and drain blocking. Howson et al. (2021a) included both blanket and raised bog plots with a range of management histories; none of the sites had been reprofiled but some had drains blocked, drains and furrows blocked or neither blocked. A range of tree felling techniques had also been used, including whole tree removal, conventional harvesting where brash was left, trees felled-to-waste leaving all trees on site compressed into the furrows, and whole tree mulching. Of these approaches whole tree mulching elevated DOC the most.

1.8.6. Geophysical effects

The author is only aware of one study which has considered geophysical effects in the specific case of afforested peatlands was based in Finland and found that forestry albedo effect had a net warming effect in comparison to natural areas (Lohila et al., 2010). They do not account for evapotranspiration difference although they reason that the drainage associated with forestry would have reduced evapotranspiration (Lohila et al., 2010). It is unclear what the net geophysical effects of peatland management in Scotland might be and investigating them is outside the scope of this thesis.

1.8.7. Conclusion

The combination of lower CH₄ emissions and carbon sequestration in tree growth associated with conventional restocking potentially makes restocking some existing productive plantations on deep peatland carbon-neutral or result in net carbon sequestration in the short to medium term. Conversely, forest-to-bog restoration may result in an increase in net greenhouse gas emissions in the short to medium term due to raised CH₄ emissions and a decrease in the rate of carbon sequestration in vegetation. However, forest-to-bog restoration may have the best climate change mitigation potential in the long term as it prioritises protecting long term carbon storage in peat, in contrast to restocking where there may be net loss of carbon from the below-ground carbon stores, with the deficit substituted for with carbon stored in forest products which are likely to be less stable.

Climate change will have an impact on the climate forcing effects of the management options. The effects of climate change, including raised CO₂ concentrations, are predicted to in general increase the Net Primary Productivity (NPP) of forests in Scotland (Yu et al., 2021). This would potentially make the trees in restocked peatlands and trees in PEW a larger carbon sink. However projected climate change is likely to increase the risk of carbon loss from large areas of Scottish peatland (Ferretto et al., 2019) creating further uncertainty over the climate impacts of the different management options.

There is a great deal of uncertainty about how PEW compares to the two more established management options. The guidelines propose PEW for sites which are not well suited for forest-to-bog restoration or conventional restocking. It is therefore not necessarily important to consider the effects of PEW relative to a typical forest-to-bog restoration or conventional restocking project as these might typically be established on sites that are at least to some extent suited for these management options. Instead PEW is described for sites where both forest-to-bog restoration and conventional restocking might be challenging and so less likely to provide as the desired ecosystem services (such as being a net carbon sink). Ultimately there is very little data on what benefits and challenges might be associated with PEW – a deficiency that is addressed by this thesis.

1.9. Thesis aims, objectives and outline

This thesis is the first scientific investigation of PEW. Each chapter provides evidence for answering different questions relating to PEW. Each chapter is not only intended to inform on specific implications of PEW but also the broader implications and relevance of the work to peatland and environmental management in general.

The previous sections of this chapter have highlighted the considerable knowledge gaps that exist on the impacts of PEW and also how or why such a management strategy might be enacted. As the first scientific study of PEW this thesis has two broad aims. 1) Assess opinion of PEW in stakeholder groups with a professional interest in afforested peatlands 2) Assess the climate forcing impact of PEW. The main objectives of this thesis are:

- 1) Assess the opinion of PEW of stakeholder groups with a professional interest in afforested peatlands

This objective is addressed in Chapter 2 where the main stakeholder perspectives on PEW are identified and are placed into the framework of existing environmental restoration theory. This sets the context for the PhD and how it relates to policy and practice. This context is used to guide the direction and approach of the subsequent physical science chapters.

- 2) Assess the climate forcing impact of PEW

Given the emphasis of Scottish peatland and forestry policy on carbon balance a main objective of this thesis is to understand how PEW may impact on climate forcing, which is addressed in Chapters 3-5. The climate forcing effect of peatland management options are complex and multifaceted as highlighted by section 1.6 and so multiple approaches were taken to assess this aim.

The relative recency of PEW as a concept means there are no well-established areas of PEW created according to the guidance. In order to have field sites with well-established woodland sites are used which were established before the concept of PEW existed but now appear analogous to what modern day PEW creation projects might create. The two different field sites represent quite different models of a PEW and under quite different conditions. The field site that is investigated in Chapter 3, in Rumster Forest, is situated on afforested blanket bog

in Caithness. It consists of many small plots planted with different native and non-native tree species mixes established as a second rotation on an afforested peatland. The management is relatively intensive for a PEW with trees being actively planted on peat with still active main drains. The field site that is investigated in Chapter 4-5, Flanders Moss, is situated on a lowland raised bog in the central belt of Scotland. Two plots are used at this site, both underwent forest-to-bog restoration 22-years ago but using two different restoration methods. In both the main drains were blocked and the conifer crop felling but in one the brash from the conifer crop was left on site and subsequently became colonised by stunted birch trees, the other had all tree material of the previous crop completely removed and has not subsequently been colonised by birch. Using two such different field sites enhances the breadth of the thesis. Although the focus of this thesis is on PEW, the field work has more generally applicable implications which are explored further in each chapter.

Chapter 3 aims to quantify above ground carbon stock in different second rotation woodland types on an afforested blanket bog including native species and non-native restocking options after 20 years of growth. Chapter 3 also aims to identify how difference in above ground carbon storage impact on the below ground peat carbon store.

Chapter 4 aims to quantify and compare the contemporary CO₂ and CH₄ flux from the vegetated peat surface of two plots one of which has been colonised with stunted birch regeneration. The study aims to understand causes of the difference in fluxes within and between the plots. This study also compares how the two different management approaches have affected DOC concentration in pore water. Chapter 5 develops a novel methodology to quantify the CH₄ and CO₂ fluxes directly from the birch trees in the plot used in Chapter 4 and therefore account for this less known source of gas fluxes.

A general discussion at the end of the thesis summarises the finding of the thesis and their implication for policy.

Linking Chapter 1 to Chapter 2

Having carried out the literature review summarised in Chapter 1 and having had many conversations with practitioners and policy makers, it became clear there were a wide range of ideological and pragmatic approaches to management of afforested peatland and Peatland Edge Woodland (PEW). It was clear that exploring professional stakeholders' opinions of PEW would be fundamental for developing a deeper understanding of this management option and to contextualise it in the broad framework of conservation and land use. It was also clear that there was not only a large amount of scientific uncertainty over the effects of PEW projects but also conflicting stakeholder interests. A better understanding of what PEW was interpreted as being and what uncertainties stakeholders considered most important would be valuable for determining the research questions and field sites for the rest of the PhD.

2. Innovative Restoration or Site Abandonment?

Stakeholders' Perceptions of Native Woodland

Establishment on Scottish Peatland

2.1. Implications for practice

- Novel restoration strategies provide an opportunity to investigate the underlying attitudes of those with a professional interest in ecological restoration.
- Differing perceptions of nature divide opinion on the utility of novel restoration techniques.
- Establishing native woodland to restore ecosystem services on degraded historically open peatland is being considered and practiced in Scotland with a range of interpretation on the structure and functions the woodland should fulfil.
- Research is needed to better understand the viability and effects of native woodland established on peatland and should account for the range of interpretations of how and why woodland should be established on peatland.

2.2. Abstract

For the last century large areas of peatland around the world have been drained for commercial forestry causing substantial biodiversity loss and release of soil carbon as carbon dioxide. Substantial areas of degraded peatland are now being actively restored. However, restoration poses its own challenges with the potential to be very costly while being slow to provide desired benefits. This study investigates stakeholder opinion on novel strategies to restore ecosystem services. The focus is on Peatland Edge Woodland (PEW) a new management option described in recent Forestry Commission Scotland guidance as a low-density predominantly native woodland which can be established after the first commercial forestry rotation on naturally open peatlands. PEW is proposed to restore peatland ecosystem services curtailed by commercial forestry on sites where neither continued forestry nor forest-to-bog restoration are seen as priorities and where

these are considered difficult to achieve. This study uses a mixed methods approach to explore how stakeholders with a professional interest in afforested peatland management have reacted to the concept of PEW and what factors underpin this. Various interpretations and reactions to PEW were identified and participants' views of nature are important in determining these. Interpretations ranged from perceiving PEW as having very low tree densities on well restored sites which would be expected to prevent carbon loss from below-ground sources, to perceiving it as having high tree densities on sites in less good condition with the expectation that carbon sequestration in the trees would compensate for losses from below ground. Participants with opinions opposed to PEW associated PEW with being artificial, without natural analogue and consequently inherently unstable. In contrast, participants with positive attitudes liken PEW to natural habitats or embrace the concept that natural processes might potentially create a valuable habitat in a site that would have limited potential for open-bog restoration or conventional restocking.

2.3. Introduction

The importance of ecological restoration is shifting to the political mainstream, a trend accentuated by the United Nations having declared 2021-2030 as a 'Decade on ecosystem restoration' (United Nations, 2019b). With growing political and economic support for ecological restoration the objectives of ecological restoration increasingly need to be pragmatic and rooted in the interests of society in general (Choi, 2007; Martin, 2017).

Traditionally, ecological restoration is based on the concept of historical fidelity whereby projects aim to either restore a natural or cultural ecosystem previously present in terms of structure and composition or restore it to its original trajectory before human disturbance (Society for Ecological Restoration International Science & Policy Working Group, 2004). Increasing human disturbance such as climate change, land use and species introduction can make traditional approaches to ecological restoration more challenging (Hobbs et al., 2009; Higgs et al., 2014; Lennon, 2017). The concept of novel ecosystem restoration has developed in response to this and is an approach that does not aim to restore the pre-disturbance ecosystem and potentially accepts multiple potential trajectories for a site in the future (Choi, 2007). This flexibility is argued to allow the recovery of ecosystem functions and the establishment of more resilient habitats on sites where conventional restoration may be difficult to achieve (Hobbs et al., 2009; Higgs et al., 2014; Lennon, 2017). Restoring ecosystems to "novel states" often puts

emphasis on the ecosystem services a site might be able to provide, which may be similar or different to those the site might have originally provided. The objectives for ecological restoration and how much novel components should be embraced as part of projects is a contested issue (Miller and Bestelmeyer, 2016).

Degraded peatlands are an obvious priority for ecological restoration as despite covering ~0.3% of terrestrial land area are responsible for 5% of global anthropogenic carbon dioxide (CO₂) emissions (Joosten, 2009; Joosten et al., 2016; Leifeld and Menichetti, 2018). Conversely peatland in good condition provides a multitude of ecosystem services, such as fire prevention, water purification and carbon storage (Kimmel and Mander, 2010; Bonn et al., 2014). The naturally waterlogged conditions in which peatlands form make them unproductive for direct economic exploitation leading to a history of peatlands being drained to increase productivity for practices such as forestry, agriculture, or peat mining (Bain et al., 2011). Draining peatlands results in the loss of ecosystem services, for example the lowered water table increases the oxidative decay of peat resulting in the release of CO₂ (Joosten, 2009; Lindsay et al., 2014b). The loss of increasingly valued ecosystem services in combination with the often-low productive value of drained peatland is motivating international efforts to restore large areas of peatlands (Schägnier and Schaller, 2008; Bain et al., 2011)(Schägnier and Schaller, 2008; Bain et al., 2011).

In Scotland peatlands cover over 20% of land area (Cannell et al., 1993; Chapman et al., 2009), an estimated 17% of which is commercially afforested (Vanguelova et al., 2018). A majority of Scottish peatland forestry was established on formerly treeless low nutrient ombrotrophic peatland (Sloan et al., 2018b). This is similar to some nations like Ireland but contrasts with other nations like Finland where peatland forestry is more common on naturally wooded peatland or more nutrient rich fen peat (Sloan et al., 2018b). Scottish peatland forestry was promoted in the second half of the 20th century on the grounds that it would decrease the UK's dependency on timber imports while creating rural employment and utilising otherwise unproductive land. However, due to doubt concerning the extent to which these benefits were delivered and environmental concerns a lot of the government incentives for establishing forests on peatlands were removed in the late 1980s (National Audit Office, 1986; Stroud et al., 2015) with increasing regulation and guidance against the expansion of peatland forestry coming into effect (Forestry Commission, 1990; Patterson and Anderson, 2000; Forestry Commission Scotland, 2014).

The establishment of new plantations on peatland is no longer supported by forestry policy (Forestry Commission Scotland, 2014) but there is some debate over what should be done with the existing plantations, many of which are now reaching the end of their first rotation. The

two options which have thus far dominated future management of these sites are restocking with commercial tree species and forest-to-bog restoration but there is disagreement between stakeholders about the appropriate balance of the two. This can create a degree of antagonism between stakeholders with different professional interests in the management of afforested peatland (Payne and Jessop, 2018). The current potential for antagonism is rooted in historic tensions as the peatland plantations were historically actively established and promoted by the Forestry Commission (the government department and statutory body responsible for forestry in Scotland at the time) and private forestry companies while several conservation Non-Governmental Organisations (NGOs) and the Nature Conservancy Council (NCC, the UK government's nature conservation agency at the time) campaigned against them.

In 2014 and 2015 Forestry Commission Scotland released new guidelines on how afforested peatlands should be managed at the end of a first rotation (Forestry Commission Scotland, 2014, 2015), the decision-making framework from this is summarised in figure 2.1. This guidance intended to balance the perspectives of different stakeholder groups. These guidelines are very focused on how management affects carbon balance, reflecting a general shift in the importance of climate change mitigation in Scottish and international policy. The guidelines support forest-to-bog restoration as a priority if the site meets certain environmental criteria or if the site will not support at least 20% tree canopy cover. In situations where restoration is not an environmental priority and the site can support good commercial tree growth, defined as at least yield class eight for Sitka spruce, then conventional restocking is advised. The guidelines posit that above this growth rate trees will sequester carbon at a rate equal to or greater than the carbon released by the peat as a result of a second rotation, a conclusion the guidelines base on (Morison et al., 2010). The guidelines propose that if a site does not meet the criteria to be restored or restocked then the area should be converted to a low-density, low-intensity, predominantly native woodland; this has been termed Peatland Edge Woodland (PEW).

On appropriate sites, PEW is intended to provide biodiversity and ecosystem service benefits of both peatland and woodland (Forestry Commission Scotland, 2014, 2015). For example, the lower density of the trees may allow some recovery of the site's water table, thereby reducing oxidative loss of carbon from the peat, as forest-to-bog restoration would while the growth of the trees will sequester carbon, hypothetically resulting in no net loss of carbon (Forestry Commission Scotland, 2015). In this way PEW is proposed to combine the benefits of conventional restocking and open-bog restoration on sites where neither of these alternatives would be easily practicable (Forestry Commission Scotland, 2015).

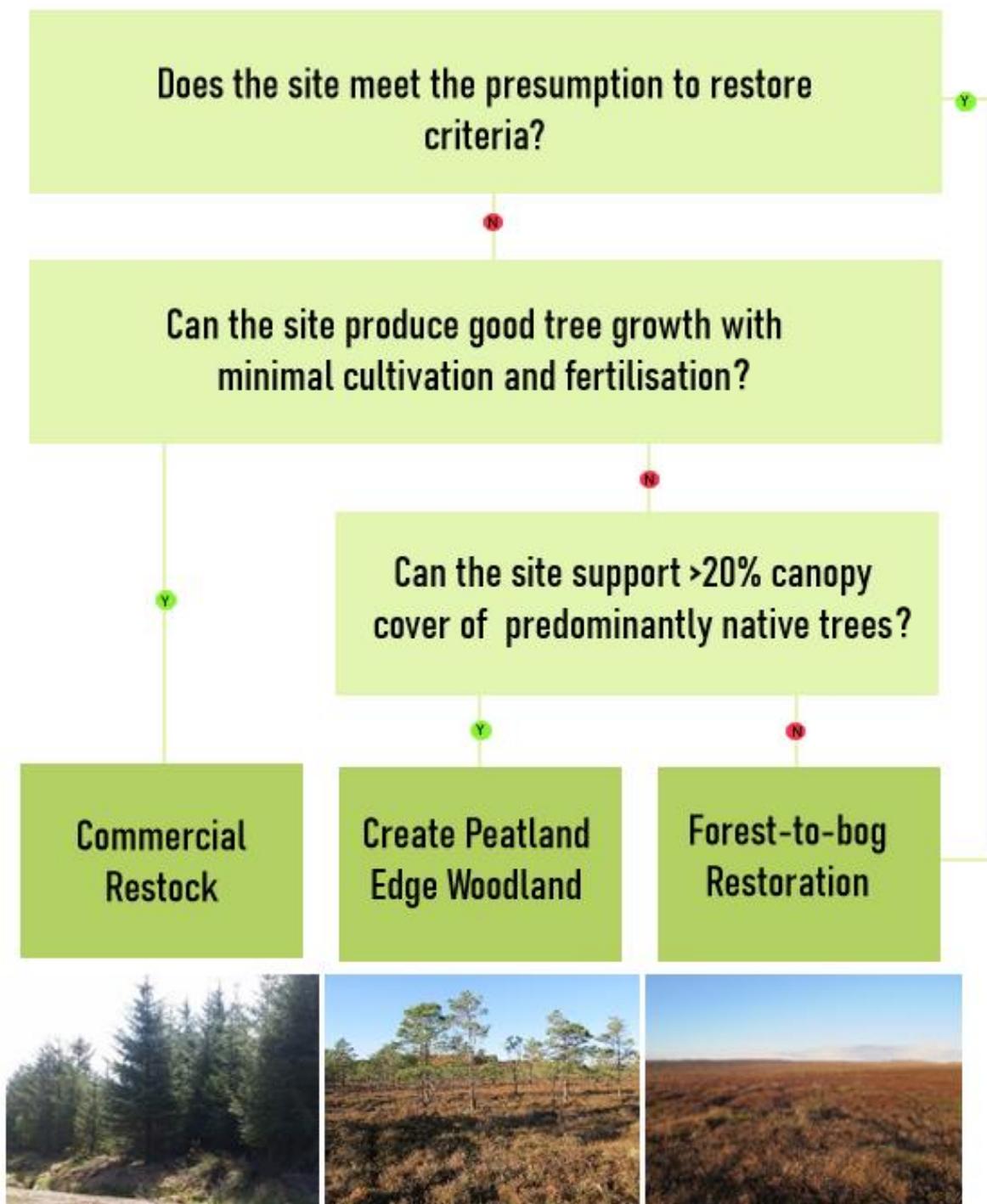


Figure 2.1: Decision making framework deciding the future management of afforested peatlands, modified from Forestry Commission Scotland (2015) and with author's images to illustrate the options

There is no direct research or case studies on whether PEW would provide the proposed ecosystem service benefits. Also, little is known about potential trajectories of an area established as PEW. Tree colonisation of degraded peatlands is common in Scotland and internationally and is a challenge for projects trying to restore open bogs (Van Seters and Price, 2001; Sotek et al., 2019). This natural colonisation could be utilised by those establishing PEW (Forestry Commission Scotland, 2015) but it is unclear how possible it would be to create a stable low density woodland cover on disturbed peatland. There has also been suggestions that PEW has been proposed as a way to justify cheaply abandoning sites that could be restored to open bog with proper investment (RSPB, 2014).

The guidelines on PEW do not explicitly refer to PEW as a novel ecosystem but it is described as a 'new model' for managing this land and shows many of the characteristics of a novel ecosystem. PEW is not necessarily required to restore the original peat formation function of a site, instead carbon sequestration through tree growth is expected to at least equal carbon loss due to peat decomposition (Forestry Commission Scotland, 2015). Non-native tree species are also stated to be permissible in non-woodland-edge situations if they improve the functionality of the habitat (Forestry Commission Scotland, 2015). Finally, most undrained ombrotrophic peatlands in Scotland are generally quoted as being naturally treeless (Payne et al., 2018), therefore any woodland present could be considered a novel state.

Counter to PEW being a novel ecosystem is that there are rare examples of bog woodland in Scotland (Wells, 2001; McHaffie et al., 2002; Joint Nature Conservation Committee, 2007) which could be a model for PEW as they contain native woodland and retain bog vegetation. The extent to which Scottish ombrotrophic peatland might be wooded without the historic influence of human activity is unclear (Warren, 2008) but influences such as deforestation, commercial afforestation, inappropriate grazing levels, and burning may have reduced it (MacKenzie and Worrell, 1995). The PEW guidelines states that although PEW has not been specifically designed for this purpose it could be modelled on priority habitats such as EU Habitats Directive Annex 1 Bog Woodland. So, depending on how PEW is created, it could be interpreted as recreating a forgotten natural stage, not a novel ecosystem.

This study focuses on PEW as defined by the Forestry Commission Scotland's comprehensive guidelines but the idea of restoring degraded peatland sites to native woodland has been suggested by other sources and in other parts of the UK (Cariss, 2011; Anderson et al., 2014) and internationally (Renou et al., 2006; Corrigan and Nieuwenhuis, 2016). There is also a precedent of restoring degraded peatland to other novel ecosystems such as lakes (Higgins and

Colleran, 2006). The large gaps in understanding around PEW, in combination with stakeholder groups having different management priorities, means there is high potential for different interpretations and value judgements to have been made of PEW. The aim of this study is to understand how different stakeholder groups with a professional interest in afforested peatland management have responded to the concept of PEW and what the factors which underpin differences in this response are. The study explores this aim within terms of the following objectives: determining 1) how widely the idea of PEW has been disseminated amongst stakeholder groups in peatland management, 2) what stakeholder motivations are for establishing PEW, 3) what professional stakeholder concerns are for establishing PEW, 4) how (if at all) stakeholders would create PEW, 5) if different stakeholder groups differ in opinion on PEW. It is hoped that this study will encourage debate and review around this novel restoration technique and novel ecosystem restoration of peatlands and other habitats in general.

2.4. Materials and methods

A two-part mixed methods exploratory sequential design is used to investigate stakeholder attitudes towards PEW, where the results of an initial qualitative investigation (Stage 1) were used to develop a quantitative survey (Stage 2) (Greene et al., 1989). The mixed methods approach was necessary due to the lack of prior knowledge about the key issues for the varied stakeholders to be sampled. The integration of qualitative and quantitative methods allowed exploration and identification of the key themes in detail during the qualitative stage and then test the results further through specific targeted questions in the quantitative survey to assess the generalisability or magnitude of the effects with a larger sample (Fetters et al., 2013). The mixed methods approach has the additional benefit of triangulation where confidence in inferences are improved when supported by two methodological approaches (Molina-Azorín and López-Gamero, 2016).

Stakeholder groups involved in afforested peatland management were identified in an initial exploratory phase reviewing literature and having informal conversations with potential stakeholders. Payne & Jessop (2018) had previously identified key stakeholder groups in afforested peatland management, and this was used as the main basis for the exploratory phase. The following stakeholder groups were identified: public sector forestry, private sector forestry, conservation NGOs, governmental or statutory bodies, other private sector land users (such as windfarms and estate owners) and research organisations.

Stage 1: Qualitative survey

The aim of the qualitative survey was to understand the breadth of opinions that existed relating to the overall objectives rather than being representative. For this reason, purposive sampling was used to select a diverse range of participants in different stakeholder groups. A degree of 'snow-balling' sampling was also used when interviews highlighted oversights in the sample. In total 15 interviews were conducted ranging in length from approximately 20 minutes to an hour.

Interviews were semi-structured following a schedule (see Appendix 1 for a full copy of interview the interview schedule) that included key questions and themes to cover, including their awareness of PEW, their motivations and concerns for establishing PEW and how they would envisage establishing and maintaining PEW, but giving scope to explore issues important to the participants as has been promoted for stakeholder analysis (Grimble, 1998). Participants have been kept anonymous. All interviews were audio recorded and manually transcribed by the same investigator assisted by Express Scribe Transcription Software (NCH Software, 2017). A thematic analysis was carried out on the transcripts assisted by NVivo 11 qualitative data analysis software (QSR International Pty Ltd, 2017). Initial codes were developed from reading the transcripts, these codes were refined, and the key themes identified through comparisons within and between texts.

Stage 2: Quantitative survey

The quantitative survey was intended to test the generalisability of the findings of the qualitative survey. Likert and multiple-choice style questions were developed based on the results from the qualitative survey in order to investigate the objectives further in a quantitative survey. The qualitative stage highlighted various issues that could affect how participants answered questions, for example some participants thinking of PEW as a shallow peat 'rand' woodland. The quantitative survey therefore provides a definition of PEW taken from the Forestry Commission Scotland (2015) and asked participants to use this definition when answering the subsequent question. Another common issue in the qualitative stage was participants remarking that the answer to some of the questions would be very site specific. These issues were considered when writing the questions so that they should be clear and logical to all participants. The nature of the quantitative survey meant that participants had to answer for a typical situation without scope to explain any particular context in detail. There is potentially a lot of variability between the sites

that different participants manage so the context could be quite different. The quantitative data therefore gives a more simplistic impression but is contextualised by the qualitative stage.

The survey tool Qualtrics (Qualtrics, 2005) was used to construct an online survey. The survey consisted of 45 closed questions. The quantitative survey starts with two opening questions, similar to those used by Payne and Jessop (2018), the first was to confirm that the participant had a professional interest in afforested peatlands or forest-to-bog restoration. The second asked participants to specify their employer or interest in afforested peatlands. This was to assess the representativeness of the survey but also to facilitate the identification of systematic differences between stakeholder groups. The remaining questions asked directly about PEW grouped under the four subheadings: 1) awareness and reception, 2) establishment and management, 3) motivations and 4) concerns (see Appendix 2 for a full copy of the survey).

Participants were recruited by distributing a hyperlink to the survey. The hyperlink was sent out to a modified version of the mailing list used by Payne and Jessop (2018) which had a very similar target audience. To increase the number and representativeness of participants the hyperlink was also distributed by social media (e.g. twitter) and published in a number of relevant newsletters.

In total 69 surveys were received, of these 11 had not been completed beyond stating their stakeholder group affiliation (which was an introductory question before the main survey), these respondents were not disproportionately from any stakeholder group. These mostly incomplete responses were discarded. Of the remaining 58 responses 22 of these had at least some missing responses, equating to approximately 11% unanswered questions across all analysed surveys. Missing responses were not disproportionately associated with any stakeholder group. Furthermore, there is no significant relationship between whether the survey was complete and how familiar participants reported being with PEW ($t=0.336$, $p=0.738$) or how often they thought PEW might be a good management option ($t=0.680$, $p=0.499$). This provides confidence that the missing answers will not be disproportionately from any particular viewpoint.

Of the 22 missing responses six had more than a third responses missing. Of these six participants, two participants stated their stakeholder group as being 'other private sector organisation' which represented 40% of those from this stakeholder group. It is unclear whether this might reflect a systematic difference, possibly whereby those from this stakeholder group felt the survey did not capture their perspective well, or alternatively the format of the survey was not well suited to this group (e.g., this group may experience disproportionately high time pressures and be unable to take time to fill in the survey). Given the small sample size it could also

very plausibly be a random difference. Of these two, one claimed to be familiar with the precise definition of PEW while the other stated that they had not heard of PEW before, so there is no indication that level of knowledge was an issue in response rate for this stakeholder group. Excluding the two 'other private sector organisation' responses the remaining four responses containing more than a third unanswered questions were each from different stakeholder groups.

Question 7 (which asked about what factors might limit the popularity of PEW) and Question 14 (which asked participants to rate images out of five for how well they represented PEW in their opinion) disproportionately had missing answers. On re-examination question 7 appeared to be a difficult question to answer as it requires the participant to speculate about what might affect the popularity of PEW amongst all stakeholders in general rather than asking for their personal opinion. Question 14 had been a challenging question to construct and had almost been excluded from the final survey. The question required pictures of PEW when no purpose grown PEW exists, necessitating the need for pictures of proxy sites rather than purposely created PEW sites. It is also difficult to capture a sense of tree size and density as well as the ground flora composition, surface moisture, level or drainage in only one picture. The disproportionately high missing answers on this question are therefore interpreted as participants finding it difficult to interpret what is being shown in the pictures.

The 58 useable surveys had a reasonable coverage of the stakeholder groups (figure 2.2), with a particularly high representation of participants who worked for conversation charities.

SPSS (IBM Corp., 2020) was used to explore the data, generate descriptive statistics and test for pairwise interactions between variables. R (R Core Team, 2021) was used to make imputed data sets with predictive mean matching, carrying out K means cluster analysis and Principal Component Analysis (PCA) using additional packages Mice (Van Buuren and Groothuis-Oudshoorn, 2011) and Cluster (Maechler et al., 2021). Multivariate tests were carried out on ordinal variable data which were first normalised on a scale of 0-1 to avoid bias that can occur when questions are asked on different measurement scales. Multivariate tests were carried out on an imputed data set including all surveys with greater than 50% responses (n=52) and a complete case data set consisting of only fully completed surveys (n=36), the results of using both data sets were compared.

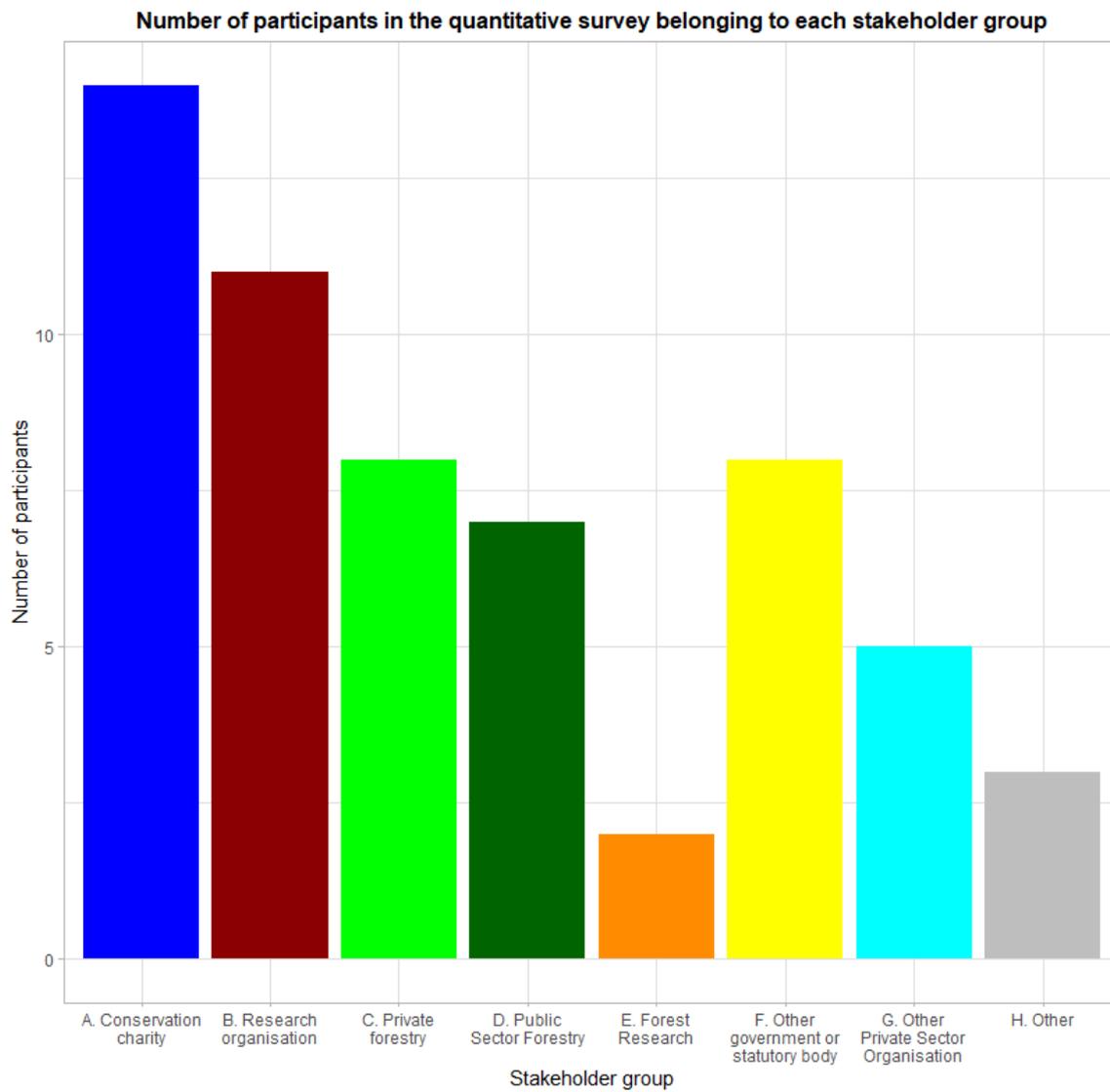


Figure 2.2: Number of participants in the quantitative survey belonging to each stakeholder group.

2.5. Results

2.5.1. Awareness

The study highlights that awareness of PEW has permeated across all the stakeholder groups investigated. Across all participants in the quantitative survey 24 participants (41%) were familiar with the official guidelines on PEW with a further 18 participants (31%) having heard of the term. Awareness of PEW was greatest in public sector forestry, but in each of the other stakeholder groups more than 50% participants reported familiarity with the term 'Peatland Edge Woodland'. Sixteen participants (28%) were involved in planning or managing PEW, these were disproportionately in public sector forestry (6 out of 7 public sector forestry participants).

2.5.2. Motivations

Most qualitative stage participants (regardless of level of support or opposition for PEW) emphasised the need for a holistic approach to peatland management which considered the multiple benefits a management option might provide; some even criticised the Forestry Commission guidelines for focusing too much on carbon storage. The following were identified as potential benefits of PEW established in appropriate places: habitat connectivity, habitat diversity, supporting biodiversity associated with woodland edge habitats, diversity, recreation of naturally wooded peatland, climate change mitigation, landscape aesthetics, providing a barrier to non-native regeneration out of adjacent commercial plantations, flood mitigation, soil erosion prevention, low intensity wood extraction, satisfy legislative obligations or targets, and PEW being more achievable/cheaper than alternative management options. The quantitative survey showed that from all the potential benefits of PEW listed in the qualitative survey those linked to creating natural habitats and enhancing biodiversity were most commonly voted for while only two participants (3%) thought the more conventional use of supplying wood was a main benefit of PEW (figure 2.3).

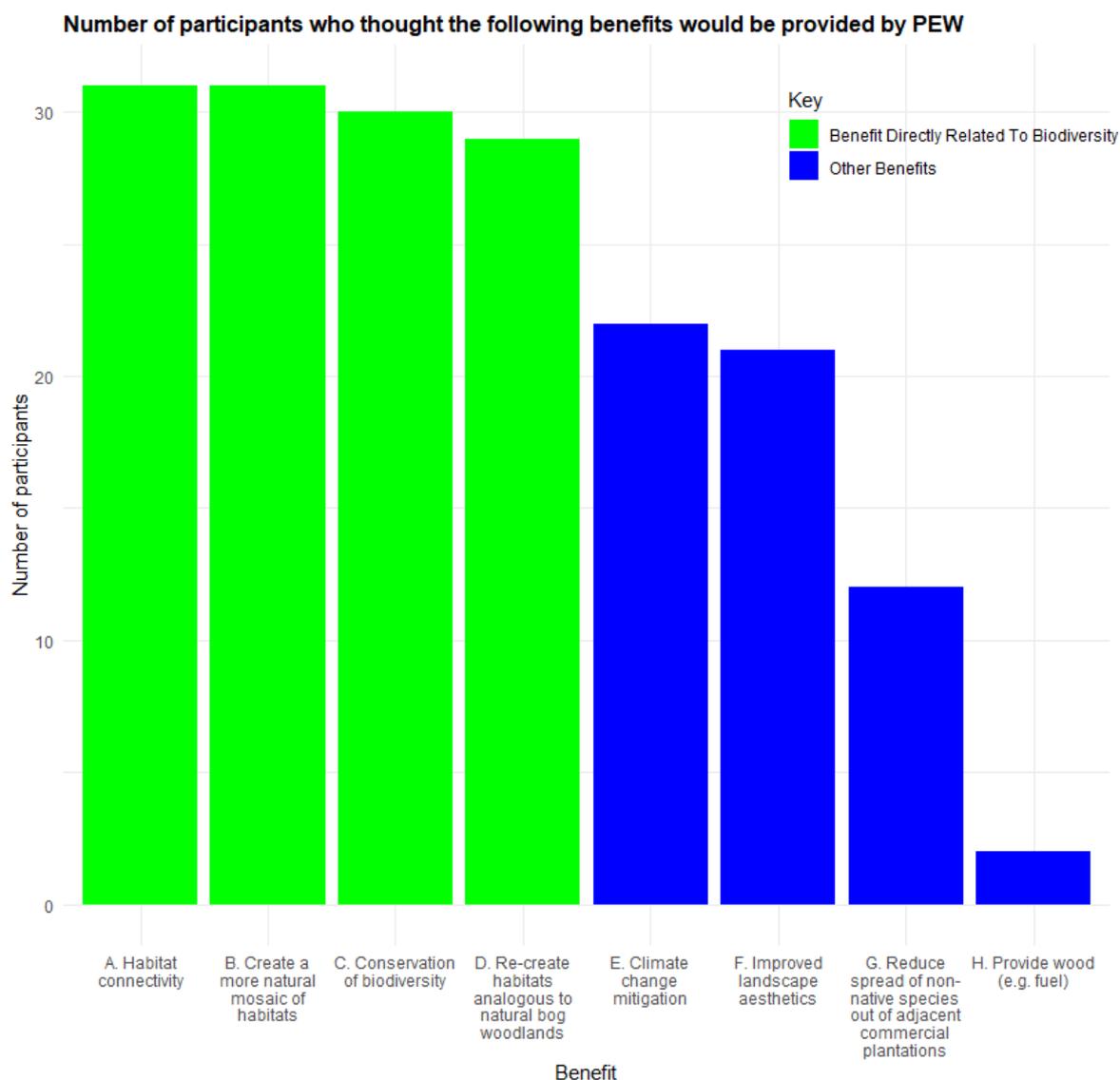


Figure 2.3: Graph showing number of participants stating PEW may provide certain benefits.

Both data collection stages highlighted that many stakeholders saw PEW as a positive opportunity to create a more natural and sustainable landscape. The qualitative survey brought up ideas that PEW could be a natural state for Scottish peatlands referring to how bog woodlands are more common internationally, evidence of more bog woodlands in the Scotland’s palaeontological record, the fact that anthropogenic influence may have reduced the area of bog woodland and that under current conditions certain peatland areas become wooded by natural colonisation of trees. The quantitative survey confirmed the importance of this natural view of PEW with 31 participants (53%) voting for ‘creating a more natural mosaic of habitats within a

landscape' and 29 participants (50%) voting for 'recreating habitats analogous to natural bog woodland found in Scotland and internationally' as main benefits of PEW (figure 2.3).

The idea of PEW being natural was also expressed in terms of establishment of PEW on appropriate sites potentially requiring fewer interventions compared with for example, a restored site that would need repeated clearing of tree regeneration or a restocked site that might need repeated 'beating up' to replace saplings lost due to high mortality rates. If such a site was developed as PEW, then this might provide more flexibility on managing the site. The quantitative survey supported this idea of low intervention with the idea of establishing PEW by regeneration being more popular than active planting on deep peat, 29 (50%) and 10 (17%) participants respectively.

As well as identifying that PEW is seen by some as a positive opportunity, the qualitative stage identified that a motivation for some of those who would consider establishing PEW is 'making the best out of a bad situation'. This attitude frames PEW as a pragmatic management option that provides the best benefits given constraints and a negative legacy on a site. This attitude was captured in the quantitative survey with over two thirds of participants agreeing with the statement that PEW offers the best benefits when a site is 'neither a good candidate for conventional restocking nor for restoration', although their level of agreement was generally weak, with more than half of respondents only 'somewhat agreeing' (figure 2.4).

Agreement with the statement that PEW offers the best benefits when a site is neither a good candidate for conventional restocking nor for open-bog restoration

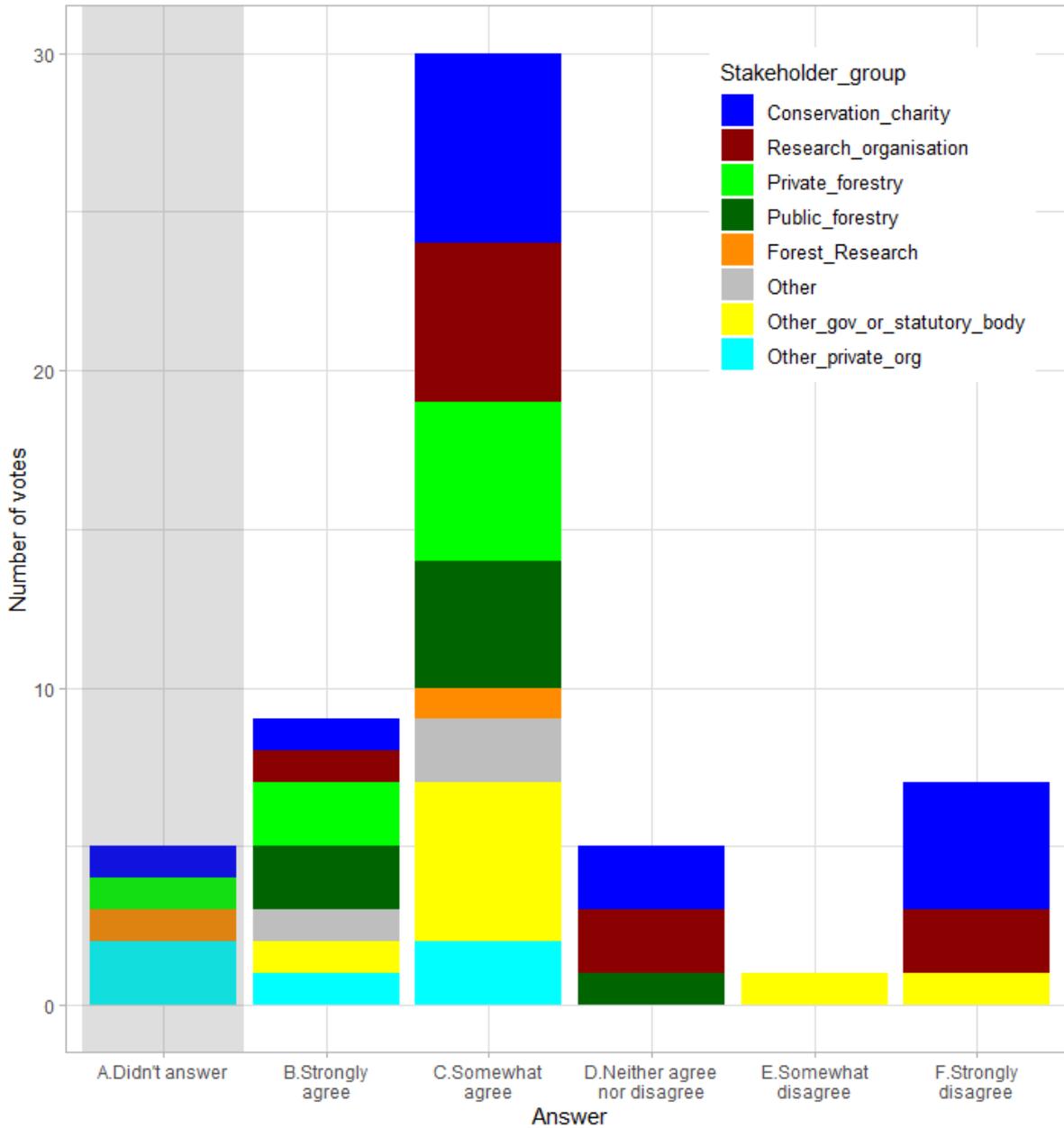


Figure 2.4: Level of stakeholder agreement on whether PEW offers the best benefits when a site is 'neither a good candidate for conventional restocking nor for restoration', each bar is divided into coloured sections depending on the number of participants from each stakeholder group.

2.5.3. Concerns

The qualitative stage identified four key concerns regarding the establishment of PEW. These were each investigated further with one or more questions in the quantitative survey where participants were asked if they were concerned about different aspects of PEW (figure 2.6).

1. Lack of evidence/opportunity cost

The qualitative stage highlighted a concern that PEW could use land and resources which could otherwise have been used for open-bog restoration or conventional restocking. Twenty participants (34%) in the quantitative survey said a lack of evidence that PEW could be established in such a way that desired benefits were achieved was a definite concern. In the absence of data on PEW, many stakeholders are concerned about when or whether PEW could represent a better option for achieving objectives such as climate change mitigation or biodiversity benefits than open-bog restoration or conventional restocking.

2. Unnatural habitat

The qualitative stage identified the opinion that if a site is not being used productively then there is a duty to restore the site to a pre-disturbance open bog state which is a highly valued conservation priority habitat. Nine participants (16%) in the quantitative survey were concerned that PEW was an artificial habitat (figure 2.5). Although the total percentage of participants who expressed this as a concern in the quantitative survey is relatively low, 75% of those who were opposed to PEW being established anywhere voted for this concern in contrast to only 7% of those who thought PEW was an appropriate management option in at least some situations.

3. Unstable/unsustainable habitat

The qualitative stage highlighted the concern among some participants that PEW could be unstable with an uncertain trajectory and future management objectives. Some stakeholders had concerns that the trees could be too successful resulting in closed canopy woodland causing further decline in the underlying peat without periodic tree thinning. In particular there was a concern that PEW would facilitate excessive non-native tree regeneration – this was selected as a

concern by the most participants in the quantitative survey with 37 participants (64%) of participants voting for it as a main concern. There was also the contrasting concern that the trees could not grow well enough, with the site reverting to open bog, making the original planting a waste of resources, 20 participants (34%) in the quantitative said this was a main concern.

Concern about the instability of PEW also related to how it might impact on surrounding areas if trees grew well. In the quantitative survey, 22 participants (38%) were concerned about spread of native trees into designated open areas. In the qualitative survey, participants were particularly concerned in cases where areas adjacent to the plantation were recently restored sites which may be more vulnerable to the regeneration of trees.

4. Establishment in inappropriate sites

There were concerns that the definition of PEW was such that some practitioners might abandon sites after felling the last crop and label any tree regeneration as PEW rather than pursue more expensive but more appropriate management of the site. There was particular concern that the pressure of national forest area expansion targets in Scotland had motivated the proposal of PEW and these pressures would drive the establishment of PEW on areas that would be more suitable for forest-to-bog restoration, 26 participants (45%) in the quantitative survey voted for this as a main concern making it the second most common concern. In the qualitative stage none of those interviewed who supported PEW expressed their support for PEW primarily in terms of reducing costs and/or satisfying planting targets but instead viewed these as subsidiary benefits of establishing PEW.

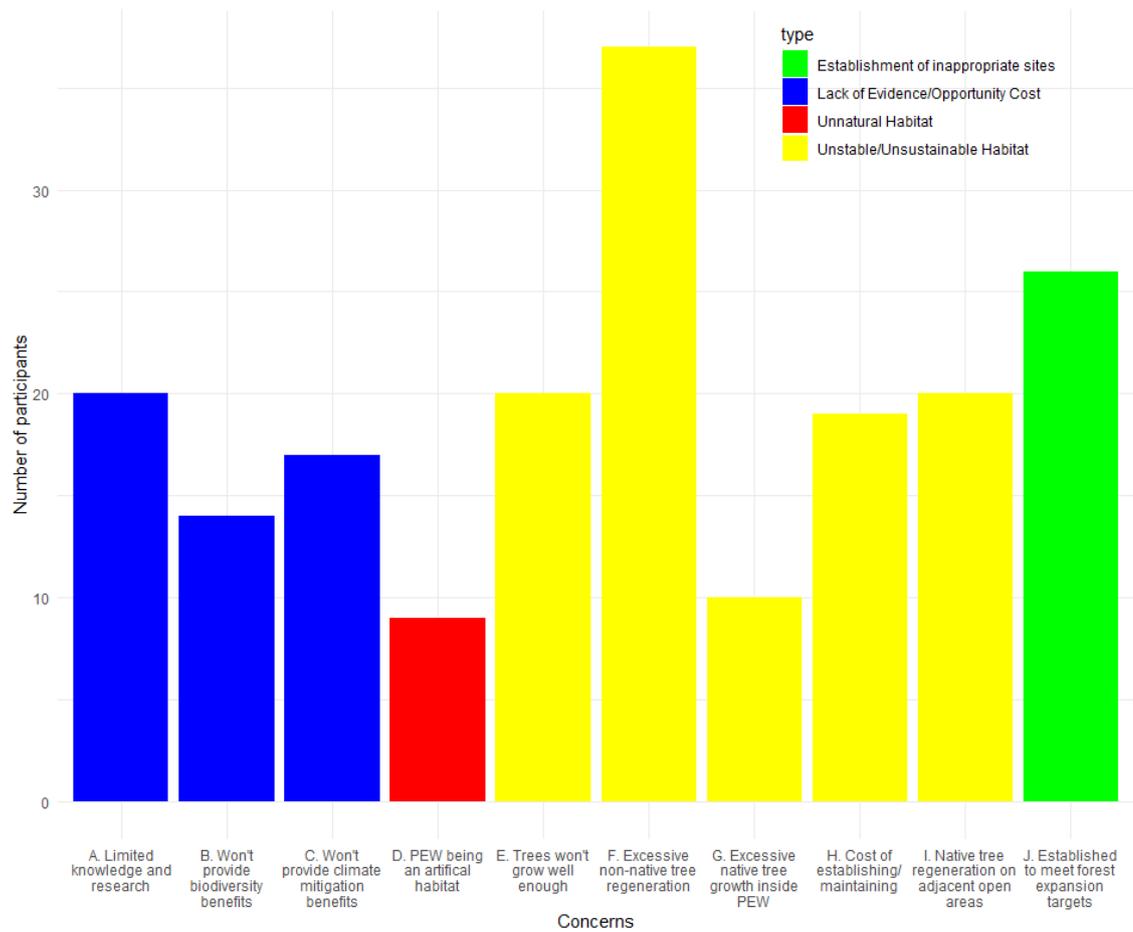


Figure 2.5: Number of participants who voted for being concerned about different potential problems of establishing PEW. The concerns are in coloured groups according to the four identified categories of concerns: 1) Establishment of PEW on inappropriate sites. 2) Lack of evidence for the benefits or PEW and/or concerns about the opportunity cost of establishing PEW when the benefits associated with other management are better understood. 3) PEW being an artificial/unnatural habitat. 4) PEW being an unstable or unsustainable habitat.

2.5.4. Establishment

Ideas about how to establish PEW varied between participants with each participant who didn't entirely oppose PEW in the qualitative survey saying it would be very dependent on the characteristics of a site and the specific objectives for it. The qualitative stage identified three main aspects of establishing PEW: a) where it is appropriate to establish it, b) composition, density, and growth rate, and c) how the desired tree cover would be established and maintained.

a) Where is it appropriate to establish PEW?

In the Quantitative survey when asked how many sites PEW might be a good management option for, 48 participants (83%) thought that there would be at least a few situations where PEW could be a good management option, while 20 participants (34%) thought it would be a good option for more than the occasional site (figure 2.6). The qualitative survey identified two situations in which a site might be suitable for PEW (figure 2.7):

1) On ecotones (Ecotone model - figure 2.7a). PEW could be established on naturally occurring ecotones where the deeper, wetter, and lower nutrient peats transition to shallower, dryer and more nutrient rich peats. Examples include approaching riparian areas, along an altitudinal gradient at the lower altitudinal margin of a blanket bog, in areas of blanket bog with variable peat depths due to uneven underlying topography and in the periphery of raised bog. It was said that PEW could be established in these ecotones potentially in combination with restoration in areas with deeper peat and conventional restocking in areas with shallower peat, with PEW forming a transition.

2) On difficult sites (Pragmatic model – figure 2.7b): PEW might be considered suitable on difficult sites where neither open-bog restoration nor restocking was seen as appropriate, regardless of the inherent properties of the site. For example, on smaller sites embedded in predominantly afforested areas where the conditions make the site unsuitable for restocking yet forest-to-bog restoration would be practically and financially untenable given the small scale and persistence of drainage and seed sources adjacent to the site. This approach to PEW appeared to be more relevant to the forestry sector which would be more likely manage such sites embedded in wider areas of active commercial forestry.

Participants views on how many situations PEW could be a good option for managing deep peat sites after commercial forestry

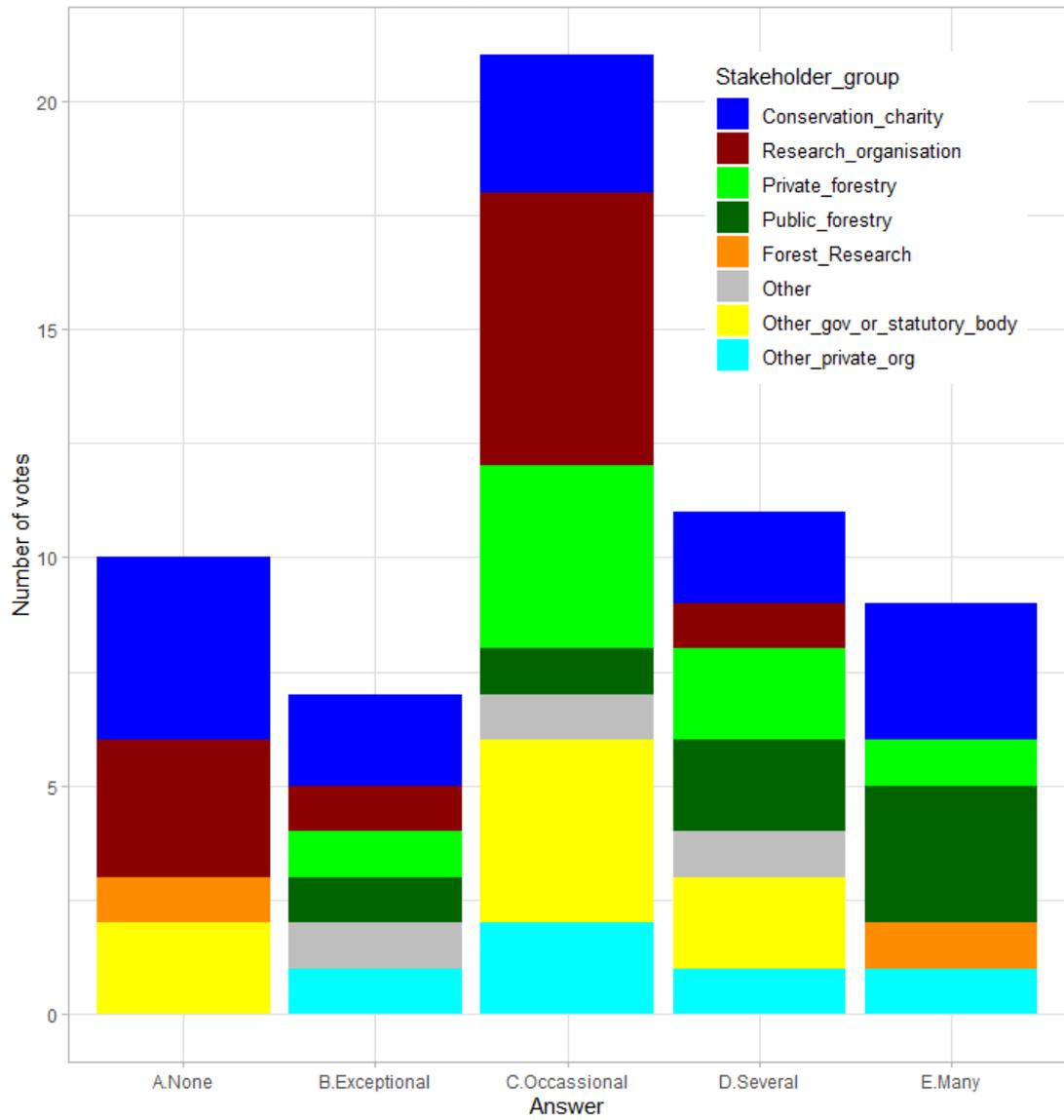
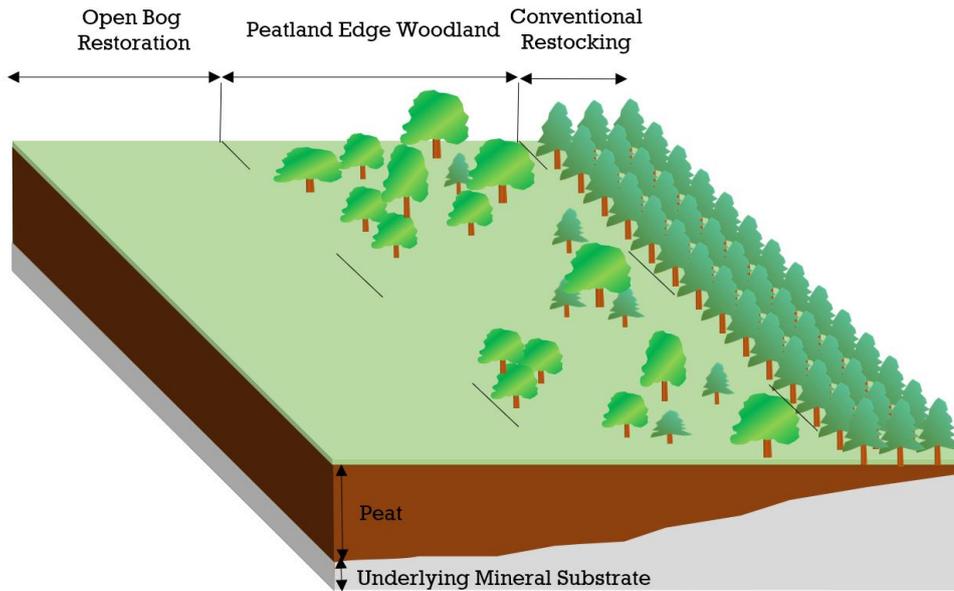


Figure 2.6: Participants’ opinion on how many situations PEW might be a good option for the management of deep peat sites after commercial forestry on. All participants answered this question.

a)

Ecotone Model



b)

Pragmatic Model

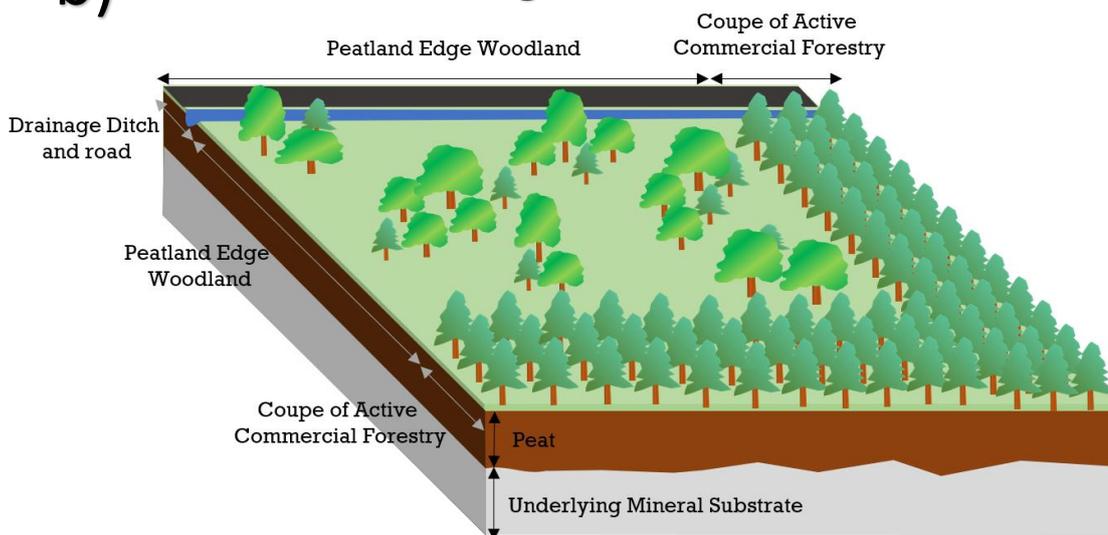


Figure 2.7: A visualisation of two different situations in which PEW establishment may be suitable. a) Ecotone model - PEW is established on a naturally occurring ecotone between areas more suitable for open-bog restoration and conventional restocking. b) Pragmatic model - PEW is established on an area of ground where PEW is deemed an appropriate option, but this does not reflect any particular transition in natural properties. There may be a range of reasons for PEW to be established for this reason but in this example PEW is established in a coupe which has been harvested but there are younger coupe of trees still growing surrounding, the presence of which may disrupt restoration efforts. In addition, a main drainage ditch would hamper restoration effect, yet it has been left to protect an adjacent road. The proximity of the road may have been an additional motivation for creating PEW (i.e. for aesthetic reasons).

b) Composition, density, and growth rate of tree cover

In the qualitative survey, PEW was most commonly envisaged as having the composition of National Vegetation Classification (NVC) W4, this is a typically *Betula pubescens* (downy birch) dominated woodland though may feature other species such as *Salix caprea* (goat willow) and *Sorbus aucuparia* (rowan). *Pinus sylvestris* (Scots pine) was also commonly suggested as appropriate in the right areas but usually with concerns that the harsh peatland conditions in combination with its susceptibility to disease would lead to poor growth and survival. In contrast, the questions in the quantitative survey which showed an image of a relatively open Scots pine woodland and the image of a relatively open birch woodland growing on a peatland had similar popularity as models for PEW in the quantitative survey.

According to the guidelines (and broader policy) PEW has to have a canopy cover of over 20% to qualify as woodland. Opinions on tree density in PEW varied, but no participants thought PEW should have continuous cover of well growing trees. The most popular options were for those with mixes of open and wooded patches either with stunted (22 participants, 38%) or well growing trees (16 participants, 28%). PEW is described as a predominantly native woodland in the guidelines (Forestry Commission Scotland, 2015) but the qualitative survey highlighted different interpretation of what proportion of non-native was acceptable. In the quantitative survey, participants generally favoured control of non-native species within tight limits with only 15 participants (26%) tolerating greater than 5% non-native canopy cover, although 3 participants (5%) did not consider there to be a need to keep non-native tree cover in any limit.

c) How desired tree cover would be established and maintained

The qualitative survey highlighted a range of ways that trees could be established in PEW. Establishing PEW with regeneration was most popular in the quantitative survey with 29 participants (50%) saying they would definitely consider establishing PEW with regeneration from natural sources. Eighteen participants (31%) indicated that they would consider encouraging native tree regeneration onto deep peat PEW sites by planting adjacent shallow peat/mineral soil with trees to act as seed sources and only 10 participants (17%) stated they would plant native trees on deep peat to create PEW.

Participants had different views on how controlled PEW development should take place. In the qualitative stage, several participants commented on how some sort of grazing control would be required to allow trees to establish. Generally, participants in the qualitative stage

advocated for keeping interventions - such as secondary planting or thinning if tree cover decreased or increased from defined targets - to a minimum. Many of the participants in the qualitative survey were open to allowing PEW to develop naturally even if this meant accepting some PEW would revert back to open bog and some would become closed canopy woodland. Many believed PEW would reach a natural equilibrium even expanding and becoming denser and then thinning and receding from some areas in response to climatic change.

Opinion varied on the need to remediate historic drainage in PEW sites. In the quantitative survey, 13 participants (22%) said no drainage remediation would be required, 14 participants (24%) said some drainage remediation (e.g. blocking main drain) would be required and 28 participants (48%) said extensive rewetting (e.g. furrow blocking, ground smoothing) would be required. The qualitative survey highlighted that those advocating for at least some rewetting saw this as necessary to recover peatland functioning, such as reducing carbon oxidation but also as part of managing tree regeneration within PEW. Some argued that rewetting should be successfully completed before allowing trees to establish as this would help control tree growth within desired limits. However, some were concerned that if the site was quickly and thoroughly rewetted then PEW would not establish well, resulting in the reduction of services such as carbon sequestration in tree growth. These participants argued that if intensive efforts were made to restore undrained conditions then it would make more sense to restore to open bog.

2.5.5. Overall comparison between variables and stakeholder groups

The qualitative stage indicated that participants' conceptions of PEW fell on a spectrum in terms of the sort of PEW they envisaged. Some imagined something close to a natural bog woodland established on well-rewetted peatlands and colonised and dominated by native tree species established through natural regeneration. These participants spoke of the need to restore the ecosystem services associated with functioning peatlands. Concerns were focused on preventing excessive tree growth degrading the peatland habitat. At the other end of the spectrum, PEW was seen as a more intensive/higher density woodland where native and non-native trees were intended to grow relatively well with benefits associated with the trees compensating for less recovery of the peatland.

K-means cluster analysis for both the imputed data and complete data, indicated that two or three clusters described the data best, results for complete data with three clusters are shown in figure 2.8. When resolving both two and three clusters, those very opposed to PEW made up one cluster (cluster 1 in figure 2.8). When including a third cluster those more open to the idea of PEW split into two groups. Interpreting the difference between these two clusters is more challenging but appears to be mostly determined by one cluster having fewer concerns (cluster 2) about establishing PEW than the other (cluster 3).

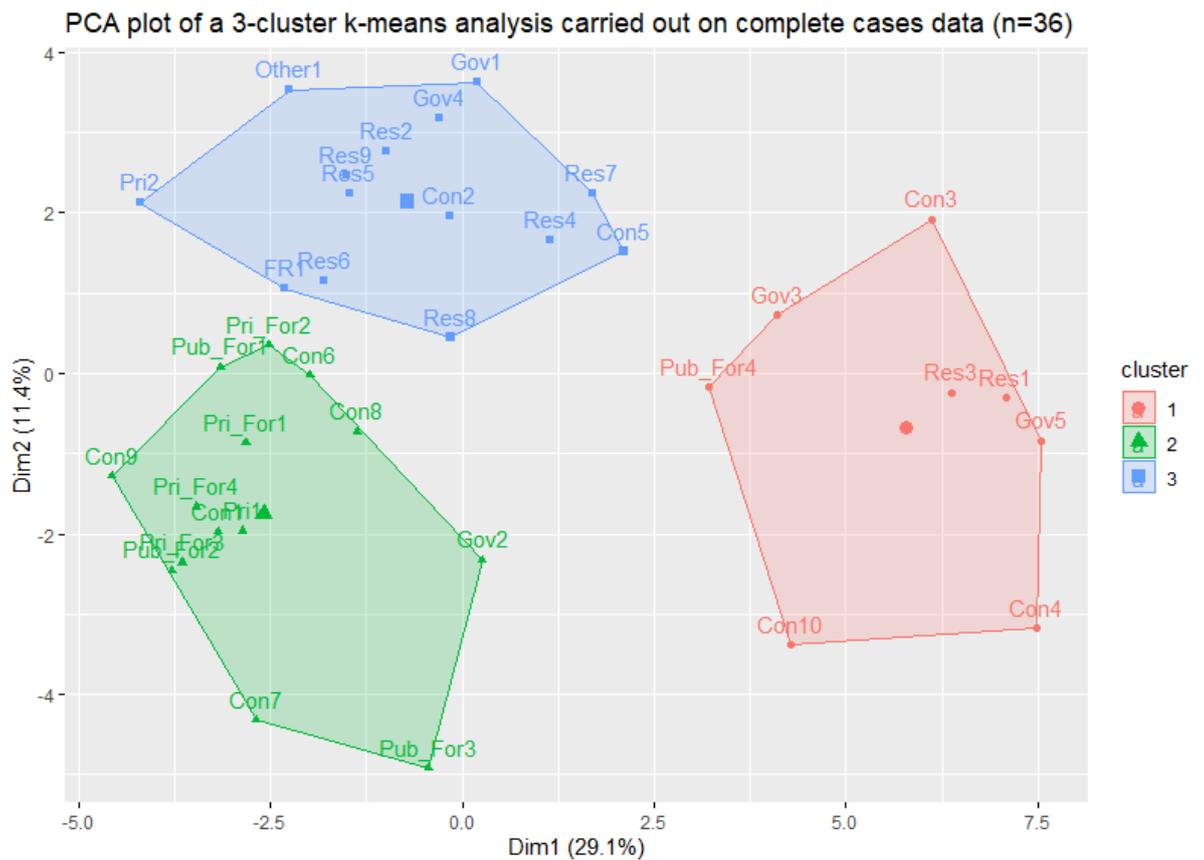


Figure 2.8: The results of a 3-cluster k-means analysis displayed on a PCA plot. This figure used all variables for complete cases only (participants who had answered every question in the quantitative survey). Each point represents a participant with the following codes indicating stakeholder group: Con = conservation, Res = Research, FR=Forest Research, Pub_For = Public Forestry, Pri_For = Private forestry, Gov = Other Government or Statutory body, Pri = Other private, Other = Other stakeholder grouping.

The multivariate analyses highlighted some clustering of members of the same stakeholder group. Those working in public and private forestry are mostly in cluster 2 (least concerns towards PEW) and those working in research are mostly in cluster 3 (more concerns towards PEW). Those working in conservation show the least clustering and therefore the greatest diversity of opinion. Although there is high variability in opinion towards PEW overall, those working in public and private forestry are significantly more likely to think PEW is a good management option for more sites than the combined sample of the other stakeholder groups (Mann-Whitney U, $p=0.04$).

2.6. Discussion

2.6.1. Awareness

The idea of PEW has been reasonably well disseminated around different relevant stakeholder groups. The group sampled in this study - stakeholders with a professional interest in the management of afforested peatlands in Scotland - is relatively small and the concept is likely less well known outside this group. The comparatively greater awareness of the concept of PEW in the forestry sector, especially the Forestry Commission, is interpreted as being due to the PEW guidelines originating within the Forestry Commission.

2.6.2. Motivations and concerns

Individual's opinion of peatlands and their management has previously been identified as complex and multi-faceted in Scotland and internationally (Byg et al., 2017, 2020; Schulz et al., 2019). An element of this complexity is balancing competing peatland land uses and uncertainty of management outcomes which has been historically understudied (Anderson, 2014; Byg et al., 2017). Decisions about ecological restoration, especially in complex settings, require value judgements to be made (Davis and Slobodkin, 2004). Hertog and Turnhout (2018) identified that those working on ecological restoration projects in the Netherlands balanced idealism and pragmatism when managing projects. The results of this study can be interpreted similarly with

stakeholder opinions on PEW being driven by a combination of participant's idealistic preferences and pragmatic considerations.

Nature is a concept that is widely recognised but has highly variable meanings to different people, in particular the extent to which humans and human activity are part of or separate from nature (Bingham and Hinchliffe, 2008; Ducarme and Couvet, 2020). Those completely opposed to establishing PEW were concerned about PEW being unnatural/artificial and not meaningfully analogous to any natural habitat. Biodiversity enhancement has been identified as a main driver for ecological restoration in this and other studies (Hagger et al., 2017). However, the view of PEW as unnatural can be a barrier to thinking PEW could enhance biodiversity as what it does support could be regarded as an inappropriate degraded state present on a site which was formerly a priority open habitat for its locations. In contrast, those who support the establishment of PEW focus on the ecological and functional benefits that the habitat might have. In particular, stakeholders generally focused on how PEW may create a low-density native edge woodland, a scarce ecotonal habitat supporting a distinct flora and fauna (Gilbert and Di Cosmo, 2003). The emphasis on what the potential benefits of different restoration options are rather than a focus on what is historically faithful is an important component of the idea of novel ecosystems (Hobbs et al., 2009; Evers et al., 2018). Whether a site is seen as being in a degraded state or just a different state depends heavily of the value-laden judgements of individual participants (Hobbs, 2016).

In contrast to those opposed to PEW, many of those advocating for PEW saw it as recreating a more natural landscape. Palaeoecological studies provide evidence that historically, trees were more prevalent in peatland dominated landscapes (Sybenga, 2020). Furthermore, the prevalence of woodland growing on Scottish peatland has been observed to vary through time with colder, wetter periods favouring open bog expansion and warmer, drier periods facilitating some tree colonisation of peatland areas (Tipping et al., 2008; Moir et al., 2010). Future climate projections for Scotland predict average temperature increases and drier summers (Werritty and Sugden, 2012) and so could favour wooded peatlands. Restoration and in particular the novel ecosystem concept does put a focus on creating habitats that are suitable for current and future climates (Hobbs et al., 2009).

Perception about restorability of afforested peatland is an important influence on participants' opinions of PEW and can be seen as a balance between a participant's idealism about nature and pragmatic judgement. Forest-to-bog restoration projects aim to set peatlands on a trajectory back to a near-natural state, which is a long-term process, and the majority of

projects have been started within the last 25 years with restoration methods still being actively developed (Andersen et al., 2017). The limits to how successful forest-to-bog restoration can be at achieving different aims under different conditions is therefore open to interpretation. Ultimately whether or not a site is considered restorable to open bog will depend on the objectives of the project and the financial constraints. Those most opposed to PEW saw an ecological moral duty to restore the pre-disturbance 'natural' open peatland and so would be inclined to pursue restoration regardless of the uncertainty around the timescale or cost of the project. These participants generally perceived other land managers who established PEW in place of open bog restoration as failing to meet the ecological priority for the site in favour of a cheaper less intensive option that maintained inappropriate forest cover (how the costs of PEW may compare to forest-to-bog restoration will be discussed in the next section, 'Establishment'). In this mindset failing to restore a site to open bog is akin to abandoning the site. Of those with more positive attitudes toward PEW there were some who saw establishing PEW at some sites as an ecological duty if they believed that PEW would be recreating habitats historically lost from the landscape. There were also those who saw the financial/social duty to provide the best benefits for a given investment and believed that PEW would achieve this on some sites.

How much intervention should be implemented as part of ecological restoration and conservation is complex (Steinwall, 2015). In this study participants generally viewed as positive reducing the interventions a site required but believed this could be achieved in different ways. Some participants thought PEW as an unnatural habitat would not be stable and would require more interventions to sustain whereas open-bog restoration was the best option to produce a self-sustaining habitat. These concerns are validated by observations of how trees colonise many degraded peatlands (Lavoie and Rochefort, 1996; Lunt et al., 2010) and that the establishment of trees on peatlands is associated with progressive declines in *Sphagnum* spp. coverage (Talbot et al., 2010) and water table depth (Fay and Lavoie, 2009). The ultimate concern is that the progressive drying of the peatland would ultimately convert the site to a closed canopy woodland, there are many examples of well-established native woodland on former open peatlands (Anderson et al., 2014). To an extent these concerns were shared by some participants more open to the idea of PEW but many suggested that at the sites where they would establish PEW the trees wouldn't grow vigorously enough to completely outcompete the peatland flora or substantially reduce the water table depth. There are examples that support this possibility where trees such as Scots pine have colonised peatlands, often after degradation, but have formed relatively stable bog woodland that co-exists with peatland vegetation in some cases over 100s of years (Wells, 2001), although an understanding of why some stable bog woodlands establish is

not well understood (Anderson and Harding, 2002). Some participants saw PEW as a more natural management option accepting tree regeneration as a way to reduce the number of interventions required in managing a site in contrast to some open-bog restoration projects where tree regeneration eventually requires secondary clearing of trees (Anderson and Peace, 2017)

2.6.3. Establishment

Participants expressed a range of ideas on where PEW should be established, what PEW's structure and composition should be and how PEW should be established and maintained. These reflected the participants motivations and concerns for establishing PEW. Different participants interpretations of what structure and composition PEW should have can be seen as creating a spectrum of systems between forest-to-bog restoration and conventional restocking with PEW of differing tree densities in between (table 2.1). Table 2.1 highlights that PEW can range from 'low intensity' to 'high intensity'.

Low intensity PEW, which is envisaged as having a low density of native trees in hydrologically restored peatlands, describes a system functionally more similar to a restored open bog than a woodland. In low intensity PEW minimising loss of carbon from the peat is a priority and open bog flora is dominant. In table 2.1 this type of PEW is referred to as 'low intensity' PEW. Low intensity PEW resembles natural bog woodlands so PEW designed according to these views can be seen as a forgotten natural state. This view of PEW is therefore akin to traditional restoration with a sense of historical fidelity, albeit for a disputed historic state.

Interpretations of PEW involving a higher density of native trees with a non-native element would be functionally closer to a restocked plantation, e.g. the site would be managed with some consideration of reducing carbon losses from the peat but also to promote carbon sequestration in the trees to mitigate peat carbon losses. The flora and fauna in this interpretation would be closer to what would be expected in a woodland than an open bog. In table 2.1 this type of PEW is referred to as 'high intensity' PEW. High intensity PEW resembles a more novel ecosystem as its biodiversity and the ecosystem services it provides would be altered from its natural state.

Trade-offs between the participants' ideals about nature and pragmatism can be identified in the low and high intensity PEW models. Low intensity PEW can be seen as being more idealistically driven as those advocating for its establishment made the interpretation that it

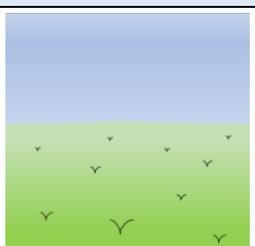
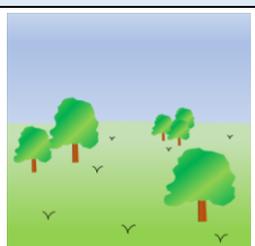
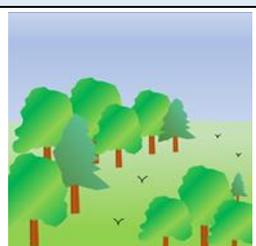
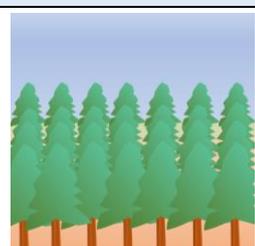
was a habitat which should naturally be more prevalent in the landscapes being managed. Establishment of high intensity PEW can be viewed as having a pragmatic influence as it does not attempt to completely reverse past degradation but instead tries to improve the state of the ecosystem within constraints. Trade-offs between the participants' ideals about nature and pragmatism can also be identified in the two different location models, the ecotone and pragmatic model described in the results (figure 2.7). The ecotone model is strongly rooted in a sense of recreating a habitat which has a strong ecological basis. However, the pragmatic model accepts constraints on a site and works within these.

There is scientific uncertainty over the viability of PEW and whether it could provide the intended ecosystem services (e.g. net carbon sequestration). Studies on native woodland established on peaty soils in Scotland have shown that trees increase the rate of respiration in the soil, thus increasing CO₂ losses (Fruggens et al., 2020; Mazzola et al., 2022) and that tree growth had not compensated for this loss (Fruggens et al., 2020). These studies also show that CH₄ emissions decrease as a result of the influence of the trees (Mazzola et al., 2022). Neither of these studies accounted for all the major pathways for greenhouse gas emissions and were on sites which had been as heavily disturbed as would be peatlands which had been productively afforested, so there is no direct scientific evidence for the impact of PEW. Regardless of whether tree growth can compensate for losses of carbon from the soil in the short term, when longer-term timescales are considered, carbon stored below ground in peat may represent a much more stable long term carbon store than carbon stored in wood (Hargreaves et al., 2003; Hermans et al., 2019). Although below-ground carbon losses may initially be compensated for by sequestration in the growth of trees, eventually carbon released from dead and decaying older trees or their products will negate this sink while carbon losses from the peat may continue in perpetuity.

How expensive forest-to-bog restoration would be compared to PEW creation is hard to estimate with forest-to-bog restoration costs already highly variable between projects (Artz et al., 2018; Okumah et al., 2019; Glenk et al., 2022). As a general guide a recent Scottish study found the mean cost of a forest-to-bog restoration project to be £3003.92/ha (Glenk et al., 2022). A comparison between the two methods would depend on a range of factors including how each management option would be established. A low intensity PEW would be likely to use relatively intensive actions to rewet the site, similarly to practice in forest-to-bog restoration projects, as this is important for reducing oxidative losses from the peat and to prevent excessive tree growth and regeneration. Conversely high intensity PEW aims to encourage tree growth relative to low intensity PEW so would be likely to use less expensive rewetting techniques, the rationale being

that the dryer conditions would promote enough tree growth to compensate for higher oxidative CO₂ losses from the peat. Another major component of the cost in PEW and forest-to-bog projects would be the felling and removal of the previous tree crop; a range of methods are used in forest-to-bog restoration depending on the project. It is plausible that PEW creation projects, especially higher intensity ones, may be more likely to use the cheaper techniques such as partial tree removal rather than whole tree removal, as there may be less concern about leaving some remains of the crop on site. PEW would have additional costs if trees were established through planting rather than regeneration. Another cost would be presented if trees needed to be replanted or thinned as part of PEW management. The large uncertainties in the costs of the various management actions in combination with different opinions of what management actions would be required means it is very difficult to estimate the relative cost of PEW creation versus forest-to-bog restoration. This uncertainty and variability is reflected in the range of opinions held by stakeholders on the costings of PEW.

Table 2.1: A framework showing how different ideas about PEW relate to forest-to-bog restoration and conventional restocking in terms of the structure, composition, and objectives of these systems. Whether or not the ground flora vegetation or strategy for maximising carbon storage would be achieved in reality is not certain. Due to scientific uncertainty and the lack of case studies for PEW systems the table can only show the intention for each habitat type.

Character Spectrum	Increasing: Tree cover Proportion non-native tree cover			
				
Type	Conventional Forest-to-bog restoration	Low Intensity PEW: Forgotten Natural State Restoration	High intensity PEW: Novel Ecosystem Restoration	Conventional Commercial Restocking
Visual rendering				
Intention of management approach	Restore to historic open state	Restore to a state considered to have once been more prevalent	Restore to a system without an obvious analogue	Generate timber/ sequester carbon in timber
Level of drainage	Drainage fully remediated	Drainage fully or partially remediated	Drainage partially or not remediated	Drainage not remediated
Native tree cover	No trees	Low density of native trees	Relatively high density of native and non-native trees	High density of predominantly non-native trees
Non-native tree cover	No non-native trees allowed	No non-native trees allowed	Non-native trees might be a component of tree cover	Typically dominated by non-native trees
Ground flora vegetation	Peatland vegetation well restored	Peatland vegetation well restored	A mix of peatland and woodland vegetation	Closed canopy woodland limits growth of vegetation
Strategy for maximising carbon storage	Prevent carbon losses from peat carbon store	Prevent carbon losses from peat carbon store	Reduce carbon losses from peat carbon store and compensate losses with growth of trees	Compensate carbon losses from the peat carbon store with tree growth

2.6.4. Stakeholder comparison

The conservation NGO stakeholder group showed the most variation in opinion on PEW. This might reflect a general greater diversity of perspective in the conservation NGO stakeholder group. Although it is beyond the scope of this study to thoroughly assess differences within stakeholder group the qualitative survey highlighted that those working in conservation operated on a range of scales including small scale/low budget projects, large scale/high budget projects and campaigning/advocacy. Working at different scales has previously been identified as indicating different motivations and approaches to ecological restoration (Clewell and Aronson, 2006). In addition, different historic backgrounds to the conservation NGOs may have affected an organisation's culture (Odor, 2018), for example a conservation NGO with a history of campaigning to protect large expanses of open peatland areas from forestry (Stroud et al., 1987) may be wary of creating a new form of woodland on peatland, while there are many other conservation NGOs without this background which therefore may have a culture more open to ideas about woodland on peatlands. In contrast, the relatively low diversity of opinion towards PEW within stakeholder groups such as forestry and research may be due to participants working in a similar context.

Determining the composition of a fully representative sample of stakeholder groups involved in afforested peatlands is challenging. However, the quantitative stage captures a diverse set of opinion with reasonable representation from all the major stakeholder groups. More respondents to the quantitative survey would improve confidence in the results, especially in stakeholder group comparisons. There may have been a bias if those with more knowledge of PEW were more likely to participate in the survey. There may also have been an effect of involving participants as individuals rather than approaching organisations for an official response. However, the corroboration of the results from the qualitative and quantitative stage samples which were recruited in different ways, is evidence that the findings are valid.

While identifying stakeholder groups it was clear that PEW might be of interest to community forestry, something that was also suggested by some of the participants at the qualitative stage. Unfortunately, the small scale of community forestry projects made relevant groups difficult to identify and recruit. The focus of this study has therefore been on stakeholder groups that typically operate at larger scales.

2.7. Conclusion

The study has shown the existence of a diverse range of opinion about PEW within those with a professional interest in Scottish afforested peatlands. Differences in participants ideological beliefs about what states of peatlands can be considered natural and what the objectives for land management should be along with pragmatic considerations underpin motivations and concerns for establishing PEW. This in turn affects their approach to if, where, and how PEW should be established.

Conceptions of PEW vary; those who were opposed to PEW reject it on the grounds that it could allow site abandonment after forestry without consideration of how a site could best be managed. They also reject the idea that PEW could ever provide better ecosystem services (e.g., more carbon storage) than either forest-to-bog restoration or conventional restocking. Some of those considering establishing PEW described it in terms of an arguably forgotten or lost state for the landscape (low intensity PEW) while others described it in a way more closely akin to a novel ecosystem which had a different biodiversity and ecological functions than the previous open peatland (high intensity PEW). There would be a clear benefit for greater communication between and within stakeholder groups to improve understanding of different perspectives and approaches on PEW and peatland management in general.

This study has highlighted that PEW is being created in a range of different ways for a range of different objectives. The variability this engenders will be a challenge for future research and guidance that attempts to give a generalised understanding of PEW. This is an existing problem for research on forest-to-bog restoration or restocking as all field sites have a unique context and management history (Andersen et al., 2017). However, there are existing well recognised terms (e.g. fell-to-waste, mulching, mounding etc) which help contextualise more precisely the type of management a field site has received undergoing restocking or forest-to-bog restoration. The recency of PEW means that there has not been a specific vocabulary developed to refer to different PEW management approaches. The framework this study puts forward of PEW management options existing on a spectrum between conventional restocking and open bog restoration with some PEW projects being 'high intensity' and others being 'low intensity' may help contextualise research and guidance as it provides a way for specific case studies to be placed on a scale relative to other projects.

The study has highlighted substantial disagreement between different stakeholders with a professional interest in afforested peatland management as to the short/long term viability of

PEW and the benefits or problems its creation might cause. There is a clear need for further research into the outcomes of PEW creation to inform the debate on PEW and guide when and how PEW is established.

It remains to be seen if PEW will be established on a large scale or what form or forms this will take but already the notion of PEW is influencing how many stakeholders plan and conceptualise the management of their sites. Forestry Commission Scotland (now Scottish Forestry) deliberately limited detail in its specification for PEW, stating that the guidance may be refined by experience of practitioners. The current flexibility of PEW as a concept matches the current uncertainties of what role PEW might fill and the benefits it may yield.

Linking Chapter 2 and Chapter 3

The last chapter found that different stakeholders have adopted different positions on PEW, both positive and negative, in the absence of substantial scientific evidence to support these stances. There is a need for greater scientific certainty to inform the discussion on PEW. In this next chapter, evidence is provided on the carbon storage capacity of different potential types of PEW. Although the field site for this chapter was established ~20 years before the concept of PEW was developed, it resembles 'high intensity' PEW as defined in the previous chapter. The aim of this following chapter is to provide evidence of how modern PEW woodland may develop and what impacts it may have on carbon storage.

3. Carbon Storage and Habitat Potential of Native Woodland as an Alternative to Continued Non-Native Commercial Forestry on Deep Peat

3.1. Abstract

During the second half of the 20th century ~20% of deep peat in Scotland was planted with commercial forestry plantations resulting in considerable carbon release and habitat degradation. The expansion of this forestry is no longer permitted and large areas are being actively restored to open peatland. However, many sites are not deemed suitable for restoration yet have limited viability for continued productive use. For such sites an alternative management option has recently been suggested, termed Peatland Edge Woodland (PEW) where a predominantly native woodland could be established at a low density and with low intensity cultivation (e.g. no or limited artificial drainage). It is reasoned that this type of management will provide some of the carbon capture benefits associated with tree growth while limiting soil degradation and supporting more diverse ecology compared to intensive non-native commercial woodland. This study investigates whether PEW could provide carbon benefits by comparing carbon storage in second rotation deep peat non-native commercial stands and adjacent native woodland (analogous to PEW) and investigating what impact it may have on carbon storage in the peat. Native woodland has established and stores carbon, however non-native commercial woodland treatments were considerably more effective at sequestering carbon and had a carbon density on average 6.8 times greater than native woodland treatments. There is some evidence that a trade-off may exist with the lower carbon storage in native woodland types been associated with higher plot water tables which could mean lower CO₂ emissions from below ground, although no direct measurements of this were taken. The study demonstrates that PEW may have some benefits over alternative management options, but more work is required, in particular on the effect of PEW on below ground carbon storage, to better quantify trade-offs between different environmental impacts.

3.2. Introduction

In near-natural conditions peatlands support a specialist biota and provide a wealth of ecosystem services including water purification, fire prevention and carbon storage (Joosten and Clarke, 2002; Parish et al., 2008; Bonn et al., 2014). Peatlands form when primary productivity exceeds decay so that carbon rich organic material accumulates forming peat soils (Clymo, 1978). Carbon accumulation is slow relative to many other vegetation types such as forest but has long term stability so that over thousands of years, peatlands can accumulate several metres of carbon rich peat (Frolking et al., 1998). As a result, despite covering less than 3% of the Earth's land surface, peatlands store twice the carbon than is stored in all the world's forest biomass (Parish et al., 2008). The conditions under which peat accumulates are usually acidic, nutrient poor and waterlogged, and were traditionally perceived to have inherently limited value for conventional land uses. Historically, particularly in Western Europe, peatland have been cut to provide fuel or drained and fertilised to make them more productive for forestry and agriculture (Verhoeven, 2014) but these practices result in declines in the natural ecosystem services provided (IUCN, 2018b). Peat layers that are not waterlogged are susceptible to oxidative decay which allows carbon stored in the peat to be released as carbon dioxide (CO₂) (Sloan et al., 2018b; Leifeld et al., 2019).

Peatland restoration to pre-disturbance conditions is receiving increasing attention and investment as peatland ecosystem services become more recognised and valued (Leifeld and Menichetti, 2018; Humpenöder et al., 2020). Peatland restoration has been shown by ecosystem service valuation methods to be an economically prudent activity in terms of the value of the benefits generated for society (Moxey and Moran, 2014; Horsburgh et al., 2022). Peatland areas are important contributors to the greenhouse gas balance of Land Use, Land-Use Change and Forestry (LULUCF). With LULUCF counting towards national greenhouse gas inventories there is impetus for improved management of peatlands in order to achieve emission reduction targets. For example, the UK's Nationally Determined Contributions target, which is to reduce greenhouse gas emissions by two thirds on a 1990 baseline by 2030 and ultimately reach net zero greenhouse gas emissions by 2050 (Committee on Climate Change, 2020).

A major component of many restoration projects is to 'rewet' the site (raise the water table) which can reduce oxidative CO₂ emissions (Günther et al., 2020; Evans et al., 2021). Rewetting also typically increases methane (CH₄) emissions which is a greenhouse gas with a warming potential approximately 28 times that of CO₂ (Myhre et al., 2013) but despite this

offsetting effect rewetting has been identified as an important climate change mitigation approach for many peatlands (Günther et al., 2020; Evans et al., 2021).

However, although peatland restoration is receiving considerable and increasing investment, even the most recent Scottish government target to invest in the restoration of 250,000 ha of peatlands by 2030 equates to the restoration of just 17% of degraded peatland area (Horsburgh et al., 2022). Land managers do not always deem the restoration of an area of peatland and its ecosystem services (such as carbon sequestration and storage) as financially or practically viable (Shepherd et al., 2013; Anderson, 2014; Forestry Commission Scotland, 2015; Were et al., 2019). In instances where peatlands have not been deemed viable for restoration yet continued commercial use is unprofitable, they have sometimes been converted to different ecosystems such as lakes, grassland or native woodlands, which have the potential to provide new environmental and economic benefits (Farrell and Doyle, 2003; Higgins and Colleran, 2006; Kasimir et al., 2018). Kasimir et al. (2018) highlights an example in Sweden when the combined economic and carbon benefits of restoring commercially afforested peatlands to willow woodland or reed canary grass with partially restored water tables was greater than for fully rewetted peatlands. There is also uncertainty over whether climate change may make sustaining some peatland areas in their current state inviable in the long term (Ferretto et al., 2019). Novel habitat creation may potentially become more important in restoration work as a result of climate change (Hobbs, 2016) but it is not easy to predict how peatland restoration will be ultimately affected by this.

During the second half of the 20th century large areas of deep peat in Scotland were drained, enriched with fertiliser and planted with commercial non-native forestry plantation (Sloan et al., 2018b), deep peat forestry now covers an estimated 17% of Scottish peatlands (Vanguelova et al., 2018). The establishment of these plantations was ecologically highly damaging (Stroud et al., 1987; Lindsay et al., 2014c) and the difficulty of productively afforesting deep peat sites meant that a lot of the afforestation was only economically viable due to substantial government subsidies and tax breaks (Stroud et al., 2015). The expansion of forestry on deep peatland (defined as greater than 50 cm deep) is no longer permitted in Scotland (Forestry Commission Scotland, 2015; Forestry Commission, 2017) but decisions need to be made about what should be done with existing plantations after their current crop is harvested.

Forestry Commission Scotland has created guidance which lays out ecological criteria for when an afforested peatland site should be restored to open bog at the end of its rotation. If these criteria are not met then the site can be considered for conventional restocking if a good

growth rate can be achieved with minimal intervention (Forestry Commission Scotland, 2015). The guidelines state that if a good growth rate can be achieved then carbon sequestration in the trees will equal or exceed the carbon lost from the peat carbon store due to the disturbances of replanting and the artificially lowered water table, as suggested by some UK based studies (Morison et al., 2010; Lilly et al., 2016; Vanguelova et al., 2018).

Although it is established as forestry policy that afforested peatlands are net sinks when the growth rate of the trees is high enough, some of the models used to conclude this have been questioned (IUCN, 2014). Other work done to develop tier 2 emission factors, for the UK greenhouse gas inventory, estimated that carbon lost from afforested peatland across the whole of the UK was 4600 kt CO₂e yr⁻¹ (Evans et al., 2017): approximately 1% of annual UK emissions. Furthermore, substituting carbon lost from peat for carbon stored in timber will be unlikely to remain a net sink of carbon in the long term because unlike carbon stored below ground in active peatland which might have the capacity to stay stored for 1000s of years, carbon in timber and wood products is much more vulnerable to being released to the atmosphere across shorter time scales (Hargreaves et al., 2003; Lindsay et al., 2014c; Hermans et al., 2019).

The guidelines suggest that in the situation where a site is neither a good candidate for restocking nor forest-to-bog restoration, a low density, low intensity native woodland should be established, termed 'Peatland Edge Woodland' (PEW) (Forestry Commission Scotland, 2015). The guidelines proposed that on appropriate sites this will provide the best combination of biodiversity, landscape, and carbon benefits possible for the site. PEW was envisaged as a low-density woodland that would allow some recovery of the peatland, thus reducing oxidative CO₂ losses, while offsetting any CO₂ losses with CO₂ sequestered in the growth of trees (Forestry Commission Scotland, 2015). This can therefore be seen as an "intermediate option" trying to combine benefits of both forest-to-bog restoration and conventional restocking. However, there is no direct scientific evidence underpinning this concept and its opponents have suggested that PEW would in fact be ineffective in providing benefits (RSPB, 2014). It is unclear how PEW would trade-off against conventional restocking or forest-to-bog restoration in terms of greenhouse gas balance and other ecosystem services such as biodiversity, water quality, water regulation, well-being, provision of timber.

The aim of this study is to compare the effect of establishing native woodland plots (analogous to PEW) compared to continued conventional non-native forestry on deep peat. More specifically, the study's objectives were to: 1) Compare the relative capacity of native woodland to store carbon compared to non-native woodland. The hypothesis is that non-native woodland will

be able to store more carbon in tree biomass than native woodland as the non-native tree species have selected for their relative vigour in peatland plantations. 2) Determine the effect different tree planting treatments and biomasses have on proxy measures of peat carbon storage condition. The hypothesis is that the proxy measures (tea bag decay and stabilisation rates and water table depth) will indicate higher peat decomposition with higher tree biomass since it is reasoned that plots with lower tree biomass should experience lower impacts of tree growth.

3.3. Methodology

3.3.1. Study site

This study made use of a long-term experiment established by the Forestry Commission in 1995 within Rumster Forest, an afforested area of deep peat former blanket bog located in Caithness, northeast Scotland, at approximately 58°19'53.75"N, 3°20'31.38"W. (figure 3.1a). This long-term experiment was originally established to investigate the growth rate of different native and non-native tree species planted on deep peat after a complete first rotation of a commercial crop species – in this case lodgepole pine (*Pinus contorta*). The trees were established with low intensity methods, e.g., planting on small mounds and no new drainage with minimal fertiliser use of 650 kg/ha of potassium/phosphate fertiliser added in 2008 after 12 years of growth, as is comparable to current guidance for deep peat forestry (Forestry Commission Scotland, 2015). The experiment consists of small ~18 x ~18 metre plots with each species mix treatment replicated in three plots structured into three blocks. Variation in the natural conditions of the forest plots were evident, e.g., some areas appeared wetter and peat depth varied across the site. It is unknown how the location or layout of the three blocks was originally determined, but each plot does roughly correspond to a different edge of the forestry coupe (figure 3.1c). There were no statistically significant differences in peat depth, water table depth or soil surface volumetric water content between the blocks.

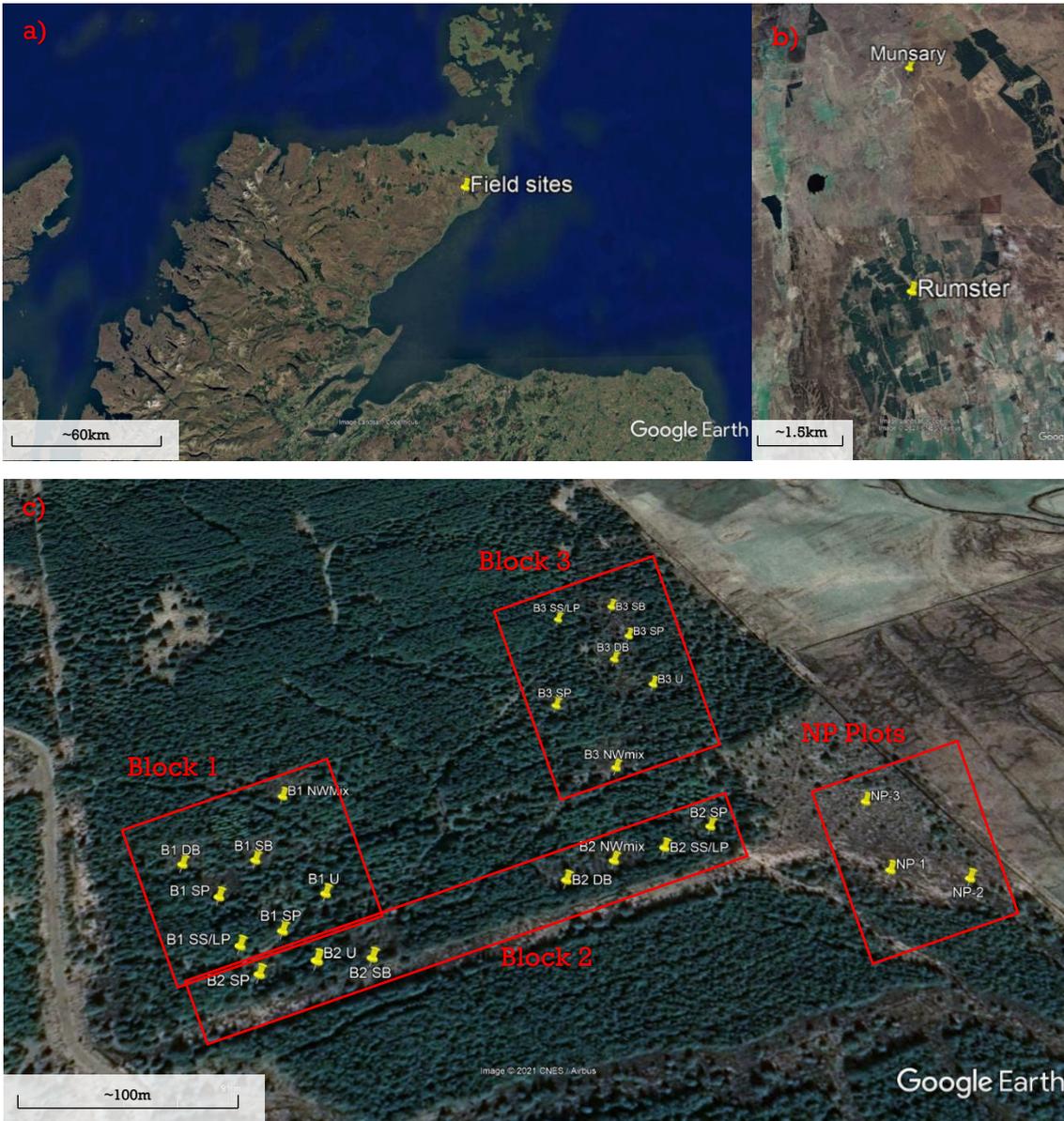


Figure 3.1: Map of field site location and layout using satellite imagery from Google Earth Pro ([Google Earth](#)). a) Shows field site position in the northeast of Scotland. b) Shows the approximate 7 km distance between the never planted plots at Munsary and the rest of the plots at Rumster. c) Shows the layout of the experimental plots at Rumster with each point representing a different plot. Each point is labelled with the block number the plot is in and the abbreviated treatment of the plot which are listed in table 3.1.

Our study investigated a total 27 plots, three replicates of nine treatments. Each treatment was assigned to one of three categories: 1) native woodland (PEW analogue) treatments 2) non-native woodland (conventional restocking) treatments, 3) unafforested blanket bog controls. The unafforested controls are not part of the original forestry experiment so do not conform to the other plots' block design. The details of the composition of the tree planting and management history of the nine treatments can be seen in table 3.1.

The plots consist of surviving planted treatment trees and regeneration (i.e. growth of self-seeded trees) of Sitka spruce and lodgepole pine. Regeneration was cleared in 2008 except in the self-seeded treatment plot where regeneration has been allowed to develop. Despite the clearance, regeneration is still a prominent feature in some of the plots. The success of the planted trees is variable within and between plots with some plots experiencing high mortality rates with surviving trees being very stunted (<1 m tall) while other trees grew better (the tallest tree was 10.8 m). In general, the non-native plots have high survival and growth rates than the native-woodland plots.

Field work was carried out on the site from July 2018 to March 2019. For each plot the carbon storage has been estimated for the trees, ground flora and below ground carbon store. Different proxies for peat decay rate were measured for all plots and vegetation surveys were carried out. These methods are described in more detail in the next sections.

3.3.2. Carbon storage in trees

Non-native plot yield classes

The yield class of each non-native woodland plot was estimated using Forest Yield (Mathews et al., 2016) in order to contextualise the productivity of the plots. Forest Yield is a program that is widely used by Scottish foresters to estimate yield class sizes based on the measurements of stand top heights. Forest Yield was used to estimate the general yield class of the Sitka spruce in the two conventionally restocked treatments using the height of the tree with the largest measured DBH in each plot.

Table 3.1: The details of the composition of the tree planting and management history of the nine treatments.

Category	Details of the original tree planting in 1995 and management history(including treatment code)	Location	Survival rate of the trees in each plot by 2019
Native Woodland (PEW analogue) treatment	NWmix = native woodland mix Planted with a mix 60% downy birch (<i>Betula pubescens</i>), 10% rowan (<i>Sorbus aucuparia</i>), 30% willow species (10% <i>Salix aurita</i> , 10% <i>Salix caprea</i> , 10% <i>Salix cinerea</i>)	Block 1, 2 & 3 – Rumster Forest (figure 3.1c)	Block 1: 75% Block 2: 64% Block 3: 63%
	DB = downy birch Planted with 100% downy birch	Block 1, 2 & 3 – Rumster Forest (figure 3.1c)	Block 1: 41% Block 2: 73% Block 3: 34%
	SB = silver birch (<i>Betula pendula</i>) Planted with 100% silver birch	Block 1, 2 & 3 – Rumster Forest (figure 3.1c)	Block 1: 44% Block 2: 64% Block 3: 41%
	SP1 = Scots pine (<i>Pinus sylvestris</i>) Planted with 100% Scots pine	Within Rumster Forest experiment	Block 1: 6% Block 2: 11% Block 3: 9%
	SP2 = Scots pine Planted with 100% Scots pine	Block 1, 2 & 3 – Rumster Forest (figure 3.1c)	Block 1: 25% Block 2: 36% Block 3: 14%
Non-native (conventional restock) treatment	SS/LP = Sitka spruce (<i>Picea sitchensis</i>)/lodgepole pine (<i>Pinus contorta</i>) Planted with 50% Sitka spruce, 50% lodgepole pine	Block 1, 2 & 3 – Rumster Forest (figure 3.1c)	Block 1: 97% Block 2: 98% Block 3: 100%
	SS = self-seeded Plots left unplanted but self-seeded trees have established a canopy dominated by Sitka spruce and lodgepole pine	Block 1, 2 & 3 – Rumster Forest (figure 3.1c)	Block 1: Na Block 2: Na Block 3: Na
Open peatland control	NP = never planted Mostly treeless plots on an area of peatland adjacent to the forest which has never been ploughed or planted with trees	NP Plots – Rumster Forest (figure 3.1c)	Na
	NN = near natural Peatland in near natural condition	Munsary Nature Reserve (figure 3.1b) ~7km away from Rumster Forest	Na

Tree measuring and destructive sampling

To more precisely estimate the carbon stored above-ground in the trees in each plot, measurements were taken from a randomly selected 25% subsample of all the trees planted and the number of planted trees that were still alive was recorded. In addition, every self-seeded tree taller than breast height (140 cm) in each plot was measured. The measurements taken were: diameter at Breast Height (DBH), trunk basal diameter (BD) - defined as 15 cm off the ground - and total tree height from base (H).

Several trees had multiple trunks growing from their base (within 15 cm of the soil surface), these were treated as separate trees and measured separately. Some of the conifer trees split into two or more roughly equally sized stems part way up the stem. In these cases, if the split occurred below breast height each stem was measured separately but were treated as the same tree if the split occurred above breast height. Some trees were only just above breast height, in which cases the DBH measurement was necessarily taken on the widest branch in the canopy at breast height. The willow species growing on the site were generally low growing with numerous widely spreading stems; therefore a pair of canopy cover measurements (north-south and east-west) were taken in place of stem diameter measurements.

To build site specific allometric equations a number of individual trees (Sitka spruce (*Picea sitchensis*) n=19, lodgepole pine (*Pinus contorta*) n=15, Scots pine (*Pinus sylvestris*) n=12, downy birch (*Betula pubescens*) n=14, silver birch (*Betula pendula*) n=12, rowan (*Sorbus aucuparia*) n=5, willow (combined *Salix aurita*, *Salix caprea* and *Salix cinerea*) n=6) for each species were felled across the size range of the trees measured. The felled trees were immediately partitioned into different components and weighed. For conifers the components were stem and branch, with the stem being further divided every 2 meters. For broadleaves where it was not easy to clearly identify a main stem relative to its side branches the tree was split into the following diameter classes > 5cm, 5-2 cm, <2 cm. The >5 cm component was split into 2-meter sections if applicable. Conifer needles were not weighed separately and were included as part of the weight for which ever component category they were attached to. Since trees were felled in winter the mass of broadleaf leaves were not accounted for by the sampling.

Once each component of a felled tree material was weighed an approximately 2-5 cm cross-section disk subsample was taken from each component. The subsamples were weighed on the same day as felling, oven dried at 60°C until there was no further weight change and reweighed. These dry weights were used to estimate the dry mass of the tree section it had been taken from by multiplying the ratio of wet weight to dry weight of the subsample by the wet

weight of whole tree component. The dry mass estimates of all the components from each individual tree were summed to give an estimate for the dry weight of the whole trees.

The proportion of wet weight to dry weight for the component of broadleaf trees with a diameter <2 cm and also the branch component of conifer trees were highly variable within an individual depending on where the section was taken. Consequently, for six trees of each species a complete branch was collected on the reasoning that a greater mass of sample would be more representative of the component and therefore give more consistent wet mass to dry mass ratios. For each species an average dry weight of these branches was calculated, and this average value for each species was used to estimate the dry mass of conifer branches and broadleaf branches <2cm in place of the wet to dry mass ratio calculated for the specific subsection taken for each tree

Carbon content analysis

A subsample of the tree component subsamples was analysed with an Elementar Ltd MACRO Cube to give the percentage of dry mass which was carbon. The analyser combusts the samples converting all carbon present to CO₂ and uses a thermal conductivity detector to measure total carbon content of the sample. To prepare samples for analysis thin (~1-2cm) cross sections were sawn off samples with diameter less than 5cm. For larger samples a clean drill bit was used to drill a hole to the centre of the sample and the wood shaving from this hole were collected. For five samples with 5 cm diameter both methods were used, and the carbon content of both samples were compared, which showed close agreement between the methods (varying within 1% carbon and with no consistent difference between methods). The samples collected by both methods were turned to a fine powder with a ball mill. The milled samples were then oven dried for 24 hours as the process can result in the dry samples absorbing some water. 0.05 g of each dried sample were then weighed out into tin foil cups. The cups were sealed, pushing the air out and rolled into a ball. Each sample was then run through the analyser. The analyser was calibrated with glutamic acid, the mass of which is 40.8% carbon. Glutamic acid was also used to allow drift correction for every 10 samples.

All the subsamples from an individual tree for each species were run through the MACRO Cube to assess the variability within an individual. The CN analysis showed carbon content to be relatively constant within and between individuals of each species. The largest variation between samples of the same species was 4.5% carbon for Scots pine with the other species much less

variable. A carbon content weighted mean was calculated for each species based on the relative weight of the component each subsample was taken from.

Additionally, five subsamples of the 2-5 cm tree component for the broadleaves and a trunk component of ~2-5 cm from each conifer were then analysed to assess the variability between species, an average for each species was also calculated. Scots pine showed the most variability between individuals with percentage carbon varying by 2.5%, the other species showed less variability of ~ 1% carbon content difference between individuals. The mean percentage carbon for each species derived from this subsampling approach and the subsampling approach described in the previous paragraph was within 0.6% carbon of each other. In light of the small variation, the author is confident in the accuracy of using the weighted mean of %C for each species as determined by the first method was used to determine the ratio of dry mass to carbon for the trees.

Allometric equations

The data collected from the felled trees were modelled to create allometric equations estimating carbon content of a tree from tree measurement (Picard et al., 2012). Effect variables tested in these models were DBH, tree height (H), trunk basal diameter (BD) and the combined effect variable ' $DBH^2 \times H$ ' and ' $BD^2 \times H$ '. For willow the variables tested were height, canopy area (calculated by taking a mean of the canopy diameter measurements and then calculating the area of a circle with this diameter) and canopy volume (canopy area * height). Modelling was done separately for each species, except willow species which were pooled since only a total of 6 trees were available for destructive sampling across all willow species. Initial explorative analysis indicated power relationships between the response variable and effect variables and power models described the relationship well. Power models had higher R^2 values and better met the assumptions of normality and homoscedasticity relative to linear or exponential models.

Allometric equation were derived in R (R Core Team, 2021) using the linear model function to determine coefficients. Modelling was done with natural log transformed response (tree carbon content) and predictor (e.g. $BD^2 \times H$) variables so that the models generated would be power models. A model with all the possible predictor variables was first constructed. Then the stepAIC() function from the R package MASS (Venables and Ripley, 2002) was used, to carry out Akaike Information Criterion (AIC) backward stepwise model selection to produce a more simple model with fewer predictor variables. The method reduces the number of predictor variables in a

model by selecting models with lower AIC – a statistic which balances selecting models with greater likelihoods while minimising the number of explanatory variables included in the model (Akaike, 1973).

For Sitka spruce, lodgepole pine, Scots pine, downy birch and rowan, power model based on BD^2H alone had the lowest AIC values. For silver birch and willow a power model based on BD and canopy volume respectively had the lowest AIC. The selected model for each species all had R^2 values greater than 0.95 (table 3.2). The best model for each species as determined by backward stepwise AIC model selection were checked against the assumptions of equal variances with Breusch-Pagan tests and normality with Shapiro-Wilk tests which were predominantly non-significant (table 3.2).

Table 3.2: Details of the final models selected for each tree species.

Tree Species	Model	N	R^2	a coefficient	a coefficient standard error	b coefficient	b coefficient standard error	Shapiro-Wilk test	Breusch-Pagan test
Sitka spruce	Carbon = $a(BD^2H)^b$	19	0.987	-6.894	0.234	0.807	0.022	$p=0.241$	$p=0.623$
Lodgepole pine	Carbon = $a(BD^2H)^b$	15	0.966	-7.099	0.460	0.809	0.042	$p=0.178$	$p=0.009$
Scots pine	Carbon = $a(BD^2H)^b$	12	0.985	-8.508	0.361	0.909	0.036	$p=0.383$	$p=0.644$
Downy birch	Carbon = $a(BD^2H)^b$	14	0.961	-7.481	0.433	0.818	0.048	$p=0.930$	$p=0.940$
Rowan	Carbon = $a(BD^2H)^b$	5	0.982	-8.068	0.576	0.852	0.066	$p=0.986$	$p=0.172$
Silver birch	Carbon = $a(BD)^b$	11	0.941	-3.420	0.360	2.050	0.172	$p=0.034$	$p=0.526$
Willow	Carbon = $a(\text{Area} * \text{Height})^b$	6	0.985	-18.351	1.270	1.210	0.075	$p=0.3058$	$p=0.9847$

The above-ground carbon content for every tree measured in each plot was calculated using the best allometric equation for each species. Since the modelled response variable was log transformed the inverse natural log had to be taken to give above-ground carbon storage. To account for below ground carbon storage the default above ground to below ground biomass ratio determined by meta-analysis in IPCC (2003) was used. The assumption was made that the ratio of biomass to carbon stored in the below ground components of a tree was the same as calculated for the above-ground components.

Carbon density

Since a subsample of the planted trees had been measured in each plot the total carbon stored (C) in planted trees in each plot was estimated by:

$$C \text{ planted trees} = \frac{\text{Sum C in measured planted trees}}{\text{proportion of the living planted trees measured}}$$

Each treatment plot had been originally established with slightly different sizes, with a mean plot size of 322 m². A scaling factor for each plot was calculated as follows:

$$\text{Scaling Factor} = \frac{\text{Plot Area}}{322}$$

Since the number of trees planted in each plot was always 64 it was assumed that plot size will have minimal effect on tree biomass in planted trees. However, since all self-seeded trees in the plots had been measured and the number of self-seeded trees expected in a plot would be influenced by the plot size the following equation was used to estimate the total carbon stored in self-seeded trees:

$$\text{Rescaled C Selfseeded} = C \text{ Selfseeded} * \text{Scaling Factor}$$

The estimate for carbon stored in all trees growing in a plot was calculated by:

$$C \text{ in all trees} = C \text{ selfseeded trees} + C \text{ in planted trees}$$

Finally, the scaled plot estimates were converted to carbon stored per hectare by multiplying the estimate of carbon stored in each 322m² plot by 31.06.

Carbon storage in vascular ground flora

The carbon stored in vascular vegetation other than the trees (excluding trees greater than 1.4 m) was also estimated in case this was a significant variable between treatments. Vascular vegetation was collected in 20 cm X 20 cm quadrats at three randomly selected coordinates in each plot. The samples were oven dried at 60°C until a constant temperature was reached, each sample was then split into its constituent species and weighed. For each species all the material was taken from three randomly selected quadrats where that species was present.

The vegetation samples were prepared for carbon content analysis in the same way as tree wood samples with the samples being milled, re-oven dried, and 0.05 g weighed into tin foil cups for analysis in a Elementar Ltd MACRO Cube. The mean carbon content and standard error

was very consistent within and between species except for a clear difference between woody species (Mean=51.423, SE=0.256) and herbaceous species (Mean=47.070, SE=0.317). Therefore, carbon content of all herbaceous species mass in a sample was multiplied by 47.070 and all the woody species mass was multiplied by 51.423 to give an estimate of carbon content of each quadrat. The total carbon content of all the vascular plants in the three quadrats in each plot were scaled up from the combined area of the quadrats (0.12 m²) to one hectare (10000m²) to give an estimate of carbon storage density per hectare.

3.3.3. Effect of tree growth of peat carbon store

Carbon storage in peat

An estimate was made of carbon storage below ground within the peat of each plot to contextualise the relative size of the carbon stored in the trees. Peat depth was measured from the top of the bryophyte ground flora when gently compressed to the bottom of the peat profile with a Russian corer. The Russian corer allows samples from the base of the peat profile to be taken to visually confirm when the base of the peat had been reached. This approach should provide more accurate results than peat probes as the base of peat in Rumster Forest was quite woody and mostly underlain with a soft clay, the combination of which made it difficult to judge the depth with peat poles alone (Parry et al., 2014).

One plot from each block and one plot from each the control treatments were randomly selected (n=5). The random subsample of plots was cored using an intermittent peat coring strategy developed by (Kauffman and Donato, 2012) and adapted for use on temperate and boreal peatlands by (Chimner et al., 2014). The approach splits the peat profile into sections at 0–15 cm, 15–30 cm, 30–50 cm, 50–100 cm and then every two meters or to end of the peat. For every section a 5 cm subsample was collected in the middle of the range (i.e. 5-10, 20-25, 40-45, 70-75, 195-200 cm etc). The volume of these samples was estimated using water displacement. Samples were then oven dried at 60°C until a constant temperature was reached. The dry weight and volume were used to calculate dry bulk density (BD). The carbon content of the peat samples was then analysed in the same way as for the tree wood and ground flora samples – the samples were finely ground with a ball milled, re-oven dried, 0.05 g weighed into tin foil cups for analysis in a Elementar Ltd MACRO Cube to give a percentage carbon per unit mass. The percentage carbon per unit mass were converted into a percentage carbon per 1cm³ for each sample:

$$C_{peat} 1cm^3 = \%C \times BD$$

The carbon stored in each layer was averaged across the 3 subsamples taken from the forestry experiment plots. For each plot the depth of each layer was multiplied by the carbon stored in $1cm^3$ of peat to give the carbon content for the complete profile of peat for a $1cm^2$ area. This was then scaled to one hectare to give an estimate for carbon density.

Tea Bag Index

The Tea Bag Index method (Keuskamp et al., 2013) was used to compare litter decay rates in the plots. The method uses pairs of Lipton green and rooibos tea bags to a measure of decomposition rate and litter stabilisation. In each plot, two sets of five pairs of tea bags attached to a cane at 10 cm intervals were buried 6 cm below the surface – the cane was used to make it easier to recover tea bags at the end of the experiment. For each plot, one set of tea bags was buried under the canopy of the tree closest to the vegetation monitoring plot, while a second set was buried under a gap in the canopy close to the southern edge of the vegetation plot. In both cases, the tea bags were buried under vegetation representative of the area and not on top of mounds or in plough furrows. In the two open control treatments where trees were absent the two sets of tea bags were buried under two different vegetation types. The tea bags were retrieved 90 days later.

A ML2x Theta probe connected to HH2 Moisture Meter (Delta-T Devices) was used to take five soil moisture reading immediately around the area where each set of tea bags were buried. The moisture probe was inserted into the peat vertically, so the measurement was from across the first 6 cm of peat. The moisture probe was placed in positions around the buried tea bags that captured the diversity of the vegetation/microtopography within 20 cm of the tea bags. Measurements were taken on the day the canes were buried in July and the day they were recovered in October.

Water Table Depth

Water table depth was measured in each plot with one dipwell installed on original peatland surface topography (not on plough furrows or mounds) and within 2 m of the northern edge of the vegetation monitoring plots. Measurements were taken from the dipwells three times, once in July 2018, October 2018 and March 2019. All the dipwells in one of the plots had dataloggers

installed recording water table depth every 6 hours for a 4-month from November 2018 to early March 2019 to identify difference in water table dynamics between the treatments.

3.3.5. Statistical analysis

Mean carbon storage was calculated for the different treatments in the native, non-native, and total tree cover. Statistical relationships were tested for between tree biomass, planting treatments, tree species, tea bag decay, tea bag stabilisation and water table depth by, when appropriate, comparing 95% confidence intervals and testing with ANOVAs, t-tests and Pearson's R correlation in R (R Core Team, 2021) with significant ANOVAs explored further post hoc with Tukey's pairwise comparisons. When test data was non-normally distributed it was Box-Cox transformed before ANOVA and t-tests (Box and Cox, 1964) and for correlations Spearman's rank was used. All graphs were plotted using the package ggplot2 (Wickham, 2016). The sample sizes of each treatment are small (n=3) which may reduce the potential to identify statistically significant relationships so the combined categories native woodland (SP1, SP2, NWMix, SB and DB (n=15)), non-native woodland (SS/LP, self-seeded (n=6)) and open peatland (NP and NN (n=6)) were also used. When testing within native woodland type variation a combined Scots pine (SP1, SP2 (n=6)) and combined broadleaf (NWMix, SB and DB (n=9)) were also compared.

3.4. Results

3.4.1. Carbon Storage in Vegetation

Carbon storage in trees

The two non-native conifer restock options had substantially higher mean tree carbon storage than the native woodland plots although the carbon stored in the SS/LP plot had less variation than the self-seeded plot (figure 3.2a). As a consequence of this variability the self-seeded plots did not store significantly more carbon than the native woodland plots, but the SS/LP plots store more carbon than the native woodland plots. Combined non-native woodland treatments stored significantly more carbon than combined native woodland treatments and combined open peatland treatment and the native woodland stored significantly more carbon than the open

peatland treatments (figure 3.2b). The mean carbon stored in the non-native plots was 6.8 times higher than the average in the native plots. There was a small amount of non-native regeneration in the NP control plots and trees were entirely absent from the NN control, the tree carbon storage was not significantly different from zero when the two open peatland controls were combined. Using the data that does not account for carbon storage in tree roots decreases the amount of carbon stored in tree relative to the below and above-ground biomass combined data, but the general trends of the data remain the same.

The amount of carbon stored in self-seeded non-native tree regeneration in the native woodland restock plots makes a highly variable contribution relative to the planted native trees, ranging from less than 2% to over 90% of total tree carbon. The mean carbon storage in native trees is markedly higher for the SB and NWmix plots than the other native woodland restock plots (DB, SP1 and SP2) (figure 3.2c). There is no statistically significant relationship between native tree biomass and non-native biomass in native tree plot but there was significantly more non-native tree biomass in the combined Scot's pine plots than the combined broadleaf plots ($t=2.398$, $p=0.032$).

Individual planted non-native trees stored significantly more carbon than native trees ($T=18.906$, $p<0.001$). Mean carbon storage in individual planted native trees also varied significantly by species ($F=12.480$, $p<0.001$). Willows only represented 30% of the trees planted in NWmix but the mean carbon storage per willow tree planted was the highest of all native woodland species 8.77 kg C (table 3.3), this was significantly higher than any other species except silver birch. Downy birch and rowan stored the least carbon.

The general yield classes for the non-native treatments showed a similar pattern to the carbon estimates from the allometric equations. The SS/LP treatment had a similar mean to the self-seeded treatments (11 and 10 respectively) and there was less variability in SS/LP than the self-seeded treatment (confidence intervals were 1.31 and 4.53 respectively).

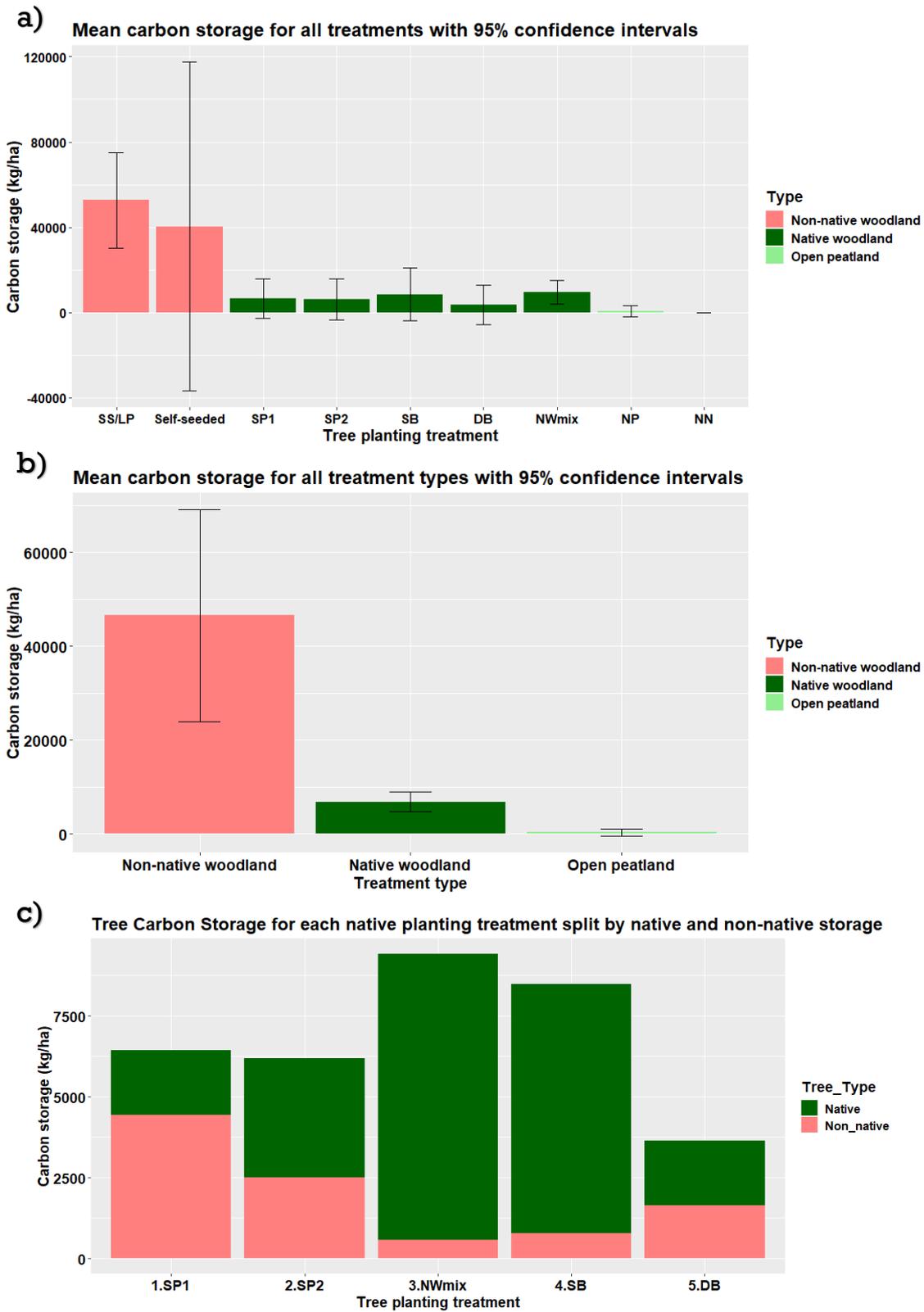


Figure 3.2: a) Mean carbon density for different planting treatments b) Mean carbon density for different groups of planting treatments c) Mean carbon density in native woodland treatment indicating what proportion of carbon storage is in self-seeded non-native regeneration.

Table 3.3: Mean carbon stored in individual planted trees across all plots. The two non-native species Sitka spruce and lodgepole pine have the highest mean carbon content. There is significant variation in the amount of carbon stored in different native species. The Non-native trees also have high survival rates. Native trees have lower but variable survival rates and there is no association between mean carbon content of living tree of a species and that species survival rate: rowan which stores the least carbon per tree at the highest survival rate and Scots pine which had the lowest survival rate has a relatively high per tree carbon storage.

Species	Carbon stored in individual planted tree		Survival of planted trees	
	Mean (kg)	Standard Error	Percentage (%)	Standard Error
Rowan	1.5	0.5	83.3	9.6
Downy birch	1.9	0.2	61.1	8.2
Scots pine	5.0	0.6	19.0	4.4
Silver birch	5.4	0.9	49.5	7.3
Willows	8.8	1.6	59.3	16.5
Lodgepole pine	19.4	1.2	99.0	1.0
Sitka spruce	21.5	2.1	99.0	1.0

Carbon storage in vascular ground flora

Carbon storage in the vascular ground flora represents an important carbon store relative to tree carbon storage under some conditions. For the native restock plots the mean contribution of carbon in non-tree vascular plants was 32% that of the native trees. Carbon storage in ground flora in native woodland plots was not significantly different from the two open control treatments, but the commercial restock treatments stored significantly less. There was no significant difference between different native woodland treatments but there was a significant negative correlation between total biomass in a plot and carbon stored in ground flora ($S = 918$, $\rho = -0.6392857$, $p = 0.01226$). Of the non-tree vascular vegetation carbon was mostly stored in the ericaceous shrubs *C. vulgaris* and *E. tetralix*, which contributed 94.3% of the carbon across all the plots.

3.4.2. Metrics of peatland condition

Water table depth

Water table was closer to the surface when tree biomass was lower with a significant positive linear relationship between water table depth and tree biomass in restocked plots in July 2018 ($R^2=0.34$, $p=0.00572$), October 2018 ($R^2=0.44$, $p=0.00106$) and March 2019 ($R^2=0.53$, $p=0.00185$) (figure 3.3). However, no significant relationship was found for the data of just native woodland treatments. Water table was significantly lower in the non-native restock plot than the open peatland and native woodland plots for all three months for which water table was measured. Open peatland had a water table significantly closer to the surface than the native woodland plots for October and March but not July where they were not significantly different.

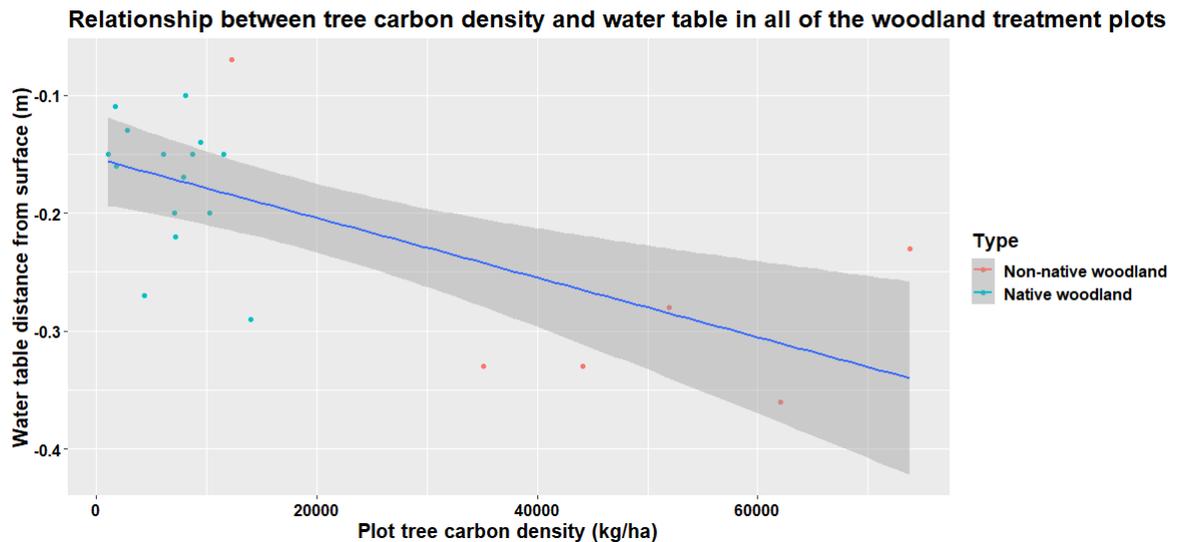


Figure 3.3: Scatter plot of carbon storage density and water table depth. The blue line is the linear regression line with the shaded area covering the 95% confidence intervals, illustrating a significant positive relationship. The colour coding of the point also highlights the difference between non-native and native woodland plots.

Tea bag decay rates

Tea bag decay and stabilisation rates were not significantly different in the tea bags buried under tree canopies and away from tree canopy cover (table 3.4). There were also no significant differences between tree biomass and either litter decay rate or stabilisation.

Peat carbon store

The estimate for below ground carbon stored in the peat was several of orders magnitude greater than stored in tree or non-tree vascular plants. The mean carbon storage of the peat in woodland treatment plots was 3,390,112 kg/ha (standard error=275,607). On average the peat under the non-native woodland plots stored 77 times more than the trees and peat under the native woodland plots stored 483 times more carbon than in the trees. The peat in the woodland treatment plots had a mean depth of 2.34m but was fairly variable across the plot ranging from 0.83 meters to 3.91 but all of the area would still be classed as deep peat by commonly used Scottish depth criteria (Forestry Commission Scotland, 2015).

Table 3.4: Mean and standard error of tea bag contents decay rate (k) measured as the proportion of liable material lost per day in rooibos tea bags over 90 days (%/day) and Stabilisation (S) measured as the proportion of hydrolysable green tea which remains after 90 days (%). The values are similar between treatments.

Grouped treatments	k_Canopy		k_Gap		S_Canopy		S_Gap	
	Mean	Standard Error	Mean	Standard Error	Mean	Standard Error	Mean	Standard Error
Non-native woodland	0.0102	0.0006	0.0010	0.0004	0.1334	0.0082	0.1618	0.0121
Native woodland	0.0105	0.0003	0.0103	0.0003	0.1254	0.0052	0.1234	0.0096
Open peatland	0.0111	0.0019	0.0114	0.0004	0.1585	0.0319	0.1580	0.0267

3.5. Discussion

3.5.1. Carbon storage in native and non-native planting treatments

The initial hypothesis that non-native restock options will have sequestered more carbon than the plots planted with native trees is supported by this study. There is a wide range in carbon storage between different species in each native restock treatment. Individual planted Scots pine, silver birch and willows on average stored more carbon than rowans or downy birch. However, the high mortality rate of Scots pine meant that the Scot's pine treatment plots on average stored less carbon than silver birch or willow plots highlighting the importance of both tree survival and growth rate in determining plot carbon density.

Self-seeded non-native trees made a variable but sometimes substantial contribution to plot carbon density in the native treatment plots, in particular contributing to the tree carbon density of Scots pine plots. During data collection it was observed that the Scots pine generally had a high height to canopy area ratio in comparison to the broadleaf species which canopies were relatively more widely spread (with the exception of rowans). In combination with the low survival rate of Scots pine plot this meant the overall canopy cover of Scots pine plots appeared to

be very low. Potentially the reduced shading from the canopy allowed more regeneration to establish better in the Scots pine plots, implying that the other native woodland tree treatments were having an inhibitory effect of non-native regeneration. Previous work has highlighted that high native woodland canopy cover can prevent colonisation of non-native trees in other ecological systems (Gómez et al., 2019).

Lindsay (2010) took a theoretical approach to show that peatland ground flora may store a substantial amount of carbon relative to trees and highlighted that this source of carbon storage is often overlooked. This study quantitatively demonstrates that vascular ground flora can make a substantial contribution to carbon stored in native woodland planting treatments. It also shows that increased carbon stored in tree growth may partly offset by decreasing carbon storage in the vascular ground flora. The vascular ground flora data highlights that this is a significant source of carbon storage but more comprehensive work would be needed to provide more complete and accurate estimates as is explained in a later section.

3.5.2. Impact of trees on peat carbon store

The carbon stored in the peat underneath the plots far exceeds the carbon stored above-ground in all the treatments highlighting the importance in understanding how the treatments affect below ground carbon storage in determining the carbon benefits of each option. However, the peat stock represents the net accumulation of carbon over millennia, not the carbon stock change during the second rotation. Therefore, tea bag decay/stabilisation rates and water table depth measurements were measured as different proxies to infer the effects of the treatments.

Water table depth is an important factor in determining the emission CO₂, CH₄ and nitrous oxide (N₂O) – important greenhouse gases (Bechtold et al., 2014). Lower water table depths indicates that a higher rate of oxidation is likely occurring and therefore greater CO₂ emissions even if concurrent CH₄ emission rises partly mitigate this effect (Evans et al., 2021). All the planted plot treatments had lower water tables than the open peatland treatments indicating that both native and non-native woodland types have higher oxidative losses than open bog. The weak but significant correlation between tree biomass and water table depth indicates that greater tree growth may lower the water table and therefore higher CO₂ emissions. However, the causality of this relationship isn't clear as although tree growth of peatland can cause a lowering of the water table, tree growth can also be increased by lower water tables (Sarkkola et al., 2010).

Therefore, it could be that a greater tree carbon store has grown where the water table was naturally lower. However, the significantly lower water tables in the non-native planting treatments compared to the native woodland treatments is unlikely to be due to natural site variation as the replicates and block design reduces the chance of treatment being consistently established under different natural conditions.

The tea bag experiment did not yield significant differences between plots, so the tea stabilisation and decay were similar across all plots indicating that the different treatments do not affect decomposition of the peat. However the Tea Bag Index is only one simple measure, some previous research has shown that tea bags can yield insignificant results in peatlands even when conventional litter bag methods have found significant differences (Macdonald et al., 2018). It may be that other methodological approaches would have found significant difference between plots. In addition, the results only apply to the conditions where they were buried. All bags were buried at a 6 cm depth, there may have been significant differences between treatments at other depths. A majority of water table depth measurements were deeper than 6cm so this depth may not be affected by any water table depth difference between plots. Low water tables would not explain the lack of effect of differences between wooded plots and open peatland plots where the water table did rise above 6cm. Potentially even if decay and stabilisation rates are similar the constraints on decomposition may differ between the open peatland and wooded plots.

3.5.4. Implications

Trade-off between different impacts and implications for PEW

The conventional restock plots had relatively low general yield classes, however, except for one self-seeded plot, these were all higher than yield class 8. According to the Forestry Commission Scotland (2015) guidelines this means the whole site could be a candidate for conventional restocking, but the borderline nature of its yield class also makes it a responsible candidate for PEW creation. Given the relatively productive nature of the site it is reasonable to speculate that if a similar site was assessed under the modern guidance, and it was decided that PEW should be created on the site, then the objective of the PEW would be more similar to the high intensity model of PEW identified in the previous chapter. The native woodland treatments in this plot appear similar to the description of the high intensity model for PEW.

This study indicates which species may be most appropriate for different objectives: At the site used in this study willows were the most effective species per tree at sequestering carbon. Broadleaf species were more effective than Scots pine at preventing non-native regeneration which was a motivating factor identified in the previous chapter.

The native woodland plots in this study indicate that PEW established in a similar way would be capable of some carbon storage while promoting vegetation diversity. From this study it remains unclear the extent to which the carbon sequestered in the trees of PEW would offset carbon losses from the peat soil. One previous study on carbon storage of native tree growth on peatlands has found evidence that carbon sequestered in trees does not compensate for CO₂ losses from peats (Friggeri et al., 2020), however this was on a site which did not have a history of afforestation before the native trees were established and additionally did not consider the effect of tree growth on other greenhouse gases such as N₂O and CH₄.

The reduced water table depth in native woodland plot relative to open plots indicates that PEW would probably be less effective at reducing carbon losses from the peat carbon store than open bog (Evans et al., 2021). There is also indication that the non-native restock options lower water tables more than the native restock option which would imply increased oxidative losses from the peat in non-native treatments compared to native (Evans et al., 2021). However, this study cannot indicate to what extent the carbon sequestered in the native or non-native trees may compensate for peat losses. It is unclear how the native woodland plot water table depth would have compared to open bog created by forest-to-bog restoration (rather than open bog which had never been afforested as in this study). Previous work on forest-to-bog restoration in the same region as the study site indicates that even after 17 years water table has not completely recovered to near natural levels (Gaffney et al., 2018).

Allometric equations

BD has been used to calculate allometric equations and is particularly commonly used for shrubby/multi-stemmed trees (Návar et al., 2004; Chojnacky and Milton, 2008; Berner et al., 2015; He et al., 2018) but the author is unaware of any study which has tested allometric equations using BD²H. For most of the species studied models based on BD²H were selected as the best, indicating that this maybe an overlooked metric for the construction of allometric equations. In particular BD²H was found to not only best describe several of the species which

grew as shrubby trees at the field site but also the Sitka spruce and lodgepole pine trees which were growing to a fairly 'standard' form (typically having one straight trunk).

Relationships between metrics such as DBH and crown area/volume are well established (Verma et al., 2014; He et al., 2018) but it is uncommon to take tree crown measurements in the field for the purpose of above-ground carbon estimation yet the high R^2 value of the model for multi-stemmed low growing willows indicates that this is an effective approach. Using canopy volume may therefore represent an effective way to rapidly estimate biomass of spreading multi-stemmed relatively short or 'shrubby' growing trees.

An initial exploration was undertaken to explore how the site specific allometric equations developed during this study compare with carbon storage estimates when applying allometric equations developed for the same species under similar conditions in other studies. This exploration has indicated pre-existing equations give values in a similar range to the site-specific equations including those which used a different predictor variable such as DBH. Further work on this data set could better contextualise the new equations developed in this study

3.5.5. Methodological limitations and improvements

Tree carbon storage

Each tree planting treatment that been replicated three times as part of the lay out of the pre-existing forestry experiment utilised by this study. This level of replication allowed some statistical comparison between treatments. However, the samples sizes are small which may limit which effects would be detected. In this study sometimes related treatments were pooled but this limits what can be tested for – for example difference between different native woodland treatments.

Tree felling was carried out in winter, so the carbon estimates for the native restock treatments, several of which were mainly composed of deciduous species, do not include carbon stored in summertime leaf mass. The equations for silver birch, downy birch, willow and rowan are therefore representative of carbon storage in the trees during winter but would be expected to underestimate carbon storage in summer. The proportion of tree carbon in leafy biomass is typically proportionally greater for smaller trees (He et al., 2018) so the relative importance of small trees for carbon storage would be increased relative to the winter results. Previous studies on the same or similar shrubby trees growing on peatlands indicates that leafy biomass may account for ~13.2-22.4% (He et al., 2018), this percentage is likely smaller for silver birch and

downy birch trees in the taller range of trees at Rumster (Johansson, 2007). Percentage increases in carbon storage in this sort of range are not likely to affect the general trends of the results.

Several assumptions were made to calculate the allometric equations and carbon storage estimates:

- The above/below ground biomass ratio from the IPCC were used to estimate below ground carbon storage, with the study making use of one ratio for all conifers species and one for all broadleaf species. A strong relationship exists between above ground biomass and below ground biomass but the ratio can vary substantially between species and growing conditions (Finér and Laine, 1998; Sinacore et al., 2017).
- The Sitka spruce and lodgepole pine allometric equations were based on destructive sampling of both self-seeded and planted trees on the assumption that they would both have similar growth patterns. The high R^2 values for the equations derived indicates that there was not substantial variability between self-seeded and planted trees.
- When calculating the allometric equations, whether or not felled individual trees stems were from the same tree was not accounted for. This is a violation of the assumption of independence between samples used in linear regression.

The magnitude of the differences between non-native and native plots gives confidence that violations of these assumptions would be unlikely to change the main results of this study.

A further uncertainty is how stable the carbon sequestered in each tree species might be, and the extent to which wood products from the trees would substitute for more polluting materials. The non-native tree planting treatments resembled conventionally restocked commercial plantations, and as such could be harvested on rotation. The native plantings were proxies for PEW and under this management the woodland would potentially never be harvested, and so all the products of tree growth would stay in-situ. The maximum size of the tree carbon store would therefore be dependent on the rate at which woody material decayed on the site and if any of this woody material, such as the roots, would have reduced decay rates due to preservation in the peat. The large below-ground carbon store would, under the right conditions, be capable of storing carbon for thousands of years, which is a key argument for those who advocate prioritising reducing losses of below-ground carbon stores above balancing carbon losses from the below-ground stores with carbon sequestration in trees (Hargreaves et al., 2003; Hermans et al., 2019).

Carbon storage in ground flora

The sampling approach for the vascular ground flora was limited, sampling 0.12m² of ~322m² plots (~0.04% of plot area)) so the data is expected to have poor representativeness. One source of evidence for this was one quadrat in SP2 block 3 contained a far greater amount of carbon than any other quadrat due to the quadrat including the base of several substantial *Calluna vulgaris* stems.

This study did not account for the bryophyte lower or below ground vascular carbon storage as this material was deemed too challenging to extract and distinguish from peat material which formed before the plots were established. Potentially bryophyte carbon content could be estimated from Net Ecosystem Exchange (NEE) rates for the dominant species multiplied by the age of the plots.

Quantifying impact of tree treatments on peat carbon storage

None of the measures assessing the condition of the peat carbon store were directly measuring the impact of the tree treatments on peat carbon storage. The peat coring methodological approach used was able to measure carbon stocks at the point where the core was taken from, but the approach cannot determine whether there has been a change in stock due to the second rotation, and therefore whether the above-ground stock compensate for any potential losses. In order for this to be determined a much more intensive methodological approach would be required that was beyond the scope of this study. For example, the peat core based methodology employed by Sloan (2019) where carbon stocks accumulated over the same time period (dated by tephra) were compared. This approach may not have worked well at the Rumster field site since as the site was in the second forestry rotation it was difficult to confidently identify areas where the original peat stratigraphy had not been disturbed by ground preparation work which is essential for this approach to work. At the site used in this study the land was double mouldboard ploughed in the first rotation and then hinge mounded in the second rotation. In addition, both of these groundwork features had undergone subsidence. If cores were required to have an undisturbed stratigraphy then they should be taken from half way between the furrows from the first rotation. This is the area least likely to be affected by spoil from the furrows. An area should be chosen that is away from where mounding has taken place for the second rotation trees. However, given the level of disturbance and ambiguity of site interpretation it would be hard to guarantee any cores would have an undisrupted stratigraphy. An alternative approach would be

to have taken greenhouse gas flux measurements from the peat surface which would give an estimate of contemporary greenhouse gas exchanges. There is also some debate over the extent that a unit mass of carbon stored in trees should be treated as equivalent to the same mass stored in peat since the later has the potential to store carbon over much longer periods of time if under the right management (Hermans et al., 2019).

3.6. Conclusion

Establishing native woodland on degraded peatland, such as Forestry Commissions Scotland's concept of PEW, has been proposed as a way to create environmental benefits from peatland sites that are not obvious candidates for continued exploitation or forest-to-bog restoration. This study has shown that native woodland established in afforested peatlands can provide some benefits. However conventional restocking sequestered considerably more carbon in the tree biomass.

This study only has limited indirect evidence for the extent to which the different tree planting treatments affected greenhouse gas emissions from the peat and the extent to which this offset the carbon benefits of the different planting treatments. Additional research is required to further quantify the carbon balance of these systems, including the greenhouse gas emissions from the peat itself. Given the size of the below ground carbon store relative to the carbon stored in trees a greater understanding of this is important for a fuller understanding of PEW.

This study has highlighted some of the trade-offs associated with different native and non-native tree planting treatments. However, peatland management is based on consideration of many factors including several not explored by this study such as animal diversity and downstream water quality effect which would need to be explored further in future research.

Linking Chapter 3 and Chapter 4

The previous chapter aimed to quantify long-term carbon accumulation. However, it could not directly inform on what effect the different treatments were having on greenhouse gas flux from the ecosystem. Carbon accumulation in the trees will be a component of the CO₂ flux but the last chapter cannot directly assess the CO₂ fluxes from other components of the ecosystem such as the peat or ground flora, nor can it quantify the rate at which CO₂ is currently being sequestered by the trees. The previous study also does not explore the effect of the treatments on other greenhouse gases, especially CH₄ which is widely recognised as an important part of peatland greenhouse gas budgets.

The next two chapters focus on CO₂ and CH₄ gas fluxes. The site used in the previous chapter was an area still predominantly used for forestry. The site could be used to compare PEW-like habitats with conventionally restocked plantations but did not compare PEW-like habitats with areas which had been restored to open bog. In the next chapter two plots are compared, both of which have undergone forest-to-bog restoration management but only one of which has remained treeless, while the other has become wooded. The trees in the wooded plot are stunted native birch that have established through regeneration. The site in some ways resembles a 'low intensity' model for PEW as identified in Chapter 2 although it does not have all the features of a low intensity model so may be better referred to as intermediate intensity. Using this second field site, Flanders Moss, therefore expands the breadth of PEW models explored by this thesis.

The work done for Chapters 1 & 2 highlighted birch regeneration on open bog as a specific concern of many stakeholders both in forest-to-bog restoration projects but also in other peatland restoration contexts such as cut-over peatlands. There is only limited literature on the effects of birch regeneration on peatland and no work looking at birch regeneration on forest-to-bog restoration sites in terms of greenhouse gas fluxes. As such, working on Flanders Moss was an opportunity to provide data pertaining to a widespread issue for peatland restoration.

4. Investigating the Effect of Native Birch Regeneration on Methane and Carbon Dioxide Fluxes in a Forest-to-Bog Restoration Project

4.1. Abstract

Forest-to-bog restoration is increasingly being carried out in the British Isles. Despite this there is limited data comparing the effect of different restoration approaches to forest-to-bog restoration. Tree regeneration is a common issue in forest-to-bog and other peatland restoration contexts and some now advocate for allowing some tree regeneration. In this study, greenhouse gas fluxes were compared from two adjacent plots in a 22-year-old forest-to-bog restoration project, one plot in which the original plantation trees were entirely removed from the site such that the plot has remained almost treeless, and the second plot in which only the original tree trunks were removed, leaving the tree brash on the site, which was subsequently colonised by stunted birch trees. Both plots are no longer net emitters of carbon dioxide (CO₂) but are still net greenhouse gas emitters driven by high methane (CH₄) fluxes. Methane emissions are substantially higher in the plot which has not been colonised by birch trees, potentially due to its wetter conditions. The study provides evidence that even 22 years after forest-to-bog restoration is undertaken (consisting of tree removal and blocking of the main drains) a site can remain a net source of greenhouse gases. The study finds no evidence that the leaving of brash on this forest-to-bog restoration site and the subsequent establishment of stunted birch trees adversely affects net greenhouse gas emissions and in fact provides evidence of lower methane emissions in this scenario. The results of this study highlight a need for more evidence on when and how forest-to-bog restoration can achieve net zero or negative greenhouse gas balance.

4.2. Introduction

Peatlands are globally important carbon stores (Beaulne et al., 2021) and support a broad range of other ecosystem services and specialist biodiversity. Peatlands in Scotland alone are estimated to store 1620 Mt of carbon (Chapman et al., 2009). However large areas of peatlands around the world have been drained to make them more productive for activities such as forestry, agriculture, or peat mining (Rebekka and Chapman, 2016; Dohong et al., 2017).

In Scotland, ~17% of deep peatland area, the vast majority of which is naturally open, was drained, ploughed and planted with non-native conifer species (Vanguelova et al., 2018). The afforestation predominantly took place from the 1960s to 1980s (Sloan et al., 2018b). Initially concerns about the environmental impact of afforestation focused on the habitat degradation and how this effected biodiversity such as ground nesting bird species (Stroud et al., 1987; Lindsay et al., 1988). These concerns helped drive legislative changes that facilitated the protection of deep peat areas from forestry expansion (Patterson and Anderson, 2000; Stroud et al., 2015). Concerns about the edge effects of forestry plantations on adjacent open habitats also drove initial efforts to carry out forest-to-bog restoration in some areas of high ecological importance (Stroud et al., 1990; Wilson et al., 2014).

During the 21st century, greenhouse gas balance has become a substantial consideration in peatland management, an importance that is reinforced by major international agreements such as the COP26 Glasgow Climate Pact (UNFCCC, 2021) which sets goals requiring urgent and drastic reductions in greenhouse gas emission including those associated with Land Use, Land-Use Change and Forestry (LULUCF). Artificial drainage and the direct effect of trees (evapotranspiration and interception) lower the naturally high water table in afforested peatlands (Payne et al., 2018). This increases the depth of peat susceptible to oxidative decay resulting in release of carbon as carbon dioxide (CO₂) (Lindsay et al., 2014b). Furthermore, the disturbance and draining of afforestation can also cause elevated production of Dissolved Organic Carbon (DOC) (Gaffney et al., 2018). Dissolved Organic Carbon can accumulate in peat pore water and can potentially enter watercourses; up to 55% of DOC lost to watercourses can be converted to CO₂ either chemically by photoreactivity or biologically by bacteria (Pickard et al., 2017). However, mature plantations don't necessarily release more DOC into the catchment than open bog once they are established (Flynn et al., 2022).

Carbon losses from deep peat due to afforestation will be partially or possibly entirely compensated for by soil inputs from the trees such as root exudates and leaf litter, as well as

carbon sequestration into the trees' timber (Hargreaves et al., 2003; Vanguelova et al., 2018, 2021; Hermans et al., 2022). However, carbon storage in timber is relatively short-lived compared to carbon stored in a peatland in good condition, so many advocate for widespread forest-to-bog restoration as a long term way to safeguard the large peat carbon stores (Hargreaves et al., 2003; Hermans et al., 2019). Forestry policy in Scotland now only permits restocking of existing forestry plantations if the new rotation will be able to achieve a rate of growth that is predicted to compensate for soil carbon losses, defined as a general yield class for Sitka spruce of 8 or more (Forestry Commission Scotland, 2014, 2015).

Forest-to-bog restoration is an alternative to continued forestry. The process typically involves two main aspects – raising the water table (e.g. by drain blocking) and felling of the trees (Anderson, 2010; Andersen et al., 2017). In Scotland different approaches to forest-to-bog restoration have been and continue to be developed over the last 20 years. A majority of the studies on the effect of forest-to-bog restoration have focused on older restoration methods which usually involve drain blocking and either trees being felled-to-waste, tree trunks being removed but the lower value brash being left or whole tree removal. These studies indicate initially raised CO₂ emissions as a result of restoration work, followed by long term falls in CO₂ fluxes so that forest-to-bog restoration sites are net sinks of CO₂ within 5-10 years (Hambley et al., 2018; Rigney et al., 2018; Lees et al., 2019; Creevy et al., 2020; Mazzola et al., 2021). Another study indicates that CO₂ losses from the peat itself are substantially reduced (Hermans, 2018). Dissolved Organic Carbon concentration in pore water and in downstream water courses has been found to remain constant annually but with higher concentrations in summer (Gaffney et al., 2020, 2021), while other studies find it to rise after forest-to-bog restoration relative to afforested areas (Howson et al., 2021a) but then reduce over several years (Gaffney et al., 2018; Shah and Nisbet, 2019). Dissolved Organic Carbon losses from forest-to-bog sites are usually relative low in comparison to direct gas fluxes (Hermans et al., 2022).

Restoration, by contrast to its effect on DOC, increases methane (CH₄) fluxes from the plots relative to afforested peatlands, which typically have much lower CH₄ fluxes even than near-natural peatlands. This is because, unlike CO₂, CH₄ is produced more readily in waterlogged, anoxic conditions (Conrad, 2020). Creevy et al. (2020) finds that CH₄ fluxes rise rapidly in the initial stages of restoration but then start to reduce to near-natural peatland fluxes: when CH₄ was accounted for on a six-year-old restoration site it was found not to be a net carbon source, whereas a 17-year-old site was found to be a net sink. Conversely, Hermans (2018), finds that CH₄ fluxes rise with age from restoration tending towards near-natural peatland emissions. Although CH₄ fluxes are usually of smaller magnitude than CO₂, CH₄ is a greenhouse gas which has a global

warming potential 28 times higher than CO₂ over a hundred-year timescale (IPCC, 2013; Myhre et al., 2013), meaning that it can be the dominant greenhouse gas at some sites. Methane rise is a general consequence of peatland restoration projects with a range of management histories: there is scientific uncertainty over the net climate-forcing effect of rewetting peatlands but the predominant view is that in general rewetting drained peatlands will prevent them from being net sources of greenhouse gases in the long term (Günther et al., 2020; Humpenöder et al., 2020; Evans et al., 2021).

A common issue for forest-to-bog restoration sites – especially where at least some areas or microtopographic areas remain dryer than in near natural conditions – is the re-establishment of trees. These may be either the original crop species or native species such as downy and silver birch (*Betula pubescens* and *Betula pendula*). This factor also occurs on peatland sites degraded by other activities, as well as internationally elsewhere in Europe and North America (Fay and Lavoie, 2009; Sotek et al., 2019). There are concerns about how scrub and trees may impact on peatlands. Effects might include the lowering of water tables, a positive feedback whereby the dryer conditions created by scrub and trees facilitates greater rates of scrub and tree expansion, and loss of open habitat species (Fay and Lavoie, 2009). Removing birch trees that are establishing on peatlands is a major component of some restoration projects (e.g. Scottish Natural Heritage, 2014). Despite this effort being a common one, most research on the effect of trees on peatlands focuses on the impacts of intensively established commercial plantations, and there is only limited research on the effects of self-seeded birch trees on peatland processes.

In the context of uncertainty over the effect of birch regeneration on peatland restoration some researchers (Renou et al., 2007) and policy makers (Forestry Commission Scotland, 2014, 2015) have focused on the benefits that woodland might bring, for example promoting biodiversity associated with native woodland and carbon storage in the trees. It has been suggested that in some situations it may be appropriate to accept native tree establishment and regeneration as part of a sustainable management strategy. Forestry Commission Scotland (2014, 2015) currently advocates the establishment of a habitat they term Peatland Edge Woodland on sites which, according to their guidelines, are not good candidates for restocking with conifer or forest-to-bog restoration. This habitat is envisaged as a low density, low intensity, predominantly native woodland which could be established either through planting or natural regeneration.

There is no research specifically on the effect of native regeneration on forest-to-bog restoration sites however there is some research of similar habitats in other contexts. Friggens et al. (2020) found that planting birch trees resulted in a net loss of carbon from peatlands when

carbon loss from the peat was compared to carbon sequestration and soil carbon inputs from the trees. Mazzola et al. (2022) studied a lowland raised bog which had been colonised by Scots pine forming a low density woodland. This study found that CO₂ fluxes elevated under the canopy of trees but CH₄ fluxes were reduced relative to open bog. Even if the greater climate change potential for CH₄ is considered, the rise in CO₂ would mean areas closer to trees were greater carbon sources, however this study does not consider the carbon sequestrations or soil inputs from the trees. Research by Limpens et al. (2014) found in a mesocosm experiment that 1.5-1.8 m tall birch trees can have a drying effect through direct evapotranspiration at densities of 0.9 tree/m². However, at higher densities of 1.8 tree/m², the shading effect of trees reduces evapotranspiration from the ground flora and peatland surface to such an extent that the water table was actually higher in these areas than in the absence of trees.

The aim of this chapter is to quantify and compare the greenhouse gas contributions of two plots in a forest-to-bog restoration project. In the first plot only the tree trunks have been removed and the tree brash has been left on site, facilitating subsequent birch invasion (now consisting of approximately 20-year-old stunted trees). In the other plot trees had been entirely removed, inhibiting birch invasion. The objectives were to: 1) Quantify and compare ecosystem CO₂ flux, CH₄ flux and DOC in both forest-to-bog restoration plots 2) Compare the vegetation composition and environmental conditions between the two plots and the microtopography within them and relate this to differences in fluxes.

4.3. Methods

4.3.1. Site

Field work was conducted at Flanders Moss National Nature Reserve – a lowland raised bog located in the Carse of Stirling, UK, British National Grid: NS644978 (figure 4.1). A section of the peat dome had been drained, ploughed and planted with lodgepole pine in the mid-1970s but the trees were subsequently removed, and the main drains blocked in the summer of 1998 as part of major forest-to-bog restoration work (Pickett, 2004). Part of the work was the creation of a set of experimental plots to which different restoration techniques were applied with the aim to test their efficacy (Anderson, 2010). In all the original plots the ridge-furrow ploughing

microtopography was still obvious and no work had been done to reprofile the site or block the furrows.



Figure 4.1: Location of the two plots in Flanders Moss. Images generated from Google Earth Pro, accessed 07/12/21.

In this study, two of the plots in this now inactive experiment were used (table 4.1); one plot was established in an area where whole trees were removed while in the other, approximately 30 m away, only the tree trunks had been removed while the brash was left on site. Silver birch and downy birch trees had seeded in both 160 m² plots but had grown much more successfully in the plot where the brash had been left with six downy birch and 75 silver birch trees having grown taller than 140 cm, by contrast to the plot where whole trees had been removed, in which only two downy birch and four silver birch trees had grown taller than 140 cm (see images, table 4.1). For the remainder of this chapter the plot where brash had been left is referred to as the “wooded plot” and the plot from which the whole trees were removed as the “open plot”.

My experimental plots were embedded within the bounds of the original plots and had dimensions of 16 m by 10 m (160 m²), which crossed seven sets of plough ridges and furrows. Both plots had a peat depth of approximately 8 meters.

Table 4.1: Details of the two experimental plots' treatment and tree density.

Plot name	Wooded plot	Open plot
Image of the plot in 2020		
Restoration treatment	Original conifer crop felled but only main trunk removed – brash left on site	Original conifer crop felled and completely removed
Silver birch density (tree height > 140 cm)	0.47 trees/m ²	0.03 trees/m ²
Downy birch density (tree height > 140 cm)	0.04 trees/m ²	0.01 trees/m ²
Total tree density (tree height > 140 cm)	0.51 trees/m ²	0.04 trees/m ²
Tree height of trees >140 cm tall (mean ± CI)	2.57 ± 0.14 m	1.77 ± 0.32 m
Maximum tree height	4.585 m	2.5 m

Comparability of plots

The previous experiment on the field site had collected data from the two plots used in this study before and after the restoration treatments were applied. Data were collected on three variables from six sampling points in each plot: vegetation percentage cover by species, water table depth, and soil bulk density and percentage mass water at 10-20 cm depth and 70-80 cm depth. Water table depth was measured approximately once a month during three campaigns 1) Dec 1997 to Jun 1998 (n=7), 2) Oct 1998 to May 2001 (n=24) and 3) April 2008 to March 2009 (n=13). The first campaign took place before restoration work and the second and third occurred after restoration work. For each month the mean of the six measurements from each plot was taken (table 4.2).

Two sample T-tests were used to compare bulk density and gravimetric water content. For all T-tests the assumption of normality was checked with Shapiro-Wilk tests and equality of variance was checked with F tests. As the data significantly violated the assumption of normality paired Wilcoxon-tests were used to compare the mean water table depth data from the open and wooded plot for each campaign for each month.

Overall, the plot comparisons imply that the conditions of the two plots were similar immediately before and after the restoration treatments were applied (table 4.2). The water table depth is significantly different in all three campaigns, but these differences are relatively small; the implications of this are explored further in the discussion. The open plot has a water table significantly closer to the surface in the first campaign, but the wooded plot has a water table significantly closer to the surface in the second and third campaigns. The implications of this difference and how they may influence the study's results is explored further in the discussion.

The vegetation percentage cover in both plots before the treatment were similar. Both plots were dominated by needle litter and *Hypnum cupressiforme*. No *Eriophorum* species were recorded as present in either plot. Most quadrats in both plots contained no *Sphagnum* species. The only *Sphagnum* species was *Sphagnum capillifolium* which is found in one of the six quadrats in the wooded plot and two of the six quadrats in the open plot.

Table 4.2: Pre-existing data of the plots used in this study compared to identify any pre and early treatment differences between the plots. Measurements taken in 1997/1998 occurred before the restoration treatments.

	Date measurements taken	Mean wooded (\pm CI)	Mean open (\pm CI)	T test
Bulk density at 10-20 cm depth (g/cm ³)	1997	0.131 (\pm 0.017)	0.138 (\pm 0.019)	T=0.53, p=0.608
Bulk density at 10-20 cm depth (g/cm ³)	2000	0.114 (\pm 0.020)	0.121 (\pm 0.019)	T=0.47, p=0.651
Bulk density at 70-80 cm depth (g/cm ³)	1997	0.058 (\pm 0.004)	0.070 (\pm 0.005)	T=3.67, p=0.004
Bulk density at 70-80 cm depth (g/cm ³)	2000	0.067 (\pm 0.012)	0.062 (\pm 0.007)	T=0.73, p=0.482
Percentage mass water at 10-20 cm depth (%)	1997	85.39 (\pm 1.21)	83.06 (\pm 2.02)	T=1.94, p=0.081
Percentage mass water at 10-20 cm depth (%)	2000	89.02 (\pm 1.96)	88.51 (\pm 1.91)	T=0.37, p=0.722
Percentage mass water at 70-80 cm depth (%)	1997	94.08 (\pm 0.41)	93.00 (\pm 0.51)	T=3.233, p=0.009
Percentage mass water at 70-80 cm depth (%)	2000	93.35 (\pm 1.19)	93.93 (\pm 0.41)	T=0.92, p=0.393
Water table depth (cm) during 1 st campaign	Dec 1997 to Jun 1998	36.2 (\pm 9.52)	33.8 (\pm 9.4)	z=2.20, p=0.028
Water table depth (cm) during 2 nd campaign	Oct 1998 to May 2001	6.7 (\pm 2.3)	10.0 (\pm 2.3)	z=4.29, p<0.001
Water table depth (cm) during 3 rd campaign	April 2008 to March 2009	6.4 (\pm 3.3)	8.1 (\pm 2.7)	Z=2.87, p=0.002

4.3.2. Data Collection

Experimental layout

Previous studies have identified three topographical levels in ploughed peatlands corresponding to the plough ridge, plough furrow and the original peat surface unaffected by ploughing. These differing topographies affect vegetation (Hancock et al., 2018) and gas fluxes (Hermans, 2018; Mazzola et al., 2021) so this study was spatially structured to ensure equal representation of these three topographical levels. In the experimental plots the three topographical levels were difficult to identify reliably, possibly due to the original ploughing having been single mouldboard at relatively dense spacing, by contrast to the distinctive double mouldboard ploughing which is more typical of peatland forestry. To resolve this, an 'intermediate' topographical level was defined to replace 'original surface' as used in other studies (e.g. Hermans, 2018; Creevy et al., 2020), figure 4.2. Intermediate topography broadly corresponds to the original peat surface but also includes areas suspected to be former ridges that have collapsed due to subsidence, and former furrows that have filled in. These three levels have distinct floral and physical characteristics: 1) furrows are the wettest and lowest topographical level and are generally dominated by *Sphagnum* species 2) ridges are the driest and highest topographical level typically dominated by mosses relatively tolerant of dry conditions (such as *Hypnum cupressiforme*, *Pleurozium screberi* and *Polytrichum* species) and *Calluna vulgaris* 3) the intermediate zone has intermediate physical and floral properties to the other two levels.

In both blocks, fifteen sampling points were established in the form of five blocks of three micro-topographic positions, with each block consisting of a sampling point on each topographic level in close proximity of each other. The approximate location of each block was determined by selecting random coordinates within the plots, but the sampling points themselves were chosen at the coordinate location in order to cover the variety of dominant vegetation assemblages present. This is because vegetation type can have a major influence on gaseous carbon budgets (Creevy et al., 2020). At each sampling point, a 20 cm diameter PVC collar was installed into the vegetated peat surface to a depth of approximately 3 cm, ensuring the collar formed a gas-tight seal between the bottom of the collar and the peat. In addition, at each sampling point a dipwell was installed. Dipwells were installed at least 10 cm away from the collar so to minimise any effect they might have on conditions in the collar, but care was taken to install the dipwell at the same microtopographic level as the collar. The dipwell consisted of a 4 cm diameter, 1 m long open-ended PVC pipe with a series of holes drilled at 15cm intervals along its length to allow the

water table depth to easily equilibrate inside it. All sampling points were accessed via a network of duckboards, which minimised disturbance. All the equipment was installed over one month prior to taking measurements.

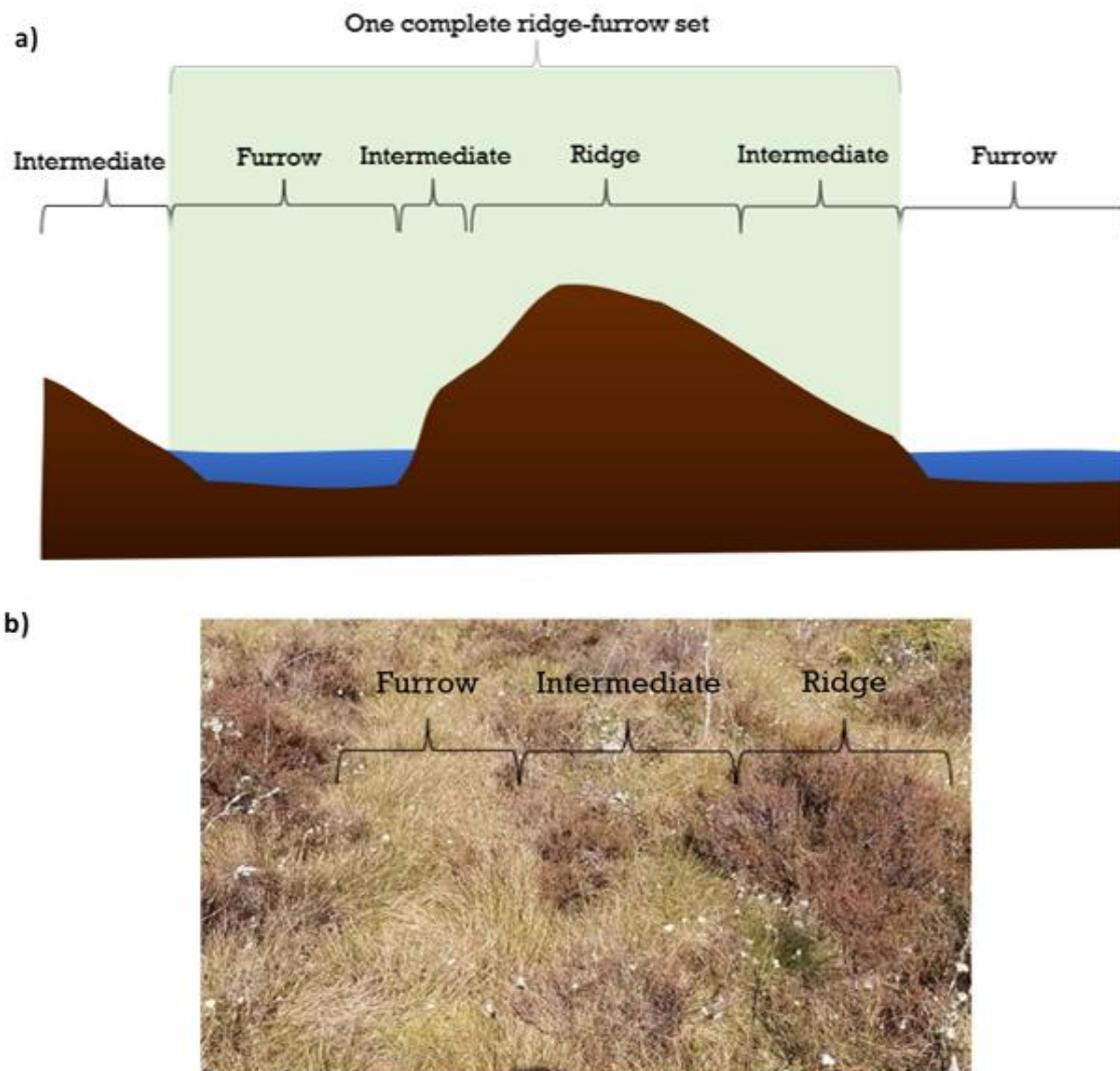


Figure 4.2: a) A representation of how the three topographical categories related to topographic level left by single mould board ploughing. The recent vegetation established is left out of this representation to emphasize the underlying microtopography of the peat. b) A picture from the open plot with the position of topographic categories overlaid.

Gas fluxes

It was originally planned that CH₄ and CO₂ flux measurements would be taken every month for a year, but equipment issues and the disruption arising from Covid-19 restrictions resulted in measurements being focused into three campaigns consisting of a total of eight sets of measurements: 1) An autumn/winter campaign consisting of measurements taken in October 2019, January 2020 and February 2020; 2) A summer/early autumn campaign consisting of measurements taken in July 2020, August 2020, September 2020, and October 2020; 3) A spring campaign consisting of measurements taken in May 2021. For all visits, gas fluxes were measured from every collar, except in December 2019 when equipment issues meant that an incomplete set of flux measurements was collected.

Methane and CO₂ fluxes were measured using a closed static chamber system at all sampling locations from October 2019 – May 2021. A static 20 cm wide x 30 cm tall transparent Perspex chamber was gas-tight sealed on the flux collars with a modified rubber in-tyre. The linear nature of the gas flux measurements taken indicated that airtight seals between the chamber and the collar and the collar and the surrounding peat had been achieved. The chamber was connected to a Los Gatos Research (LGR) Ultraportable Greenhouse Gas Analyser (UGGA) which measured CH₄ and CO₂ concentration every second and corrected for H₂O concentration. The UGGA was connected to the chamber with 3.2 mm external diameter Bev-A-Line® tubing. Gas was taken up from the chamber through a single uptake tube at the top of the chamber and returned to the chamber via a perforated ring at the base of the chamber. The use of a water trap before the inlet tube to the analyser protected the UGGA from water taken up by the tubing.

Initial testing using a 5 cm diameter computer fan with a flow rate of approximately 12m³/hr positioned ~20 cm from the bottom of the chamber showed that the fan did not significantly affect magnitude of fluxes but did increase R² values of flux calculation; consequently, all measurements were taken with a fan in operation.

Typically, two flux measurements were taken from each collar: a light measurement and a dark measurement, in which the chamber was covered by a hood made of a reflective material that prevented light from passing into the chamber. The light measurement was taken first, with each measurement lasting three minutes, after which the chamber was removed and the chamber air allowed to equilibrate with the outside air for at least three minutes before the chamber was returned to the collar with the dark cover for further three minutes. The live output of the UGGA analyser was observed during flux measurements and if a measurement appeared

unusable (e.g. evidence of chamber leakage from ineffective sealing, CH₄ ebullition event) a further measurement was taken.

For each flux measurement, PAR was measured in an unshaded area with Quantum Sensors connected to GP1 dataloggers (Delta-T Devices Ltd), which took readings every five seconds during the measurement. A shaded temperature sensor was fitted at approximately 25 cm height inside the chamber and recorded the air temperature every five seconds so that chamber air temperature could be used to correct for air volume in the chamber. Air temperature outside the chamber was also measured, at 10 cm above the soil surface using a shaded sensor connected to a GP1 datalogger. Water table depth was measured from the dipwell associated with the sampling point. Soil temperature was measured at 5 cm (Temp_5cm) and 10 cm (Temp_10cm) using a handheld temperature sensor (Foodcare, Hanna instruments) and volumetric soil moisture at 0-6 cm depth was measured at three locations immediately around the flux collar (ML2x Theta probe connected to HH2 Moisture Meter, Delta-T Devices). Temperature and moisture were not measured inside the collar to minimise disturbance.

Diurnal flux measurements

As well as the daytime seasonal measurements, two sets of diurnal measurements were taken from a randomly determined subsample of nine flux collars, three from each topographic level in both the wooded and open plots. This total of 18 collars was a reduction made to increase the number of repeat measurements that could be taken within the confines of a 24-hour period. Two diurnal campaigns were carried out, one on 13th/14th August 2020 and one on 21st/22nd October 2020. The August day was mostly clear and warm, the October day was cooler and overcast. For the August campaign a full set of measurements was taken four times in: 1) the morning 8:45-12:15; 2) the afternoon 14:00-19:00; 3) the early night 20:40-00:00; 4) the late night 02:30-6:50. The two night-time sets of measurements had negligible PAR intensities so only a single flux measurement was taken for these measurements. For the October campaign three sets of measurements were taken: 1) in the morning 8:45-13:30; 2) in the afternoon/evening 15:00-20:00; 3) at night 21:45-00:30. A light and dark measurement was taken for the morning measurements and some of the afternoon/evening measurements and only one measurement was taken for the night measurements and the later afternoon/evening measurements, as PAR was diminished to negligible levels.

Fluxes were measured from one block of three collars at a time, each consisting of one collar on each of the furrow, intermediate and ridge topographic levels. Measurements were taken alternately from blocks in the wooded and open plot. Light and dark measurements were taken for the morning and afternoon measurements but only a single measurement was taken for the night-time measurements in the dark as NEE is equal to ER overnight.

Calculations of fluxes

All gas flux rates were calculated in R version 3.6.2 (R Core Team, 2021) using the package FluxCalR (Zhao, 2019) which uses a linear fitting method to calculate gas flux rate for the section of the measurement window with the highest R^2 value (the most linear section of the measurement). Linear regression was used to calculate fluxes as the measurements generally showed strong linearity. A window length of 1.5 minutes was chosen as this provided good coverage of the measurement while typically being short enough to avoid being affected by curvature in the measurement. It is also a window-length commonly used for gas flux studies on temperate ombrotrophic peatlands (e.g. Heinemeyer et al., 2013). Some CH_4 flux measurement windows were moved from that determined by FluxCalR if they were clearly positioned on an ebullition event. In addition, some CO_2 flux windows were moved earlier in the measurement if there were signs of curvature later in the measurement.

For CO_2 flux measurements, the light measurement corresponds to Net Ecosystem Exchange (NEE) and the dark measurement corresponds to Ecosystem Respiration (ER); measuring both also allows Gross Primary Productivity (GPP) to be estimated by subtracting ER from NEE.

Methane flux in the systems measured is not expected to be significantly affected by the presence or absence of light, and as such the light and dark CH_4 flux measurements could be considered repeated measurements. It is however plausible that the disturbance from the first measurement (e.g., heating effect, ebullition) may affect the magnitude of the second CH_4 measurement, but a paired t-test on Box-Cox transformed light and dark CH_4 flux measurements ($n=237$) showed no significant difference between light and dark measurements ($p=0.122$). The first CH_4 flux measurement was the one selected for use in statistical analysis to mitigate against any undetected effect. There were a small number of exceptions in which the second measurement was used in place of the first where was evidence that the first measurement had been affected by ebullition/ CH_4 spikes.

All flux measurements were inspected for outliers using the Inter-Quartile Range (IQR) criterion where observation $> 3^{\text{rd}}$ quartile + 1.5 x IQR or $< 1^{\text{st}}$ quartile -1.5 x IQR are potential outliers. Different approaches were used to calculate the Inter-Quartile Range (e.g. calculating IQR for: all data, separately for each collar, separately for each month). This approach highlighted a small number of potential outliers in the CH₄ data however these appeared to reflect important variations in the data so were kept (e.g. several values for August which was the month with the highest CH₄ fluxes). The final dataset therefore had a complete set of values for CH₄, NEE, GPP and ER for all collars and time points.

Peat pore water DOC and nutrient concentrations

A Macrorhizon sampler with a pore size of 0.15 μm and sampling length of 9 cm (Rhizosphere, number: 19.21.35) was installed at 10 cm depth from the vegetation surface at every sampling point adjacent to each flux collar. A 50 ml peat water sample was taken from each rhizon for every set of flux measurements taken (eight sampling dates in total). Half of each water sample was filtered using 0.15 μm filters stored at 4°C and analysed for DOC using a Vario TOC Cube Analyser (Elementar Ltd). This analyser purges inorganic carbon with phosphoric acid then combusts the remaining sample, converting all remaining carbon present to CO₂, and uses a non-dispersive infrared spectrophotometer to detect Total Organic Carbon content of the sample, which in filtered samples will correspond to the DOC concentration. Conventionally DOC is defined as passing through a filter of $<0.45 \mu\text{m}$ but previous studies have shown that the majority of DOC in organic soils is below the 0.15 μm size used in this study (Chow et al., 2005). Raw absorbance values were calibrated with 5, 10, 20, 50 & 100 mg C/l NPOC solutions and drift corrected every ten samples with 50 mg C/l NPOC standards. Samples were typically analysed within five days, but Covid-19 disruption meant that the samples collected in July and August were in cold storage for up to six weeks before analysis. Previous studies on DOC samples have indicated that cold storage over extended periods such as this does not result in significant changes in DOC concentration (Peacock et al., 2015) and the results for these months do not appear to be anomalous. The other half of each water sample was frozen at -20°C as soon as possible (usually within 48 hours) and stored until time of analysis for phosphate, ammonium, and nitrate using an Autoanalyzer 3 (SEAL Analytics Ltd) which uses colorimetry to detect nutrient concentration. Raw absorbance values were calibrated with 0, 0.5, 1, 1.5 & 2 mg/L phosphate-P, nitrate-N and ammonium-N standards and drift corrected every 10 samples with 2mg N/l or mg P/l standards.

Representative of vegetation in flux collars

To assess the relative abundance of different vegetation types and representativeness of the sampling points, three transects were made through both plots in May 2021. These transects passed perpendicular to the ploughing so that in total each transect passed over seven ridge-furrows. Across each transect the width of each band of ridge, intermediate and furrow was measured. A circular quadrat the same size as the flux collars (20 cm diameter) was placed in the centre of each band (n=63 in each plot). Each ridge-furrow complex typically consisted of two intermediate areas either side of the ridge and in these cases the quadrat was placed on the intermediate zone that was the wider. In each quadrat percentage cover was estimated for the main vegetation functional groups, these being defined as: *Eriophorum* species (almost entirely *Eriophorum vaginatum*); bare ground; heather (mainly *Calluna vulgaris*); other vascular vegetation; *Sphagnum* mosses; woodland mosses (defined as *Hypnum cupressiforme*, *Kindbergia praelonga*, *Pleurozium screberi* and *Polytrichum* species). In addition, the maximum height of heather was recorded in the collars; the number of individual *Eriophorum* leaves; the soil temperature inside the quadrat at 5 cm and 10 cm depth; and the volumetric soil moisture content at 0-6 cm depth. All the data collected in the transect quadrats were also concurrently collected inside the sampling point flux collars. When differences between the transect and the flux collars existed, it was assumed that the transects were more representative due to the greater sample size (a total of 126 survey points compared with 30 from all the flux collar sampling points).

4.3.3. Data analysis

Treatment effects on the fluxes and environmental variables

All statistical analyses were carried out in SPSS version 27 (IBM Corp., 2020). Data were first tested for normality using the one sample Kolmogorov-Smirnov test and for homoscedasticity using the Levene's test. When data did not meet one of these requirements for ANOVAs, the data were transformed using Box Cox transformations (Box and Cox, 1964; Teh et al., 2017). Before transformation if the data contained any zero or negative values a constant was added to all data so that the lowest value in the data set was 1. The transformation calculated a value for Lambda (λ) (table 4.3) which best approximated the normal distribution and applies the transformation:

Equation 1-

$$\frac{y^\lambda - 1}{\lambda}$$

Table 4.3: Value of λ for Box-Cox transformed independent variables of the two-way repeated measures ANOVA.

Dependent variable	λ
CH ₄	0.119
ER	0.151
GPP	4
NEE	3.470
Pore water [DOC]	0.310
Pore water [NH ₄ ⁺]	-0.294
Pore water [PO ₄ ³⁻]	-0.074
Soil moisture	2.660
Temp_5cm	No transformation required
Temp_10cm	0.620
Water table depth	No transformation required

Two-way repeated measures ANOVAs (Pillai Trace) were used to assess the effects of plot, microtopography and seasonal time point on the dependent variables, including all fluxes (CH₄, ER, GPP and NEE), pore water concentrations (DOC, NH₄⁺ and PO₄⁻) and environmental variables (soil moisture, water table depth, Temp_5cm, Temp_10cm). When significant interactions were found between time point and microtopography or plot the treatment effects were tested further with two-way ANOVAs for individual time points. When a significant effect of microtopography was identified post hoc Tukey's HSD pairwise tests were carried out. When a significant time effect was observed, within-subjects' contrasts were used to test for sequential pairwise comparisons of the seasonal time points levels.

The transect data were used to assess the representativeness of the flux collar positions with two-way ANOVAs. The effect of whether the data were collected from a transect/collar with plot and microtopographic level included as covariates was tested for on the data collected for soil moisture, Temp_5cm, Temp_10cm and the vegetation cover. For ANOVAs where there was

not a significant difference between transect or collar it was assumed that the collars were representative of the plot for that variable.

Annual flux estimate of NEE and CH₄ in CO₂ equivalents

Linear interpolation using the `seq()` function in R (R Core Team, 2021) was used to estimate CH₄ flux, ER and NEE for each day over the course of a year starting on October 22nd 2019 and finishing on October 21st 2020. For the purpose of this estimate the May 2021 data point was treated as if it had been measured in May 2020. All the measurements used were daytime measurements. To account for the fact that GPP is 0 at night and therefore NEE at night is equal to ER, NEE for each day was calculated as follows:

Equation 2-

(Estimated NEE * proportion of daylight hours) + (Estimated ER * proportion of night-time hours)

NEE for each day were then summed to give a total annual estimate of NEE. Methane was converted to CO₂ equivalent by multiplying by 28 according to the CO₂eq over 100 years as calculated by (Myhre et al., 2013). Then, CH₄ in CO₂ equivalents and NEE were added together to give a total annual budget for each plot. Two-way ANOVA was used to test for effect of plot and microtopography on estimated annual fluxes from each collar. Tukey's pairwise test was used to test for significant differences between different microtopographic levels if there was a significant effect of microtopography.

Diurnal variation

Gas fluxes were modelled with linear mixed effect models using the 'nlme' package in R (Pinheiro et al., 2021). The following variables were included in the models as fixed effects: time of day (as a categorical variable, e.g. morning, afternoon, early night, late night); microtopographic level of the collar; whether the collar is in the wooded or open plot. Collar ID was included as a random effect to account for the repeated measures from the same collars. The interaction between the fixed variables was also tested for but were not significant so were left out of the final model. Gas

fluxes were Box-Cox transformed as the original models' residuals because the data were normally distributed as tested by Shapiro–Wilk tests.

Linear regression of fluxes against environmental variables

Linear regression was used, to identify potential key environmental controls on CH₄ emission, NEE, GPP and ER. As per the repeated measures ANOVA the data was Box-Cox transformed if it violated the assumptions of normality or homoscedasticity. The following environmental variables were modelled separately against CH₄, NEE, GPP and ER: plot, microtopographic level, water table depth, soil moisture, soil temperature at 5cm and 10 cm depth, air temperature, PAR, green *Eriophorum* leaf count, *Sphagnum* species cover, woodland moss cover, birch leaf cover, heather cover, and maximum height of the heather.

4.4. Results

4.4.1. Spatial and temporal variability of gas fluxes in the wooded and open plots at Flanders Moss

Methane fluxes

Both plots were net emitters of CH₄ and the open plot was a significantly greater source of CH₄ than the wooded plot (F= 10.577, p=0.003), table 4.4, figure 4.3a & b. Mean CH₄ flux across all measurements in the wooded plot was 0.005 $\mu\text{mol m}^{-2} \text{s}^{-1}$ (95% CI= ± 0.0015) and 0.0123 $\mu\text{mol m}^{-2} \text{s}^{-1}$ (95% CI= ± 0.0026) across all measurements in the open plot. The repeated two-way ANOVA for CH₄ had a significant Pillai's Trace for the effect of seasonal time point (F=61.711, p<0.001). Mean CH₄ fluxes were lowest in winter months and increased in spring to a peak in August before reducing again (figure 4.3b). The sequential pairwise comparison of the seasonal time points showed that each pair was significantly different from the time point proceeding it, with the exception of the January and February 2020 measurements (Appendix 3).

Methane fluxes were greater from the furrow and intermediate collars than those on the ridges. There was a significant interaction between seasonal time point and microtopography (F=2.245, p=0.024). Ridge measurements showed less seasonal variability than intermediate and furrow measurements especially in the peak emissions months of July, August, and September

(figure 4.3c). To account for the interaction separate ANOVAs on the seasonal time points were carried out; these showed a significant difference between topographic levels in October 2019, August 2020, October 2020 and May 2021. Pairwise comparison of these dates showed fluxes from ridges were significantly lower than those from the furrows and intermediate microtopographic levels.

Carbon dioxide fluxes

ER ($F=0.047$, $p=0.831$), GPP ($F=0.310$, $p=0.583$) or NEE ($F=0.475$, $p=0.498$) did not differ significantly between the open plot and the wooded plot (figure 4.4, table 4.4). There was a significant seasonal effect on ER ($F=166.543$, $p<0.001$), GPP ($F=43.299$, $p<0.0001$) and NEE ($F=5.284$, $p=0.0002$). ER, GPP and NEE were lower in magnitude in winter months than in summer months. The increase in the magnitude of GPP was greater than ER from winter to summer so that NEE was more negative in summer than in winter (figure 4.4). Microtopography did not significantly affect ER ($F=2.868$, $p=0.076$) or GPP ($F=1.562$, $p=0.115$), but did have a significant effect on NEE ($F=4.258$, $p=0.026$) with mean flux having the lowest magnitude (least negative) on the ridges and highest on the intermediate (figure 4.4).

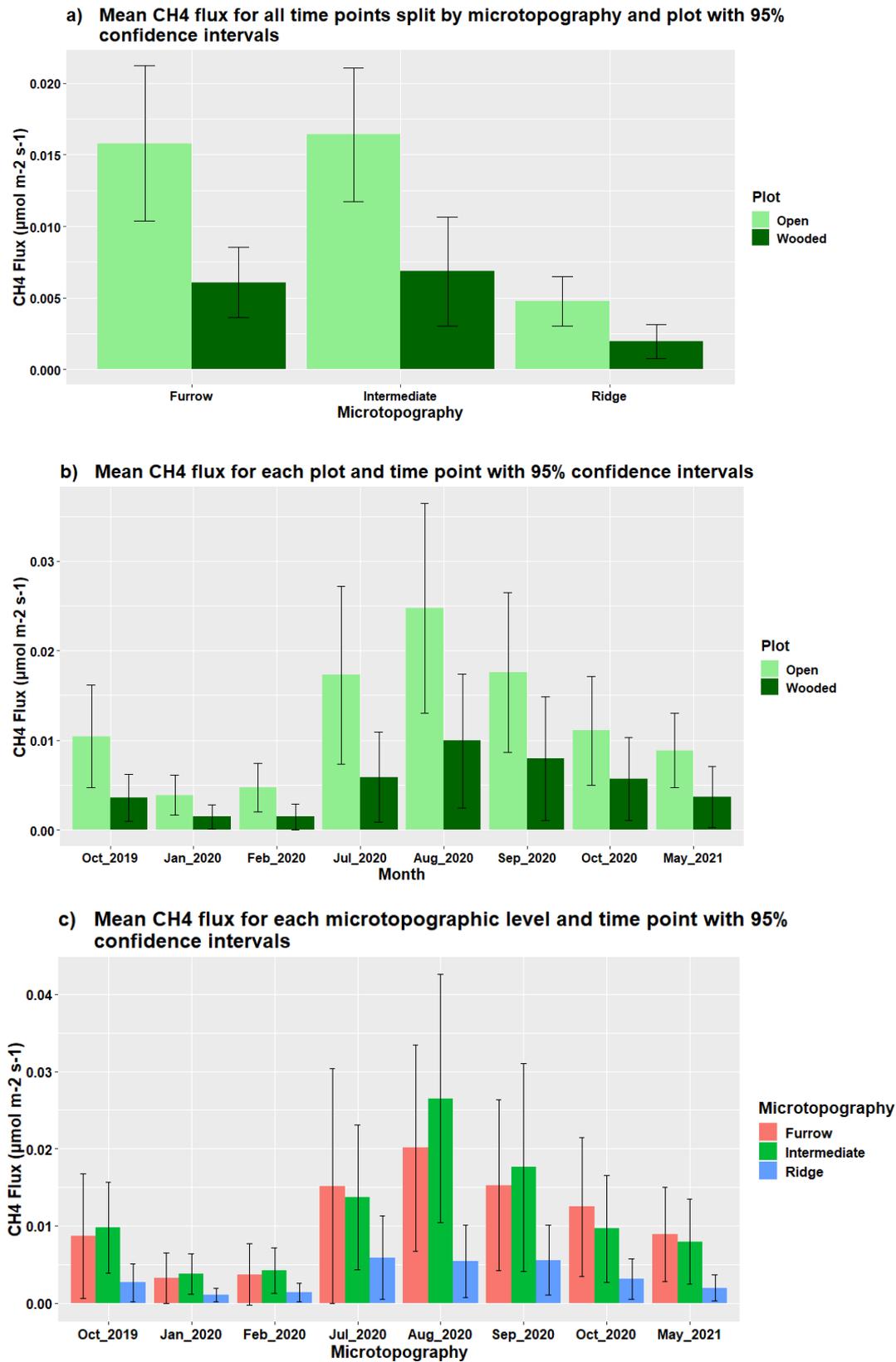


Figure 4.3: a) Mean CH₄ flux from all seasonal measurements with confidence intervals split by microtopographic level and plot. b) Mean CH₄ flux for each plot and time point. c) Mean CH₄ flux for each topographic level and time point

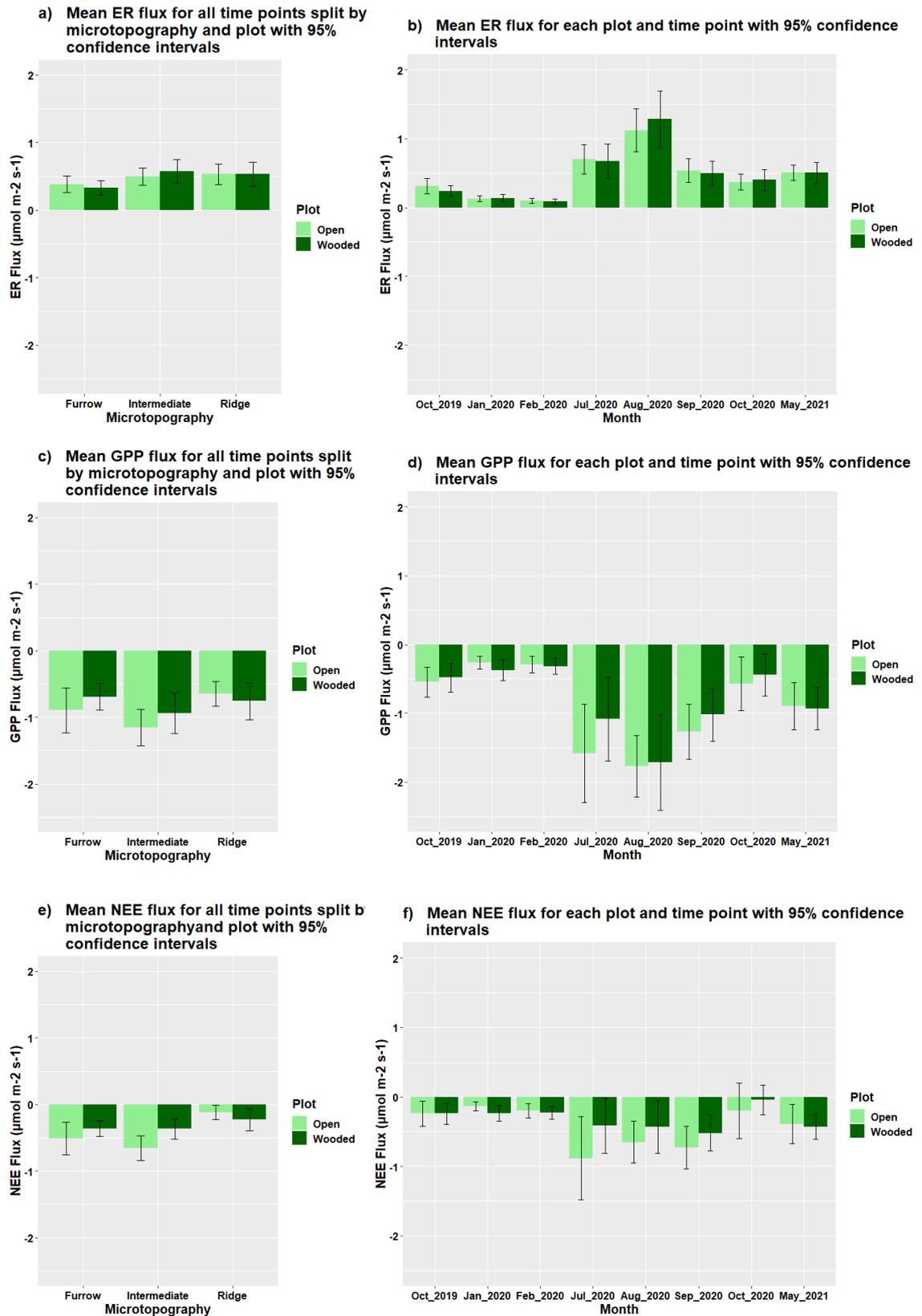


Figure 4.4: Mean ER, GPP and NEE with 95% confidence intervals for all seasonal measurements split by microtopographic level and plot in the first column of graphs (a, c, e) and split by plot and time point for the second column of graphs (b, d, f).

Table 4.4: *F* values with *p* values in brackets from the two-way repeated measures ANOVA for plot, microtopography, seasonal time point and their interactions. The effect of seasonal time point and the interaction of seasonal time point and other factors were assessed with Pillai's Trace test. Boxes highlighted in green indicate significant *p* values ($p < 0.05$).

Independent variable	Plot	Microtopography	Seasonal time point	Plot * Microtopography	Plot * Seasonal time point	Microtopography * Seasonal time point	Plot * Microtopography * Seasonal time point
CH ₄	10.577 (0.003)	3.553 (0.045)	61.711 (<0.001)	0.250 (0.781)	0.884 (0.538)	2.245 (0.024)	1.493 (0.161)
ER	0.047 (0.831)	2.868 (0.076)	166.543 (<0.001)	0.088 (0.916)	0.743 (0.639)	1.652 (0.109)	1.073 (0.410)
GPP	0.310 (0.583)	1.562 (0.115)	43.299 (<0.001)	0.321 (0.728)	0.927 (0.509)	1.316 (0.244)	0.825 (0.639)
NEE	0.475 (0.498)	4.258 (0.026)	5.284 (0.002)	0.996 (0.384)	1.165 (0.370)	1.585 (0.129)	0.824 (0.640)
Pore water [DOC]	0.089 (0.769)	1.310 (0.288)	93.112 (<0.001)	4.458 (0.271)	6.872 (<0.001)	3.770 (<0.001)	2.616 (0.009)
Pore water [NH ₄ ⁺]	0.000 (0.985)	1.117 (0.344)	4.803 (0.003)	3.459 (0.048)	1.553 (0.213)	1.477 (0.167)	1.224 (0.299)
Pore water [PO ₄ ³⁻]	0.082 (0.778)	4.151 (0.028)	25.899 (<0.001)	3.456 (0.048)	0.780 (0.612)	0.712 (0.749)	1.045 (0.434)
Soil moisture	8.802 (0.007)	10.358 (<0.001)	5.554 (0.002)	0.772 (0.473)	1.276 (0.316)	1.689 (0.099)	0.893 (0.572)
Soil temp_5 cm	0.212 (0.649)	4.373 (0.024)	576.627 (<0.001)	0.180 (0.836)	0.857 (0.557)	1.736 (0.088)	0.614 (0.836)
Soil temp_10 cm	0.584 (0.452)	2.349 (0.117)	1471.588 (<0.001)	0.089 (0.915)	2.458 (0.059)	2.880 (0.005)	0.689 (0.770)
Water table depth (cm)	11.017 (0.003)	89.807 (<0.001)	251.680 (<0.001)	0.137 (0.873)	0.645 (0.714)	1.598 (0.125)	1.122 (0.371)

4.4.2. Total annual carbon dioxide equivalents flux from the wooded and open plot

The estimate of annual NEE from both the wooded and open plot was negative, making both plots sinks for CO₂ (figure 4.5). Estimated annual NEE is not significantly different for the wooded plot and open plot (F=0.622, p=0.438). The magnitude of CH₄ emissions in CO₂ equivalents is greater than NEE for both the open and wooded plot, on average 3.4 times greater across both plots, making both plots net emitters of greenhouse gases. Taking CH₄ into account, the overall estimate of greenhouse gas emissions from the wooded plot was under half the emissions from the open plot and significantly different (F=6.863, p=0.0145). The plots remain significantly different for the annual estimate of combined CO₂ and CH₄ flux (F=4.528, p=0.043).

4.4.3. Diurnal variability

No statistically significant difference existed between the time that a flux measurement was taken and CH₄ flux in August (p=0.3019) or October (p=0.2108). Carbon dioxide fluxes do show diurnal variation with NEE (p<0.001), ER (p<0.001) and GPP (p<0.0001) all having a significant relationship between flux and time for the measurement in August, whereas in October only NEE (p=0.0023) and GPP (p<0.0001) vary significantly; ER does not vary between time points (0.6167). There is no significant interaction between the time of the measurement and the plot that the measurements were taken from for CH₄ (p=0.7405), NEE (p=0.6586), ER (p=0.8612) or GPP (p=0.5120).

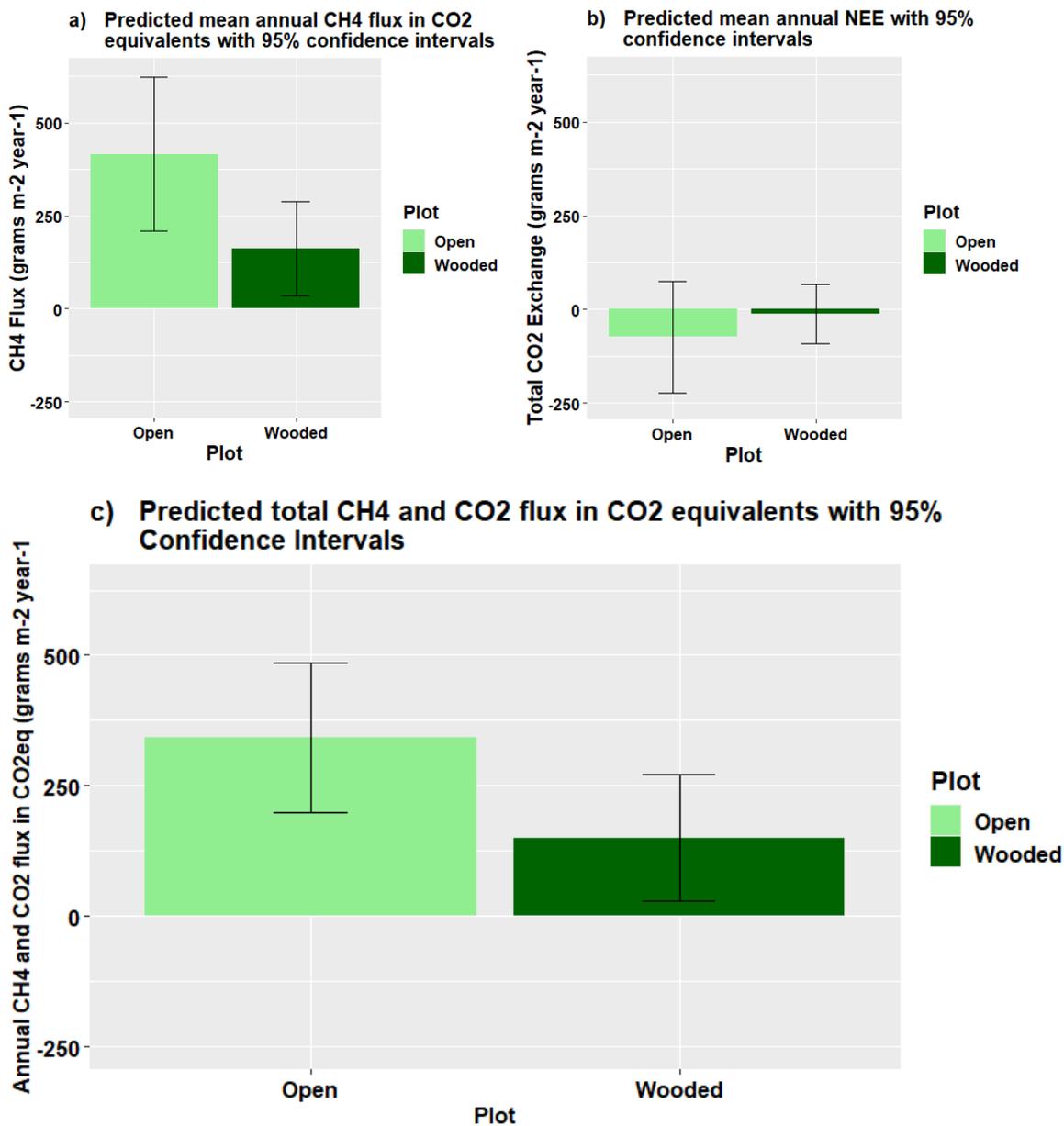


Figure 4.5: Estimates of mean annual gas fluxes in grams of CO₂ equivalents by linear interpolation of each flux collar. The predicted annual flux from each collar is split by plot. Magnitude of annual CH₄ flux in CO₂ equivalents is greater than annual NEE so that although plots have a mean annual NEE that is negative both plots are carbon sources. The 95% confidence intervals are included but are very wide, reflecting high variability between collars in each plot.

4.4.4. Spatial and temporal variability of peat pore water Dissolved Organic Carbon and nutrient concentration in the wooded and open plot

Dissolved Organic Carbon

Dissolved Organic Carbon concentrations were not significantly different between the wooded and open plot ($F=0.089$, $p=0.769$) or between topographic levels ($F=0.089$, $p=0.769$). The two plots displayed significant temporal variation ($F=93.112$, $p<0.001$) (Figure 4.6). All sequential pairwise comparisons were significant apart from September 2020 and October 2020 when DOC concentrations remained the same (Appendix 3). There was a significant two- and three-way interaction between seasonal time and plot/microtopography. Comparisons of the time point means indicate greater seasonal variability in the open plot, with mean winter DOC concentrations lower in the open plot than in the wooded plot but greater DOC concentration in the open plot than the wooded plot at other times of year (figure 4.6).

Nutrients

Seventy-eight percent of the peat pore water samples contain nitrate below the detection limit of the analysis and remaining data were low and did not show any obvious trends, so these data were excluded from further analyses. All the ammonium and phosphate samples were within detectable limits. The overall mean ammonium concentration was 0.106 mg/L (95% CI = ± 0.024) and the mean phosphate concentration was 0.043 mg/L (95% CI = ± 0.004).

Concentrations of NH_4^+ or PO_4^{3-} were similar between plots. There was also a significant interaction between plot and microtopographic level for both NH_4^+ ($F=3.469$, $p=0.048$) and PO_4^{3-} ($F=3.456$, $p=0.048$) concentrations. When one-way repeated measures ANOVA were carried out on data sets containing only one microtopographic level PO_4^{3-} concentrations were significantly higher in the open plot than the wooded plot for the intermediate level, and significantly higher in the wooded plot than the open plot for the ridge level. No significant differences were found between topographic levels for NH_4^+ . In the wooded plot PO_4^{3-} and NH_4^+ concentrations were significantly higher in the ridge level than the furrow (PO_4^{3-} : $p=0.009$, NH_4^+ : $p=0.021$) or intermediate levels (PO_4^{3-} : $p=0.35$, NH_4^+ : $p=0.020$) but there were no significant topographic effects for the open plot. Both NH_4^+ ($F=4.803$, $p=0.003$) or PO_4^{3-} ($F=25.899$, $p<0.001$) showed significant variation between repeated measurements; this variation didn't show a clear winter/summer difference (figure 4.7).

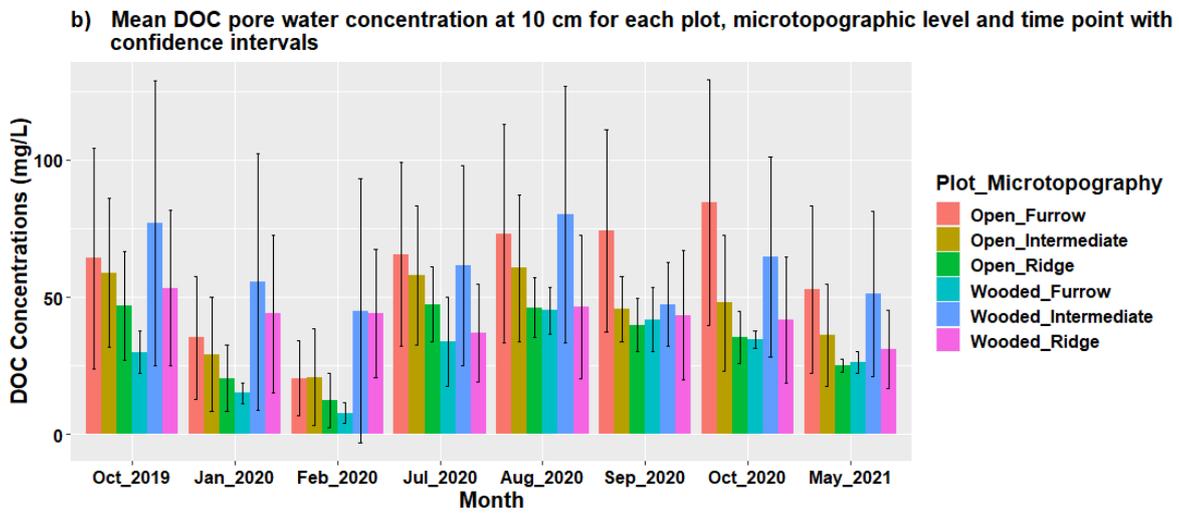
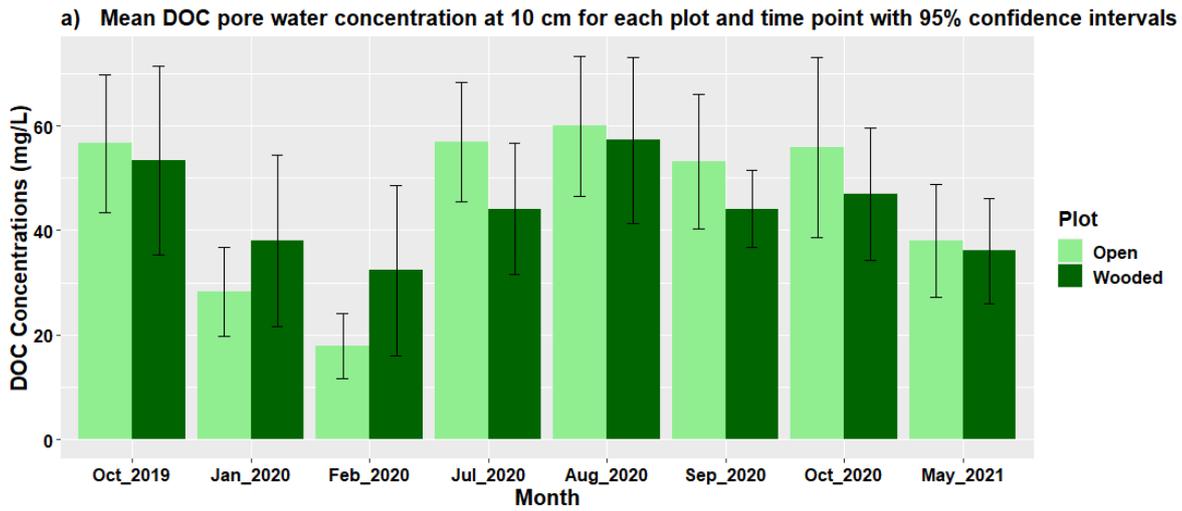


Figure 4.6: Mean DOC concentration with 95% confidence intervals: a) split by plot (wooded and open) and seasonal time point; b) split by plot, micro-topographical level, and seasonal time point.

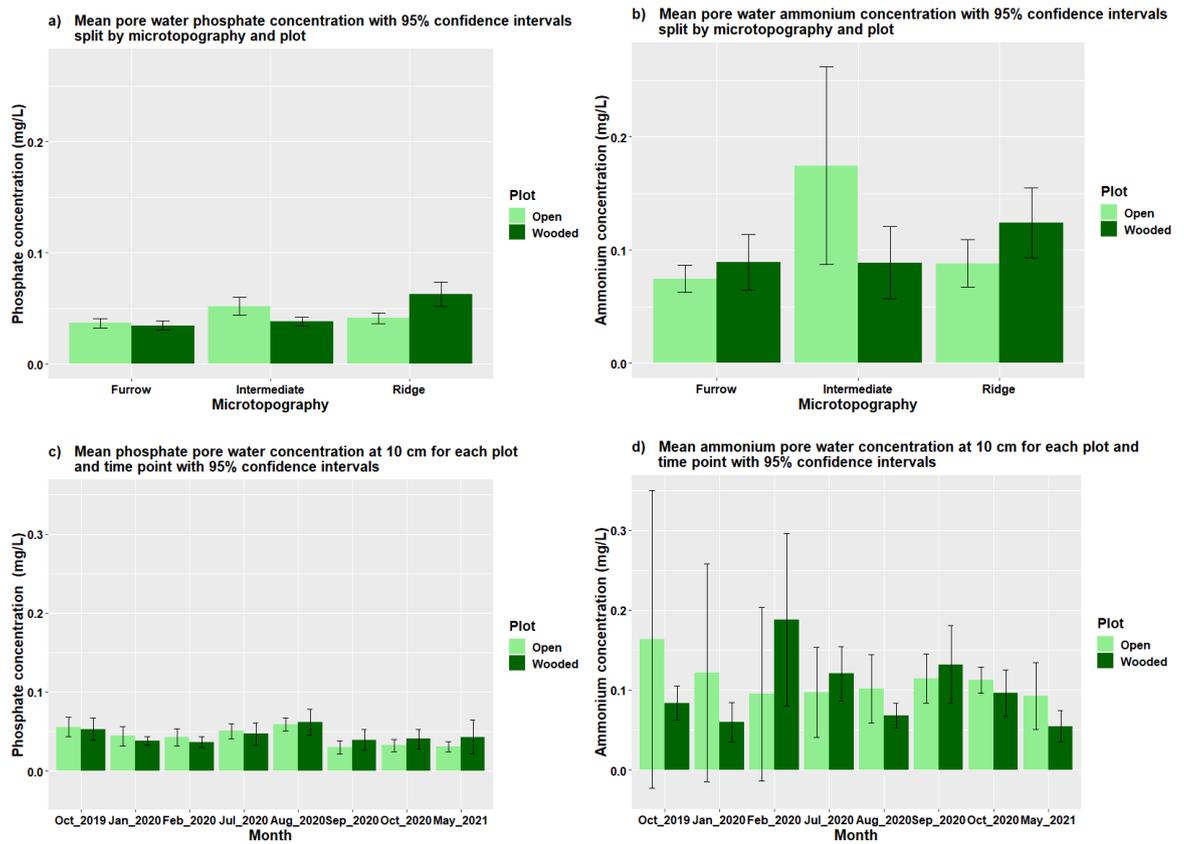


Figure 4.7: Concentrations of phosphate and ammonium in peat pore water at 10 cm depth split according to microtopographic level in a) & b) and seasonal time point in c) & d). The intermediate plot has significantly higher concentrations in the open plot than the wooded and the ridge has a significantly higher concentration of phosphate in the wooded plot than the open. This relative difference is the same for ammonium but with overlapping 95% confidence intervals.

4.4.5. Spatial and temporal variability of environmental variables and vegetation properties in the wooded and open plots

Abiotic environmental variables

The soil moisture content near the collars was significantly higher in the open plot than in the wooded (F=8.802, p=0.007) and water table depth was significantly closer to the surface (F=11.017, p=0.003) in the open plot than the wooded. Both metrics also show that for each plot the ridge was the driest microtopographic level, and the furrow the wettest microtopographic level, with the intermediate level having intermediate values (figure 4.8). The volumetric soil moisture content measured in the transects was similar to that measured in the collars.

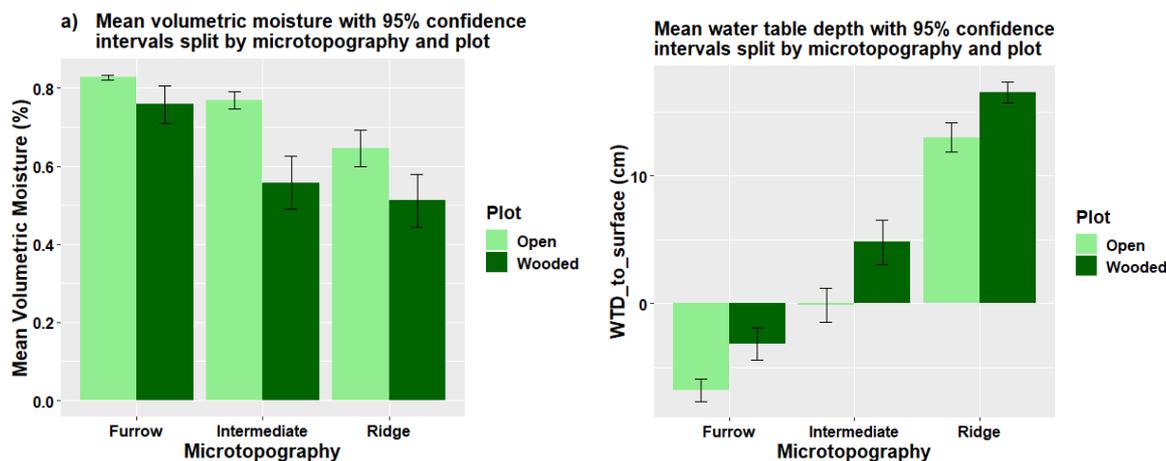


Figure 4.8: Mean water table depth and volumetric water with 95% confidence intervals split by microtopographic level and plot. Both plots show that for each topographic level the wooded plot is significantly dryer than the open plot. The plots also demonstrate that the furrow is the wettest topographic level and the ridge is the driest.

Temp_5cm and Temp_10 cm by the collars was not significantly different between plots but varied seasonally. Soil temperatures by the collars were similar to soil temperatures across the transect.

Vegetation

In both the transect quadrats and the flux collars more than 98% of *Eriophorum* leaves were *E. vaginatum* with the remaining being *E. angustifolium*; given the small size of this component *E. angustifolium* count was pooled with *E. vaginatum* as one *Eriophorum* count.

The transect quadrats showed that *Eriophorum* green leaf count in the flux collars was similar in the open (mean = 53) and wooded plot (mean = 49) ($p=0.7239$). The number of *Eriophorum* leaves on the ridges was significantly lower than on the furrows ($p=0.0120$) or intermediates ($p=0.0013$). There was a significant interaction between plot and microtopographic level ($p=0.0164$) with a more pronounced difference between the *Eriophorum* count on the ridges and furrow/intermediate levels in the open plot than the wooded plot.

The flux collars in the tree plot did not contain significantly different numbers of *Eriophorum* leaves compared to the transects ($p=0.6430$). However, the open plot flux collars contained significantly more *Eriophorum* leaves than the transect points ($p<0.001$), with the mean *Eriophorum* green leaf count in the collar being 71 leaves compared with 53 leaves in the transects (25% less than the transect).

The transects show that there was not a significant difference in the cover of heather ($p=0.6922$) and *Sphagnum* species ($p=0.3841$) between the plots and the cover in the collars and the transects in both plots was not significantly different. There was however significantly greater woodland moss cover in the wooded plot than the open plot ($p=0.0165$). Heather and woodland moss cover was significantly greater on the ridge and intermediate levels than the furrow, and significantly greater on the ridge than the intermediate level in both plots. *Sphagnum* species cover was significantly greater in the furrow and intermediate levels than the ridge and significantly greater in the furrow than the intermediate level in both plots. There was no significant difference in the cover of either heather, *Sphagnum* species or woodland mosses between the flux collars and the transects. There were significantly more fallen birch leaves in the tree plot than the open plot ($p<0.001$).

4.4.6. Relationship between environmental variables and fluxes

The single variable which explained the most variance of all CH₄ fluxes was *Eriophorum* leaf count ($R^2 = 0.423$) (Table 9). *Eriophorum* leaf count also explained a reasonable proportion of variance for GPP ($R^2 = 0.152$), ER ($R^2 = 0.118$) and NEE ($R^2 = 0.114$). Cover of other vegetation types could

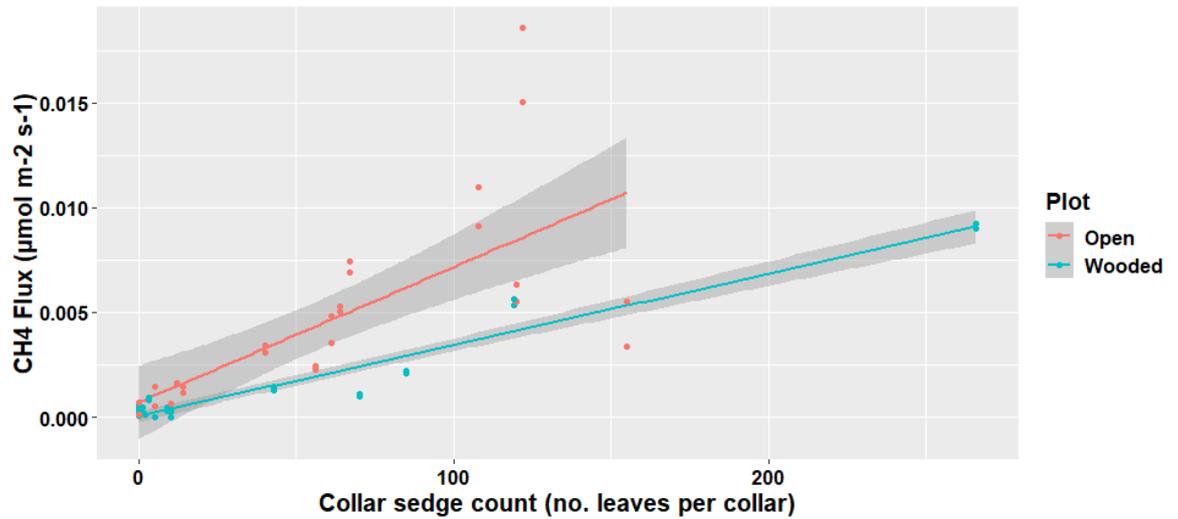
only explain a limited amount of the variance in gas fluxes. The predictive power of *Eriophorum* count was increased if CH₄ flux measurements were split by plot and seasonal time points ($R^2 \sim 0.5-0.7$). Both plots have a positive relationship between *Eriophorum* count and CH₄ flux but the slope of the relationship for the open plot is steeper than that for the wooded plot, figure 4.9.

Soil moisture had reasonable predictive power ($R^2 = 0.173$). PAR explained the most variance out of all other environmental variables for GPP ($R^2=0.388$) and NEE ($R^2=0.275$). Temperature explained a reasonable proportion of variance of CH₄ flux, GPP and ER (Table 4.5). The variance of ER was particularly explained by variance in air temperature ($R^2 = 0.558$), Temp_5cm ($R^2 = 0.542$) and Temp_10cm ($R^2 = 0.544$)

Testing effect of 25% reduction CH₄ in open plots

Since the open plot flux collars contained significantly more *Eriophorum* leaves than the transects indicated was representative (mean leaf count was 25% higher in the flux collars), and since *Eriophorum* leaf count was also found to be strongly related to CH₄ flux, the repeated measure ANOVA was repeated but with a 25% reduction of CH₄ all fluxes from the open plot. This repeated measure ANOVA still showed that CH₄ fluxes were significantly higher in the open plot than the wooded plots ($F=6.529$, $p=0.017$).

a) Winter CH₄ flux plotted against collar sedge count with shaded areas showing 95% confidence intervals



b) Summer CH₄ flux plotted against collar sedge count with shaded areas showing 95% confidence intervals

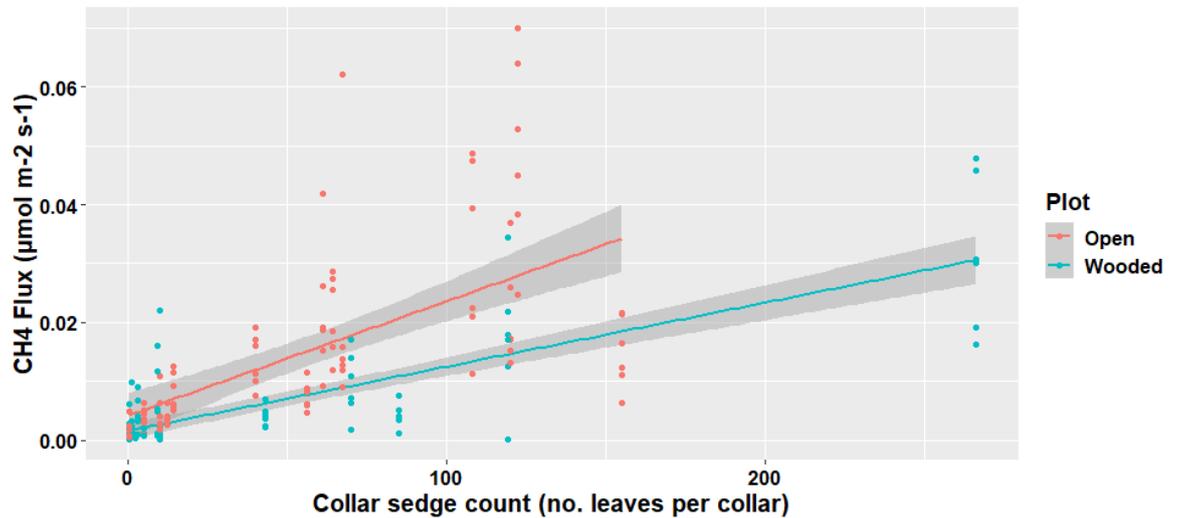


Figure 4.9: *Eriophorum* leaf count is a powerful predictor for CH₄ flux. The same *Eriophorum* leaf count predicts a greater CH₄ emission in the open plot than in the wooded plot. The data is presented as: a) winter (January 2020 and February 2020) and b) summer (October 2019, July 2020, August 2020, September 2020, October 2020 & May 2021) measurements.

Table 4.5: Single variable regression between gas fluxes and potential explanatory variables. The first number is the R^2 value of the regression, in brackets is F , the p -value associated with the F test and the value of lambda used for the Box-Cox transformation. Cells highlighted in light green have a significant p value, those in dark green have a significant p value and an R^2 of 0.15 or higher.

Explanatory variable	CH ₄	GPP	ER	NEE
Plot	0.174 (F=50.03, p<0.001, λ=0.101)	0.001 (F=0.19, p=0.665, λ=0.222)	0.000 (F=0.10, p=0.751, λ=0.141)	0.008 (F=1.83, p=0.177, λ=2)
Microtopography	0.118 (F=15.79, p<0.001, λ=0.141)	0.038 (F=4.71, p=0.010, λ=0.222)	0.051 (F=6.39, p=0.002, λ=0.141)	0.081 (F=10.46, p<0.001, λ=2)
Water table depth (cm)	0.118 (F=31.91, p<0.001, λ=0.141)	0.008 (F=1.822, p=0.178, λ=0.222)	0.072 (F=18.38, p<0.001, λ=0.141)	0.036 (F=8.95, p=0.003, λ=2)
Soil moisture (%)	0.173 (F=4.20, p=0.042, λ=0.141)	0.005 (F=1.26, p=0.262, λ=0.222)	0.036 (F=9.01, p=0.003, λ=0.141)	0.008 (F=1.99, p=0.160, λ=2)
Air temperature (°C)	0.123 (F=29.24, p<0.001, λ=0.101)	0.361 (F=117.6, p<0.001, λ=0.263)	0.558 (F=262.9, p<0.001, λ=0.263)	0.086 (F=19.47, p<0.001, λ=2)
Temp_5cm (°C)	0.111 (F=29.84, p<0.001, λ=0.101)	0.246 (F=77.5, p<0.001, λ=0.263)	0.542 (F=281.3, p<0.001, λ=0.182)	0.028 (F=6.86, p=0.009, λ=2)
Temp_10cm (°C)	0.109 (F=35.46, p<0.001, λ=0.101)	0.246 (F=77.83, p<0.001, λ=0.263)	0.544 (F=284.4, p<0.001, λ=0.182)	0.034 (F=8.40, p=0.004, λ=2)
PAR (μmol m ⁻² s ⁻¹)	0.036 (F=8.94, p=0.003, λ=0.141)	0.388 (F=150.6, p<0.001, λ=0.263)	0.213 (F=64.38, p<0.001, λ=0.222)	0.275 (F=90.34, p<0.001, λ=2)
<i>Eriophorum</i> (leaf count)	0.423 (F=174.20, p<0.001, λ=0.182)	0.152 (F=42.52, p<0.001, λ=0.222)	0.118 (F=31.86, p<0.001, λ=0.182)	0.114 (F=30.69, p<0.001, λ=2)
<i>Sphagnum</i> (% cover)	0.055 (F=13.86, p=0.051, λ=0.101)	0.006 (F=1.38, p=0.242, λ=0.222)	0.070 (F=17.83, p<0.001, λ=0.141)	0.028 (F=6.90, p=0.009, λ=2)
Woodland Moss (% cover)	0.109 (F=29.16, p<0.001, λ=0.141)	0.003 (F=0.749, p=0.388, λ=0.222)	0.012 (F=3.01, p=0.084, λ=0.141)	0.062 (F=15.73, p<0.001, λ=2)
Heather (% cover)	0.016 (F=3.76, p=0.054, λ=0.141)	0.050 (F=12.46, p<0.001, λ=0.222)	0.056 (F=14.11, p<0.001, λ=0.141)	0.031 (F=7.65, p=0.006, λ=2)
Heather (max height)	0.007 (F=1.64, p=0.201, λ=0.101)	0.018 (F=4.29, p=0.039, λ=0.222)	0.055 (F=13.88, p<0.001, λ=0.141)	0.001 (F=25, p=0.621, λ=2)

4.5. Discussion

4.5.1. Effects of forest-to-bog restoration on climate forcing

Twenty-two years after restoration work, both plots are estimated to be net sinks for CO₂ and net sources for CH₄. However, the strength of the CO₂ sink is not sufficient to compensate for the strength of the CH₄ source meaning the sites are currently having a net climate warming effect. Methane emissions in CO₂eq on average have a greater magnitude than CO₂ flux (3.4 times greater for the annual estimate). As such CH₄ emissions exceed the CO₂ sink effect to make the plots net emitters when CO₂ and CH₄ are accounted for simultaneously. The restoration of plots as sinks for CO₂ 22 years after restoration is consistent with previous work (Hambley et al., 2018; Lees et al., 2019; Creevy et al., 2020). The high CH₄ emissions found in this study are contrary to Creevy et al. (2020) results which estimate CH₄ emissions account for less than 20% of NEE. However the results are similar to Hermans (2018) who found that as sites got older their CH₄ fluxes tend to near-natural concentrations which in that study were estimated as 459 gCO₂eq m² yr⁻¹; this is similar to the 415 and 162 gCO₂eq m² yr⁻¹ estimated for the open plot and wooded plot respectively.

Forest-to-bog restoration is often proposed as a policy to prevent LULUCF greenhouse gas emissions. For example, a recent Scottish policy briefing on forest-to-bog restoration suggests net climate cooling will be achieved from a typical project after 15-20 years (Hermans et al., 2019). It is therefore problematic that the restoration sites studied are remaining sources of greenhouse gases as this will be counter-productive for reducing national greenhouse gas emissions. However, the trajectory of the site is unknown and it may be that CH₄ emissions are falling over time at the site so that the area is on a trajectory to having a net cooling effect at longer time scales. Furthermore, the forest-to-bog restoration method used at the sites studied is a relatively low intensity old method. A range of newer, higher intensity restoration methods are increasingly commonly used, for example ground smoothing, where the microtopography created through ploughing is removed. These methods may result in the establishment of net cooling systems faster than older methods; however, the author knows of no flux studies on the effects of the most recently developed higher intensity restoration methods. There is some data on the short term effects of new restoration methods on hydrology and pore water chemistry, such as evidence that mulching increases DOC and nutrient concentrations in pore water relative to other restoration methods in the short term (Howson et al., 2021a). In general higher intensity

restoration methods may increase initial carbon losses due to increased disturbance during restoration work, but are hoped to result in faster recovery in the long term (Gaffney et al., 2022).

Ecosystem Respiration and GPP were similar between the plots and microtopographic levels but NEE varied significantly between microtopographic levels, with the ridges being the weakest CO₂ sinks and the intermediates and furrows being the strongest CO₂ sinks. This significant effect is a combination of a trend in the ER data for higher fluxes from the ridges with a higher GPP from the furrows and intermediate levels. So, although neither of these trends were significant by themselves, in combination they have caused a significant effect. The GPP differences may be due to the different vegetation types which dominated each microtopographic level. Water table was lower on the ridges and this may have increased ER given that oxidative decay of the peat is facilitated by lower water tables (Evans et al., 2021). However, this study cannot actually be used to distinguish what proportion of ER was derived from the peat (heterotrophic fluxes). This would require collars to have been installed which had the vegetation removed from them.

The microtopographic variation in CH₄ is likely due to the significant water table difference between the topographic levels. There is substantial seasonal variation in CO₂ and CH₄ fluxes, with both being lowest in winter and peaking in summer, consistent with previous peatland research (Laine, 2006). The two treatment plots did not respond differently to seasonal variation.

4.5.2. Effects of restoration treatment and growth of trees on climate forcing

The wooded plot had lower CH₄ fluxes but not significantly different CO₂ fluxes from the open plot. The wooded plot would traditionally be thought of as more poorly restored than the open plot. Yet, from a greenhouse gas perspective, the wooded plot is currently estimated to be a CO₂ sink while also minimising CH₄ emissions relative to the open plot. The reasons for this have not been conclusively identified but may be due to the slightly dryer conditions under the birch canopy. Other research has identified that if water table is near the surface of a peatland (within ~10 cm) further rises in water table can result in large increases in CH₄ emissions but only small reductions in CO₂ emissions, meaning that greenhouse gas emissions might be maximised at water tables slightly below the peat surface (Evans et al., 2021). It is plausible that the slight drying effect from the trees in the wooded plot has reduced the water table closer to this

optimum. It could therefore be that in some contexts managing forest-to-bog restoration plots so that water table is kept slightly lower might minimise greenhouse gas emissions.

That tree cover reduces CH₄ emissions from deep peat has also been observed by Mazzola (2022). However tree cover is usually associated with increased CO₂ emissions from the peat surface (Hermans, 2018; Friggens et al., 2020; Mazzola et al., 2022). Although not significant, the wooded plot was a smaller CO₂ sink than the open plot so it may be that the presence of trees is reducing the plot's capacity to be a carbon sink. It may also be that the trees are currently not large enough to be significantly impacting on the CO₂ sink function of the peatland as flux studies on afforested peatlands generally focus on larger trees. It is unclear what trajectory the site will have in terms of changes in tree growth. The author observed no evidence of recent regeneration, so tree density is not expected to increase. All the trees were stunted and there was evidence that some trees were dying or had recently died but other trees appeared relatively healthy and actively growing. With this uncertainty over the rate of tree growth it is hard to predict the impact of greenhouse gas fluxes.

4.5.3. Effects of restoration treatment on environmental variables

Dissolved Organic Carbon and nutrients

There was no effect of restoration treatment on pore water DOC, ammonium or phosphate concentration at 10 cm depth 22 years after the restoration activities. Potentially the plot where the brush was left may have had higher DOC and nutrient concentrations in the years immediately after restoration as woody residuals being left on site has been observed to cause a short term rise in these concentration (Gaffney et al., 2018; Howson et al., 2021a). However, if this was the case the effects appear to have subsided and the subsequent tree growth does not appear to be affecting DOC, ammonium or phosphate pore water concentration at 10 cm depth. The DOC mean concentration from both plots was 45 mg L⁻¹, this is slightly smaller than the median concentration at a near-natural part of the same raised bog complex measured in another study (~70 mg L⁻¹ between 10-20 cm depth) (Howson et al., 2021a) which implies that pre-forestry pore water DOC concentrations have been restored. The non-significant difference between pore water DOC concentrations at 10 cm depth gives confidence that the fluvial losses of DOC from both plots would be similar, especially since pore water closer to the surface is usually the last to recover after restoration (Gaffney et al., 2018; Howson et al., 2021a). However actual fluvial

carbon losses would depend on the rate at which water was drained out of each plot and then upon the stability of the DOC as this determines whether it will be broken down to CO₂. No data was collected that could quantify these effects, so the result is only an indication that there is no difference between the treatments.

Water table and surface peat volumetric water content

The furrows had both the highest soil volumetric moisture content and water table, whereas the soil volumetric moisture content and water table was lowest in the ridges. This pattern in peatland microtopography is well documented (e.g. Hermans, 2018). At each microtopographic level volumetric water content and water table was lower in the wooded plot. From the data collected in this study the extent to which the wooded plot may be drier due to the brash being left, the birch establishment or natural difference between the plot could not be determined. However, older data collected by Anderson (2010) showed that the wooded plot was wetter than the open plot in the first two years after felling, only beginning to become drier years later. This is evidence that the tree plot is not naturally drier than the open plot and evidence that the brash – at least in the short term – was not having a drying effect. Conversely, that the trees might be responsible for the lowering of the water table is supported by this data since the wooded plot only became drier subsequent to establishment of the trees. The trees may be causing a drying effect due to evapotranspiration and interception as observed in other studies (Limpens et al., 2014).

The water table depth difference between the plots may explain the difference in CH₄ fluxes between the two plots. Water table depth has been identified a main determinant of CH₄ fluxes (Thayamkottu et al., 2021), and small changes in water table depth near the surface can have a large effect on CH₄ flux (Evans et al., 2021).

Vegetation

The difference in tree growth in the two plots is quite extreme. Although there is no evidence of nutrient enrichment in the wooded plot in this study's data there could plausibly have been nutrient enrichment from the brash at earlier stages. Previous studies have shown that initially raised nutrient concentration can fall in the years proceeding restoration (Gaffney et al., 2018). Nutrient concentrations being raised at earlier stages can allow birch trees to grow better: Tomassen et al. (2004) show that nitrogen availability was more important to the limiting of birch

colonisation than water table depth. The brash being left may also have deterred grazers such as deer, whereas deer may have more freely grazed on birch seedlings in the open plot. Grazing pressure from deer may also be a factor in explaining the extreme difference between the plots; if the advantages of growing in the wooded plot were enough to get the trees above grazing height these trees would continue to grow, causing the canopy present today. By contrast if regeneration never grew beyond grazing height in the open plot the trees would be kept suppressed by grazing. A recent estimate of deer density in the Flanders Moss area has been predicted a density of 12-18 deer per km² and this has thought to have been fairly constant for at least the last 8-10 years (Clements, 2021). This is a density which would be expected to have a substantial impact on the ability for woodland to regenerate (Putman et al., 2011).

There was significantly greater cover of heather at all topographic levels in the open plot compared to the wooded plot. This is consistent with the decline of heather cover during succession from heather moorland to birch woodland documented by Hester et al. (1991), which attributed the change to shading. The ridges in the wooded plot had significantly greater *Eriophorum* species leaf count and woodland moss cover than the ridges in the open plot. The woodland moss cover may be explained by the significantly drier conditions found in the wooded plot. Conversely, however, it would be expected that under drier conditions *Eriophorum* species would be less abundant (Hill et al., 2004). The author observed that *Eriophorum* species were less abundant under or near heather canopies – potentially the reduced heather cover on the ridges in the wooded plot may explain the relatively greater coverage of *Eriophorum* species on this topographic level.

The data for both the wooded and open plots broadly show a similar microtopographical pattern as was observed in previous studies (Anderson and Peace, 2017; Hancock et al., 2018). For example, *Sphagnum* species typically require wetter conditions and heather requires dry conditions to grow (Hill et al., 2004, 2007) and so the relative abundance of *Sphagnum* species in the wet furrows, heather's abundance on the dry ridges, and the presence of both vegetation types on the intermediate microtopography reflects this gradient.

The transect-flux collar vegetation comparison showed that the collars were broadly representative of the vegetation in the plot. However, the mean number of *Eriophorum* leaves in the transect quadrats was 75% of that in the flux collars; this difference was significant and therefore evidence that the collars were not representative for *Eriophorum* cover. Previous work (Creevy et al., 2020; Thayamkottu et al., 2021) highlights the significance of *Eriophorum* in CH₄ fluxes. The data from this study agrees with previous work that increased *Eriophorum* leaf count

was associated with increased CH₄ emissions and was the only vegetation type recorded to have a major influence on any of the gas fluxes. However, even when open plot CH₄ emissions were reduced by 25% their mean was still almost double that of the open plot (and still with a significant p-value) which indicates that the misrepresentation of *Eriophorum* presence does not explain the CH₄ emissions difference between the plots. Furthermore, the analysis showed that for a given density of *Eriophorum* leaves the CH₄ flux was estimated to be greater in the open plot than the wooded plot.

4.5.3. Limitations of the study

One spatial replicate

There is no true spatial replication in this study for comparing the effect of the wooded and open treatments as there is only one wooded and one open plot. As such, the 15 collars in each plot therefore serve as replicates below the plot level. This is a common constraint in many flux studies due to practical constraints (Haddaway et al., 2014), however caution is needed while generalising the results of this study at the main treatment level.

Approach to gas flux measurements

A fan was used in the flux chamber used for data collection in this study since there was no evidence that this affected magnitude of flux, and fan presence did improve the linearity of the measurements. However, there is some debate over whether flux chambers should be used with or without internal fans. Proponents of fan use argue that the use of a fan facilitates even mixing in the chamber which can improve accuracy and R² values (Christiansen et al., 2011). However, the use of fans is also opposed by some flux studies which consider fan use to alter fluxes due to disruption to boundary layers or pressure differentials between the inside and outside of the chamber (Davidson et al., 2002; Pumpanen et al., 2004). Some researchers see a flux measured with no fan as being a theoretically valid 'diffusive flux' by contrast to fluxes measured with an arbitrary level of fan-induced air movement (Redeker et al., 2015).

This study does not account for nitrous oxide (N₂O) fluxes which can be an important greenhouse gas in peatland systems (Lohila et al., 2011). Previous forest-to-bog research has indicated that N₂O does not have a significant effect on the greenhouse gas balances of

restoration projects (Hermans, 2018). Furthermore, ammonium availability is an important factor in N_2O production (Cowan et al., 2019), and this study shows no difference in pore water concentration of ammonium.

Diurnal variation and effects on linear interpolation

The diurnal data shows that there is no reason to doubt the validity of modelling annual fluxes at this site on daytime only CH_4 flux measurements as there is no evidence for diurnal variation in this gas. Diurnal variation in CH_4 is less well studied and less well understood than CO_2 . This study is the first study of diurnal variation with chamber techniques on Scottish forest-to-bog restoration projects. Previous studies on other peatlands have previously shown CH_4 to increase (Waddington et al., 1996; Dooling et al., 2018), decrease (Waddington et al., 1996; Bäckstrand et al., 2008) or remain unchanged during night-time measurements (Bäckstrand et al., 2008). Diurnal variation in ER was detected on the August measurements with measurements being lower at night, however this is not observed in the October data.

If ER is lower at night than during the daytime, the estimates of annual ER using the interpolation method based on daytime only measurement would overestimate ER and therefore also underestimate NEE. GPP also showed significant diurnal variation. The difference between daytime and night-time GPP was accounted for in the calculation of NEE for the interpolation method by assuming that night-time NEE would equal ER. However, the data showed that GPP can also vary significantly between daytime measurements which is not accounted for. Daytime flux measurements were generally taken a few hours after sunrise and finished a few hours before sunset, so underrepresenting times of day with lower PAR and therefore lower GPP and NEE correspondingly. The interpolation method may therefore have overestimated GPP and ER. Using a more complex model that accounts for difference in PAR such as that used by Creevy et al. (2020) would be expected to improve the accuracy of the annual estimate predictions. However, the interpolation results show that the combined CH_4 and CO_2 fluxes are dominated by CH_4 , so the overall interpretation of the final results is unlikely to have been affected.

Collecting diurnal fluxes from more collars at each topographical level and collecting light and dark measurements to allow comparisons in NEE and GPP was prioritised over higher frequency repeat measurements. This study is consequentially limited both in terms of the number of repeat measurements in a campaign and the number and annual spread of the campaigns. Increasing the number of repeat measurements and carrying out diurnal

measurements on more days would improve the statistical power and confidence in the results, and allow further investigation into the drivers of diurnal change. More campaigns, especially in winter and spring, as well as under different weather conditions in summer, would give better ability to generalise the results. Despite these limitations this study provides contextualising information for the annual budgets calculated in Chapter 4 and expands understanding of diurnal variation of CO₂ and CH₄ fluxes to peatland sites undergoing forest-to-bog restoration.

4.5.5. Policy implications

How peatland is managed is a critical component of reducing emissions from LULUCF. Large areas of peatland in Scotland are undergoing restoration work and there is increasing investment in restoring further areas of peatland. This study focused on a 22-year-old forest-to-bog restoration site on a lowland raised bog that used relatively low intensity methods (i.e. tree felling and main drain blocking). The site had been restored as a sink for CO₂, securing its long-term carbon store, but continued to have a net warming effect due to high CH₄ emissions. This is contrary to evidence summarised in the most recent Scottish policy briefings on forest-to-bog restoration carried out with similar low-intensity techniques, which suggest that projects will typically be net cooling after 15-20 years (Hermans et al., 2019). However the result maybe consistent with predictions for severely degraded Scottish peatland sites taking 20-50 years to have a net cooling effect (Artz et al., 2012b).

There is debate over how CH₄ should be compared with CO₂. The data analysis in this study is based on the global warming potential over 100 years of 28 times that of CO₂ as quoted in IPCC (2013). However, this measure is only an estimate of the effect and excludes the climate-carbon feedbacks that would make it 36 times. Furthermore, the use of the 100-year time scale although fairly standard is ultimately arbitrary. Several authors have argued that since peatlands accumulate and store carbon over thousands of years using a longer time scale such as 500 years may be more appropriate (Artz et al., 2012b; Nugent et al., 2019). Measuring on such a time scale would reduce the warming potential of CH₄ as it is relatively short lived in the atmosphere; the global warming potential over 500 years is predicted to be less than a third than it would be over 100 years (IPCC, 2007). If this global warming potential was applied to the data in this study then although the plots would still be having a net warming effect this would be much smaller and of almost a similar magnitude as the size of the CO₂ sink of the open plot. By contrast, the case could also be made that given the urgency of the climate crises and the speed at which nations are

planning to combat it, using shorter warming potentials is more relevant to management decisions.

Decisions over the future management of afforested peatlands are complex, and greenhouse gas balance is only one of several considerations. However, on greenhouse gas balance alone this study provides evidence that forest-to-bog restoration doesn't necessarily restore a net zero or negative greenhouse gas balance, even after 22 years. It is speculated that newer techniques for forest-to-bog restoration such as whole tree mulching and reprofiling/ground smoothing which removes the ridge furrow topography may result in a faster recovery of greenhouse gas fluxes, but data is lacking on this. Howson et al (2021) find that at least in the short-term DOC is higher from sites that are mulched. Methane fluxes were approximately three times less from the ridges than the furrows and intermediate microtopographies. Ground smoothing would remove these topographies so there shouldn't be areas as dry as the ridge or as wet as the furrow. This may reduce CH₄ fluxes from the areas that would otherwise be furrows but increase them from areas which would have been ridges. At this stage there is not enough research to understand the overall effect this would have on greenhouse gas emissions over time.

The wooded plot had lower CH₄ fluxes but not significantly different CO₂ fluxes from the open plot. The wooded plot would traditionally be thought of as more poorly restored than the open plot. However, from a greenhouse gas perspective this plot is currently estimated to be a CO₂ sink while also minimising CH₄ emissions relative to the open plot. The reasons for this have not been conclusively identified but it may be due to the slightly dryer conditions under the birch canopy. This suggests that managing some forest-to-bog sites in such a way that restoration has a slight drying effect may improve their value as carbon stores. Accepting a degree of tree regeneration on forest-to-bog restoration sites has been proposed as an appropriate management in some situations by Scottish Forestry, termed as Peatland Edge Woodland (Forestry Commission Scotland, 2015). The finding could be taken as evidence in support of allowing some tree regeneration on restoration sites if the primary objective for the site is to minimise greenhouse gas emissions. However, there is still long-term uncertainty over the outcome of letting birch regeneration establish on peatland sites. There was little evidence that birch trees were still establishing (there were few seedlings on site), but it was unclear at what rate the established stunted trees were growing. Potentially if the trees continued to grow, the impact they have on the habitat may increase, and therefore potentially eventually compromise the CO₂ sink property of the site. The site could be monitored in the long term to assess this.

It may also be important to consider the direct impacts of the trees on greenhouse gas balance. The trees are presumably a CO₂ sink as their growth would require fixation of carbon, however trees can also have a direct effect on CH₄ and N₂O fluxes (Yamulki, 2017). Further research would be required to understand the relative greenhouse gas contribution of trees, for example by taking direct greenhouse gas flux measurements from the trees.

Linking Chapter 4 and Chapter 5

The previous chapter aimed to identify whether restoration treatment and subsequent tree growth impacts on CO₂ and CH₄ flux from vegetated peat surface. However, the chapter does not quantify the gas fluxes from the trees themselves. It is intuitive that trees which are growing will be a net sink for carbon, but the relative importance of this sink compared to that of the vegetated peat surface is unknown. There is also a growing body of work highlighting the importance of trees as sources and sinks of CH₄, so it must be considered that the presence of trees in one of the plots mentioned in the previous chapters may have an effect on the relative differences between the plots. However, measuring fluxes from trees is a relatively new and developing field. The next chapter discusses the development of a chamber capable of measuring fluxes directly from the trees in the wooded plot at the Flanders Moss field site. The chamber is used to take some flux measurements from the trees at the Flanders Moss site, giving an indication of the trees' contribution to the CO₂ and CH₄ cycles.

5. Methane and Carbon Dioxide Fluxes from Completely Enclosed Silver Birch (*Betula pendula*)

5.1. Abstract

Trees have a direct importance in regulating atmospheric methane (CH₄) and carbon dioxide (CO₂) concentrations. Currently most knowledge of CH₄ and CO₂ tree fluxes are based on chamber measurements from small subcomponents of trees. In this study a large tree flux chamber is designed and tested which can entirely enclose the above-ground components of small trees (up to 3 metres high). The chamber is used to collect both light and dark measurements and quantify the variability of CH₄ and CO₂ fluxes from stunted silver birch trees growing on a degraded raised bog. The tree chamber design successfully detects CH₄ and CO₂ fluxes of silver birch trees. The silver birch trees were observed to mostly emit CH₄ but uptake was also recorded. Methane emission could vary substantially over short time periods even between being net sources and sinks. The magnitude of CH₄ fluxes, Net Ecosystem Exchange (NEE), Gross Primary Productivity (GPP) and Ecosystem Respiration (ER) were all positively related to tree size. The data indicates that tree stem CH₄ and CO₂ fluxes vary seasonally and diurnally. Evidence is presented that CH₄ emissions from trees are relatively small compared to fluxes from the surrounding vegetated peat surface, and conversely that the trees contribute to CO₂ fluxes at a similar scale of magnitude as the surrounding vegetated peat surface. The presented design for a large chamber could be used and modified to help provide another way to investigate whole tree fluxes and remediate extant limitations of tree flux studies, which are currently almost exclusively based on measurements from small subsections of trees.

5.2. Introduction

Trees play an important and complex role in regulating the earth's climate (Bonan, 2008, 2016). The capacity of trees to take up and store carbon dioxide (CO₂) from the atmosphere is well known, with the world's forests' biomass and soils being estimated to store more carbon than the earth's atmosphere (Pan et al., 2011). This capacity has made tree planting a major component of

governments' and organisations' mitigation strategies for climate change (Brancalion and Holl, 2020; Ares et al., 2021). It is also well established that forest soils can be globally important sinks of methane (CH₄) (Yavitt et al., 1990; Topp and Pattey, 1997) and that flooded/wet woodlands can be CH₄ sources (Bartlett and Harriss, 1993). More recently, however researchers are starting to understand the direct importance of trees themselves as sinks and sources of CH₄, adding complexity to efforts to model the effect of tree planting on climate change mitigation (Barba et al., 2019a; Covey and Megonigal, 2019).

Measuring gas flux from trees is a relatively recent experimental field with methodologies and mechanistic understanding still being very actively developed. There are at least four mechanisms which mediate CH₄ flux from trees: 1) trees acting as conduits for soil-derived CH₄ (Rusch and Rennenberg, 1998; Covey and Megonigal, 2019); 2) microbial production of CH₄ on or inside tree structures (Covey et al., 2012; Warner et al., 2016); 3) production of CH₄ in tree tissues (Keppler et al., 2006; Bloom et al., 2010); and 4) CH₄ uptake by trees possibly due to endophytic methanotrophs (Sundqvist et al., 2012). Upscaling the work from individual studies can demonstrate that trees have an important role in the global CH₄ cycle (e.g. Sundqvist et al., 2012; Pangala et al., 2017) however the uncertainty and variability in CH₄ fluxes makes comprehensive understanding of this contribution challenging.

A major challenge for upscaling the global impact of tree fluxes is that typically tree flux measurements are taken from a small subsample of tree surfaces. Most commonly small chambers are attached at one or more heights on the tree trunk (Pangala et al., 2013; Siegenthaler et al., 2016; Jeffrey et al., 2020) but chambers have also been attached to parts of branches (Rice et al., 2010; Machacova et al., 2016; Wang et al., 2016). Subsampling allows measurements from large trees to be easily taken, however in order to estimate the CH₄ flux from the whole tree upscaling calculations need to be made (Machacova et al., 2016; Pangala et al., 2017). Necessarily, these calculations have to be based on simple assumptions and estimations of variability across all of the tree surfaces. This is especially problematic given relatively poor scientific understanding of the mechanisms determining CH₄ flux and the comparative importance of these mechanisms. Studies taking flux measurements from multiple parts of the same tree show variable results, for example one study finds tree trunks are significantly bigger sources of CH₄ fluxes than branches (Wang et al., 2016) while other show the opposite (Machacova et al., 2016). Resolving the uncertainty of upscaling CH₄ fluxes to tree and ecosystem scale has been identified as a research priority (Barba et al., 2019a).

Enclosing entire trees is a way to account for the variability of fluxes across all tree surfaces. However, there are only limited examples where trees and other large vegetation have

been fully enclosed for the purpose of taking flux measurements, and none where the enclosure has been used to take CH₄ flux measurements. These previous designs have variously either required being set up around a tree permanently, or for extended periods of time (Barton et al., 2010; Pérez-Priego et al., 2010; Ryan, 2013); required several hours to set up around a tree (Wünsche and Palmer, 1997); or required permanent infrastructure to lower the chamber over the vegetation (Keane et al., 2019).

The aim of this study was to design, test and determine the capability of a tree chamber approach to measure aboveground CH₄ and CO₂ fluxes from entire small trees and apply this approach to quantify gas fluxes from stunted trees growing on a raised bog. The development of the chamber had the following objectives: 1) design and build a chamber, operatable with only two people, that is capable of rapidly and fully enclosing stunted trees growing in challenging terrain and 2) maximise the accuracy of chamber measurements. Once constructed and lab-tested the chamber was used to investigate CO₂ and CH₄ fluxes from stunted silver birch (*Betula pendula*) trees that have invaded a raised bog. These investigations had the following objectives: 1) quantify the relative contribution of CO₂ and CH₄ flux relative to the surrounding vegetated peat surface; 2) assess inter-tree variability of CH₄ and CO₂ fluxes; 3) assess seasonal patterns in CO₂ and CH₄ flux and what variables might drive those patterns; 4) quantify diurnal variability of CO₂ and CH₄ fluxes.

5.3. Methods

5.3.1. Chamber design

A chamber was designed to be able to enclose all above-ground components of trees up to 3 m in height, in order to measure greenhouse gas fluxes using a static chamber approach. The tree chamber system consists of four main parts (figure 5.1).

- 1) Main chamber - roughly cylindrical and split longitudinally into two halves. The chamber is made of 2.25 m tall steel uprights connected to a Perspex top and base plate (inner diameter = 110 cm, thickness = 1 cm) with the sides wrapped with a polythene membrane. The membrane used was 200 Micron Super Therm Polythene Sotrafa™ which tests have shown to have negligible permeability to CO₂ and CH₄. The chamber operates as a static chamber with gas being sampled through an inlet at the

top of the chamber, circulated to a portable greenhouse gas analyser (Ultra-portable Greenhouse Gas Analyser, LGR) and then returned to a manifold at the base of the chamber.

- 2) Tree collar - a permanent, collapsible tree collar which remains attached to the tree for the duration of the measurement campaign. It is a truncated cone polythene sheath (height = 75 cm) (figure 5.1b) The wide end of the sheath terminates in an O-ring that can magnetically seal to the underside of the tree chamber to form a gas-tight seal (see figure 5.1c). The narrow end of the sheath terminates in a section of PVC pipe sealed around the base of the tree trunk (figure 5.1a). The PVC pipe is fitted to the base of a tree by sawing the pipe open at one point, pulling it apart at this cut to get it around the tree, and then securing it shut with a Jubilee clip. Neoprene foam and Multibond Sealant (Marmox UK Ltd), which previous tests have shown to be impermeable to CO₂ and CH₄, is used to form a seal between the tree trunk base and the PVC pipe. The seam in the polythene cone is sealed with Multibound Sealant and reinforced with duct tape. The height of the sheath means that although the main chamber is only 2.25 m tall, trees of up to 3 m can be enclosed. A 1 cm PVC pipe is installed at the base of the collar to allow drainage but this can be sealed with a silicon bung during measurements.
- 3) Chamber support table – A two-part steel table that can be latched together either side of a tree to support the main chamber. The table has fully adjustable legs so that a level surface can be created on uneven terrain, and using the table minimises disturbance and compaction to the soil around the tree being enclosed.
- 4) Dark cover – when set up blocks PAR from entering the chamber, preventing photosynthesis so that the CO₂ flux measured will be ER not NEE. One part of the dark cover can be attached at six points around the top of the tree chamber, and hangs down to cover the sides of the chamber. The other part of the dark cover is lifted onto the chamber to cover its top. The dark cover consists of two inner layers of tarpaulin and two outer layers of reflective material. When the chamber was covered by this dark cover, PAR recorded in the chamber was consistently 0 μMolm⁻²s⁻¹.

To take measurements the chamber support table is assembled around the tree by putting each half of the table on opposite sides of the tree and latching them together with the tree positioned in the hole at the table's centre (figure 5.1d & e). The tree chamber is then lifted onto the stand either side of the tree and latched together (figure 5.1f & g). To completely seal the chamber the tree collar O ring is lifted and seals to the underside of the chamber. The two-part dark cover can also be placed over the chamber if desired (figure 5.1h).



Figure 5.1: a-h) Images illustrating the tree chamber systems components and how they assemble.

PAR transmission

PAR intensity can be a major factor in determining primary productivity, so the extent to which a chamber transmits PAR can influence CO₂ light measurements. The chamber's frame and polythene/Perspex surfaces may result in PAR entering the chamber. To test for this PAR was measured simultaneously with two cross calibrated PAR sensors (Quantum Sensor, Delta-T Devices). PAR was measured at 2 heights in the chamber (1 m and 2 m) and on the north and south facing sides of the chamber (a total of 4 positions). In each position the sensors were left logging every minute for two days. Average PAR transmission into the chamber was 85%. Shading

from the metal frame of the chamber caused greater than average reduction in PAR but the small area affected made this a minor feature, so this data was simply included in calculating the average. There was also evidence of slightly higher transmission of PAR through the Perspex top of the chamber in comparison to the polythene sides, however this difference was small so again all the data transmitted through both materials was included in calculating an overall average.

The average PAR transmission of 85% for the chamber is relatively high so should have relatively minor impacts on GPP. If the chamber was used to collect a larger data set it may be possible to calculate an accurate correction for reduction to GPP, as with Heinemeyer et al. (2013).

Leakage from the chamber

Initial leak checks were performed with smoke pellets which when lit inside the chamber could be used to visually identify major leaks, which could then be remediated. Leaks could usually be resolved by applying sealant, thus improving the gas-tight quality of the chamber. Subsequent tests where CO₂ and CH₄ were raised in the chamber and then the chamber sealed showed that these gases did not leak out of the chamber at significant rates.

5.3.2. Accuracy of the chamber

Effect of number of fans

The main chamber was initially fitted with three 10 cm diameter computer fans. Tests were carried out to determine the number of fans required to achieve similar CO₂ fluxes from sediment as would be given in a standard small chamber. The test took place with the tree chamber set up with the tree collar sealed around a glass jar in place of a tree, to remove the need to enclose a tree in order to seal the chamber for testing. To generate a CO₂ flux ~100 grams of sugar was mixed with ~2 litres of potting compost and placed in an open topped container within the chamber. Fluxes from the compost were measured using the tree chamber for 30 minutes and compared with measurements taken using a normal small chamber sealed for 2.5 minutes. The gas flux observed in the small chamber was assumed to be the 'true' flux and the fluxes calculated from the tree chamber were divided by the 'true' flux to give a ratio whereby 1 would represent complete agreement between the two chambers, <1 would indicate the tree chamber

underestimated the flux, and >1 would indicate the tree chamber overestimated the flux by comparison to the small chamber.

Fan number significantly affected the ratio of observed CO₂ flux in the tree chamber against expected flux in the small chamber, as indicated by an ANOVA - $p = 0.035$. The data meet the assumptions of homoscedasticity and normality (insignificant Shapiro–Wilk and Levene’s test). Tukey’s pairwise comparison showed that three fans were significantly different from zero fans ($p=0.043$), and two fans was almost significant from zero fans ($p=0.061$), while one fan was not significantly different from zero fans ($p=0.350$). The mean ratio of expected flux to measured flux for both two and three fans was 0.89 and 0.90, compared to 0.69 for zero fans.

The gas uptake port in the tree chamber is suspended in the centre of the chamber near the top. Therefore, if even mixing was not occurring in the chamber, then an under-estimation of the flux would be expected as the pot of soil which was positioned at the bottom of the chamber far from the uptake port. The fan test results are consistent with 2 and 3 fans improving mixing in the chamber compared to no fans. Using fans in chambers can disrupt flux measurements, for example through disrupting boundary layers (Davidson et al., 2002; Pumpanen et al., 2004) so the chamber was operated with two fans in field conditions as three fans would needlessly increase the risk of these effects.

Effect of chamber temperature

Air temperature inside a flux chamber can change relative to outside temperatures. During flux measurements temperature sensors placed at 2 m height and connected to a GP1 data logger (Delta-T instruments) were used to assess the strength of the heating effect inside the chamber across the length of the measurements. The greatest temperature changes in the tree chamber recorded under field conditions were recorded during a campaign of measurement in May when PAR was highest. The highest increase in temperature in May was 2.9°C for the duration of the light CO₂ measurements, becoming a 5.2°C increase over the longer CH₄ measurement window. The average increase in temperature during the May light measurements was 0.8°C during the CO₂ measurements and 2.3°C during the longer CH₄ measurements. The temperature increases recorded during flux measurements are in a similar range to those of other studies, e.g. Debouk et al. (2018). These temperature changes may impact the physiology of the trees and thus the fluxes they produce. However, even when the flux measurements were taken under increasing temperatures there was no evident effect on linearity or trajectory of the flux, indicating that temperature changes were not significantly impacting flux measurements.

5.3.3. Carbon dioxide and methane flux variability in stunted silver birch growing on a degraded lowland raised bog

Field site

Once the chamber development was complete, an experimental study took place in Flanders Moss National Nature Reserve, a lowland raised bog in Central Scotland. Stunted silver birch (*Betula pendula*) have seeded onto an area of peat approximately eight metres deep that had been drained, ploughed and planted with lodgepole pine (*Pinus contorta*) in the 1970s. In July 1998, as part of a restoration intervention, the commercial forestry was removed, and main drains were blocked on the site. The ridge-furrow plough microtopography remains, with most of the trees establishing on the ridges. It is believed that most of the silver birch trees would have seeded within a few years of the forestry being removed, meaning the trees measured would be ~20 years old (Anderson, 2010). The trees generally appear healthy but stunted in growth considering their age, with a mean height of 2.57m for all trees more than 1.4 m tall (i.e. approximately 74% of all the trees and seedlings, based on counts in sample plot). The tallest tree in the study area measured 4.59 m; by comparison silver birch grown on a commercial plantation in the same region had an average height of 6.05 m after just eight years (Lee et al., 2015). Carbon dioxide and CH₄ measurements were taken from five trees at the site, chosen to cover a range of sizes (e.g. different heights and Diameter at Breast Heights) within the capacity of the tree chamber. DBH² x height was also calculated as this is a metric often assumed to have a linear relationship with tree volume (Picard et al., 2012), and basal diameter² x height was also tested. These metrics were plotted against CO₂ and CH₄ fluxes and linear regression was used to identify any relationship between tree size and magnitude of fluxes.

Variation and magnitude of tree fluxes

Seasonal variability in CH₄ and CO₂ fluxes were assessed from the trees growing on the site, by taking light and dark measurement from five trees in mid-summer (August 2020), autumn (October 2020) and late spring (May 2021). Each measurement was taken for 20 minutes. Height, Diameter at Breast Height (DBH i.e., diameter at 1.3 m above ground) and the diameter at the base of the tree (basal diameter) were measured for all five trees.

In May 2021 diurnal variability in CH₄ and CO₂ fluxes was determined by taking three sets of consecutive light and dark measurements from four trees during daylight hours. To allow recovery of the trees after measurements, 20 minutes were left between measurements, during

which the chamber was removed from the tree. During these measurements Photosynthetically Active Radiation (PAR) was recorded. In addition, diurnal variation over a slightly longer timescale was tested for: a light and dark CO₂/CH₄ flux measurement was taken from one tree at three-time points; at solar midday (~13:00), early night (~20:00), and solar midnight (~01:00) across one day in October 2020. For both the May and October data a relationship was tested for between the fluxes and: time of year; Photosynthetically Active Radiation (PAR); soil and air temperature; and water table depth.

5.3.4. Flux calculations

All gas fluxes were calculated in R (R Core Team, 2021) using the package FluxCalR (Zhao, 2019) which, by default, uses a linear fitting method to calculate gas flux for the section of the measurement window with the highest R² value (the most linear section of the measurement). Chamber measurements on the field site were typically taken for 20 minutes. To determine the optimal position and length of the window for flux calculations all the measurements for CO₂ and CH₄ were plotted and examined. Different window lengths were trialled and their R² value compared.

Using linear regression to calculate fluxes was considered reasonable as CO₂ fluxes from the tree measurements generally showed good linearity for the duration of chamber closure. For some measurements, slight curvature was observed in the changing gas concentration, especially in the case of light measurements with large falls in CO₂ concentration (i.e. ~100 PPM CO₂) in the chamber. To ensure the CO₂ fluxes were not affected by duration of closure, fluxes were calculated from the start of closure as steady fluxes were achieved immediately after complete closure of the chamber. For the July 2020 and May 2021 measurements the fluxes were relatively high, meaning that three minutes were sufficient to calculate fluxes that appeared to reflect gas concentration change. In October 2020, when fluxes were lower, the measurement window was extended to five minutes.

Methane fluxes generally showed reasonable linearity for the duration of chamber closure in July 2020 and October 2020. For measurements taken in May 2021 linearity was much more variable, with some measurements even varying between being distinct CH₄ sources and sinks depending on when in the measurement period the flux was calculated. Due to the small CH₄ concentration changes it was beneficial to have longer flux windows to optimise the R² value, and it was determined that a flux window of 12 minutes was generally suitable.

Minimum Detectable Flux (MDF) for the chamber was calculated as per Nickerson (2016) to give an indication of what magnitude of flux would be detectable and what length of measurement would be required to detect various flux magnitudes. The Minimum Detectable Flux for CH₄ in this chamber over a 12 minute window was calculated as 0.00000538 μmol s⁻¹ per tree. All of the CH₄ flux measurements from silver birch trees had at least an order of magnitude greater fluxes than the Minimum Detectable Flux, except for one, which was approximately double.

5.4. Results

5.4.1. Seasonal carbon dioxide and methane flux variability

Carbon dioxide fluxes (NEE, PP and R) were lowest in October and higher in August 2020 and May 2021 (figure 5.2 a-c). All the trees were negative NEE in August 2020 and May 2021 meaning that the rate of R exceeded the rate of PP, whereas in October all but one tree had a positive NEE. At the time of measurements in October all of the trees had lost at least two thirds of their leaves and the remaining leaves were yellow or yellowing, whereas during August and May measurements the trees' leaves were green with no evidence of significant senescence.

Trees were predominantly emitters of CH₄ at all three seasonal time points (figure 5.2d). CH₄ fluxes showed less consistent seasonal trends, as well as variability between trees. The general pattern was similar to the seasonal trends in NEE, PP and R, with CH₄ fluxes highest in either August or May and lowest in October (with the exception of one tree). Three of the trees were recorded as having their lowest flux in October and the other two trees had their second-lowest flux in October.

5.4.2. Short-term variability

The diurnal measurements taken in October showed an increase in both CO₂ respiration and CH₄ flux between solar midday and solar midnight (figure 5.3).

A paired Wilcoxon-signed rank test on CH₄ measurements showed that the dark measurement (which was always taken first) had a significantly lower median ($p=0.010$) than the subsequent light measurement across the seasonal data.

I further tested for factors that might affect short term CH₄ flux variability during the repeat light and dark measurements taken from each tree in May. The effect on CH₄ fluxes of PAR intensity, chamber temperature and starting CH₄ concentration in the chamber during the May repeated light measurements was tested for in separate ANCOVAs, with the tree from which the measurements were taken being used as a covariate. PAR and temperature show a tendency to be negative predictors of CH₄ flux ($p=0.072$ and $p=0.067$), whereas starting concentration of CH₄ in the chamber is not. The same analysis was applied to the repeated dark measurements and neither PAR, temperature, or CH₄ starting concentration in the chamber were significant predictors of CH₄ flux.

5.4.3. Variability of carbon dioxide and methane flux with tree size

Of the five different metrics of tree size tested, basal diameter² x height typically showed the strongest linear relationship with magnitude of ER, GPP, NEE and CH₄ fluxes from the trees separated by season. ER in any given season was strongly positively correlated with basal diameter² x height, with the linear relationship from each season having an R² value of greater than 0.9. NEE and GPP were strongly negatively linearly correlated with basal diameter² x height, with an R² value >0.9 for data from August and May. However, there was no clear relationship between tree size and NEE/PP for the October measurements. CH₄ measurements showed weak significant linear relationships but for August and October measurements of CH₄ were positively linearly correlated to basal diameter² x height (R²=0.7 for both). There was no strong relationship between metrics of tree size and CH₄ for the May measurements.

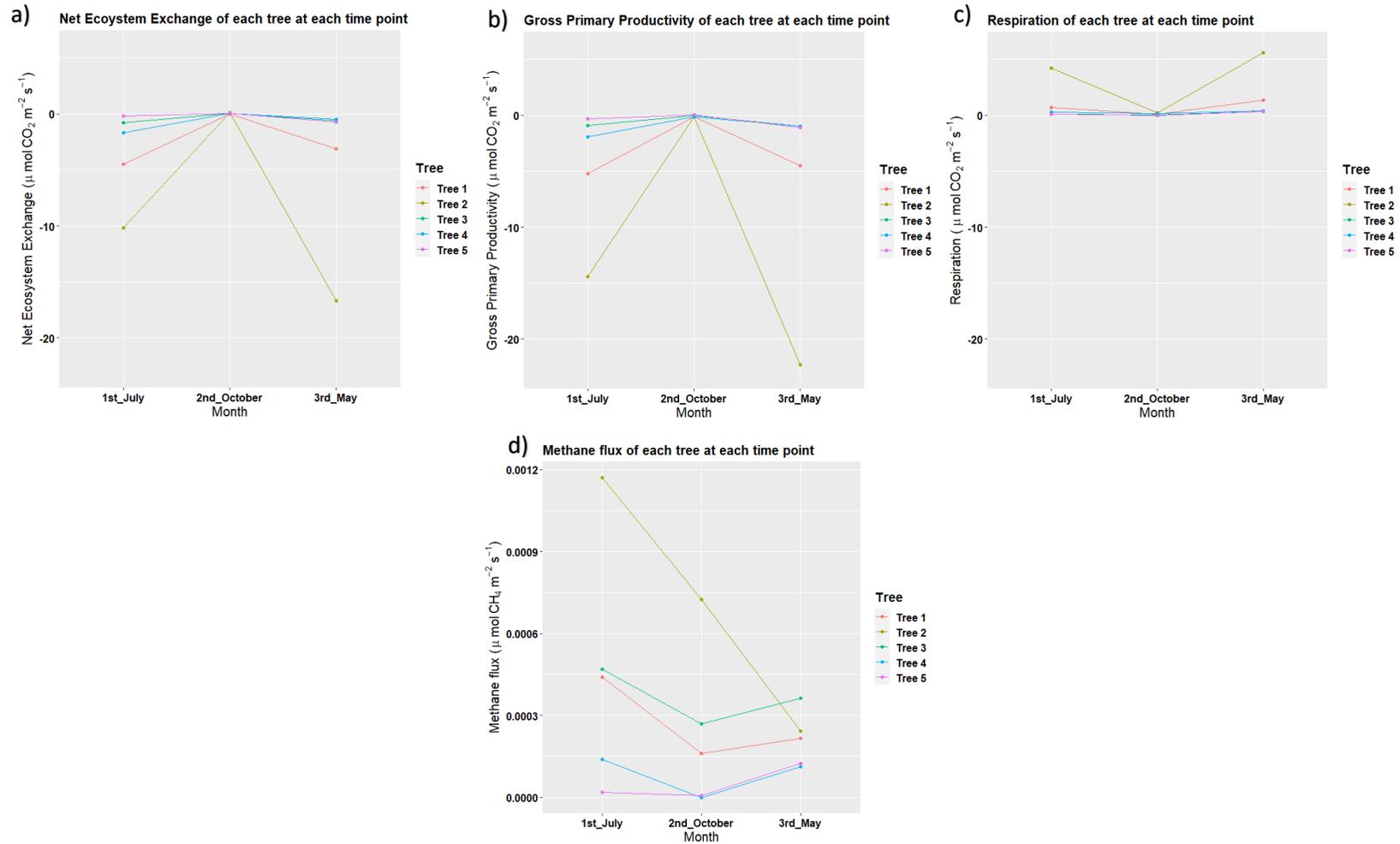


Figure 5.2: Seasonal variation in tree CO₂ and CH₄ flux: a) Ecosystem Respiration (ER), b) Gross Primary Productivity (GPP), c) Net Ecosystem Exchange (NEE) and d) Methane flux (CH₄).

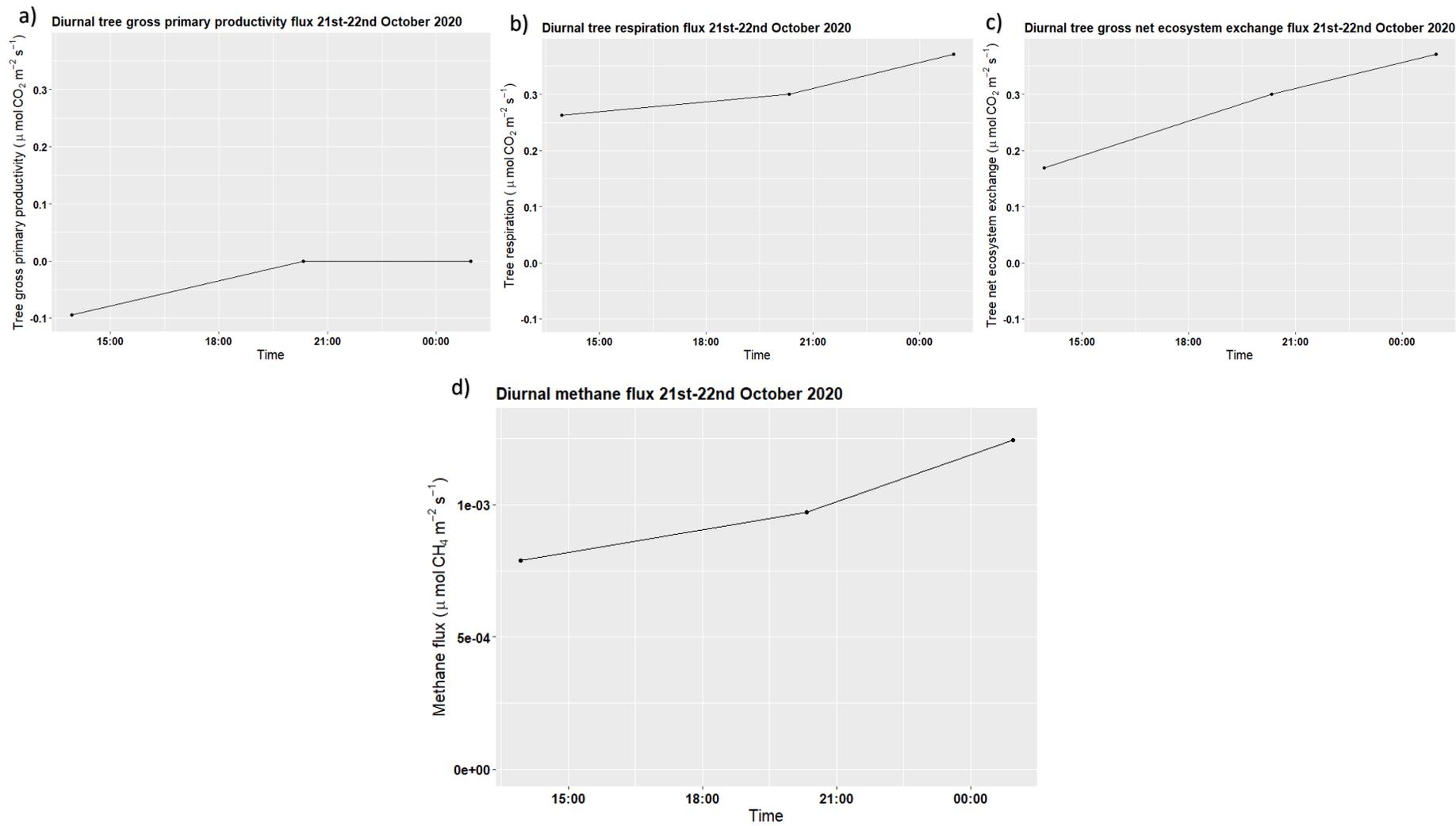


Figure 5.3: Diurnal tree fluxes in October 2020 for: a) gross primary productivity, b) respiration, c) net ecosystem exchange and d) methane.

5.5. Discussion

5.5.1. Contribution of birch trees to total ecosystem fluxes

The tree flux data in this chapter is fairly limited, but for a rough quantification of the significance of the CH₄ fluxes from the trees it could be assumed that the mean CH₄ flux from the five trees studied was representative for all the trees in the wooded plot. This plot estimate can then be used for comparison with the same plot's estimate for vegetated peat surface fluxes for the same day by scaling up the mean ground collar flux to the area of the whole plot. This method estimates that the trees contributed 1.29%, 1.38% and 1.68% of wooded plot CH₄ fluxes in August 2020, October 2020 and May 2021 respectively. This is a very rough estimate but indicates that the trees are not substantially contributing to CH₄ flux in the wooded plot. As such, the general findings of differences between the open plot and wooded plot in Chapter 4 should not be affected. The low contribution of the trees to CH₄ fluxes contrasts with Pangala et al. (2015), which found that downy birch trees on an English minerotrophic peatland site contributed 27% of the ecosystem CH₄ fluxes. Pangala et al. (2015) did however predominately measure much larger trees, up to 35 cm DBH, whereas the largest tree measured in Chapter 5 of this thesis had less than 4 cm DBH.

The direct contribution of trees to CH₄ emissions has been lacking from most research and discussion about afforested peatlands in Scotland. This study provides evidence that relatively stunted trees on peatlands are not significant sources of CH₄ relative to the vegetated peat surface. This is the first study in Scotland to measure CH₄ fluxes from tree stems growing on ombrotrophic peat. Birch regeneration on degraded peatland or peatland undergoing restoration is a common occurrence and its impacts are not fully understood. It may become increasingly common for land managers to allow low density native woodland to establish on peatlands, for example the Forestry Scotland's policy of Peatland Edge Woodland which is proposed as a low density predominately native woodland which may be accepted as an alternative to commercial forestry or forest-to-bog restoration at some sites. Further research would be helpful to form a more complete idea of the impact of these policies, in particular data from other stunted trees of different native (e.g. *Pinus sylvestris*, *Betula pubescens*, *Salix* spp.) and non-native (*Pinus contorta* and *Picea sitchensis*) tree species which regenerate on peatlands. In addition, flux measurement from larger trees would be helpful to better understand the potential greenhouse gas balance of woodland on peatland over long time scales.

The contribution of CO₂ fluxes from the trees relative to the vegetated peat surface was also compared using a similar method as used for CH₄ but instead of assuming a constant flux for the whole day the NEE of the trees and the vegetated peat surface was assumed to be constant for day light hours and the respiration was assumed to be constant during night time hours to give a whole day estimate. This estimate shows that the tree fluxes on the plot are of a similar magnitude to the vegetated peat in the same area. When measurements were taken in May the vegetated peat surface was a net sink and the trees were also a net sink of almost exactly the magnitude. For the measurement for a day in July and a day in October both the trees and the vegetated peat surface were net sources of CO₂. The estimated total tree fluxes taken were respectively 42.6% (July) and 79.7% (October) of the estimated size of the whole plot flux. The data show that on the days for which there are measurements, the trees have increased the overall magnitude of the sink or source effect of the plot. When the trees are accounted for, in May the plot became a bigger sink for CO₂ than the open plot but for the July and October measurements the extent to which the wooded plot was a source of CO₂ was exaggerated.

The relative scale of the CO₂ tree fluxes shows that they are a significant factor in the greenhouse gas balance of the plot, however there is not enough data to understand if considering the direct contribution of the trees to CO₂ flux would significantly affect the overall annual greenhouse gas budget of the wooded plot or to what extent days on which the trees are CO₂ sinks counteract days on which the trees are CO₂ sources. To more fully understand the contribution of the trees it would also be useful to improve understanding of fluxes from tree leaf litter and root systems, both of which are expected to be sources of CO₂.

5.5.2. Carbon dioxide and methane flux variability from silver birch trees at Flanders Moss

The size of the data set collected from the silver birch trees at Flanders Moss using the tree chamber is very small so further investigation would be required to explore findings further. However, the data set does indicate some important findings.

The CH₄ fluxes from the enclosed trees were predominantly positive but some were negative. This would suggest that there were at least two mechanisms affecting CH₄ flux; at least one producing CH₄ and at least one taking up CH₄. From the limited data it is unclear which mechanisms may apply to the measured trees, but it can be posited that they act as conduits for CH₄ produced below-ground in the soil, as this is commonly observed for trees growing in

wetlands (Pangala et al., 2013, 2017) including in the specific case of the related downy birch (Pangala et al., 2015).

Methane uptake in some tree species, including downy birch, has been observed in tree canopies (Sundqvist et al., 2012), and is considered potentially attributable to methanotrophic endophytic bacteria (Van Aken et al., 2004). Different parts of the tree (e.g. the trunk and the canopy) can be dominated by different and potentially opposing routes of CH₄ flux, but by measuring entire trees rather than small subcomponents thereof it is possible to measure the combined effect of these pathways.

The study identified a linear relationship between tree volume and ER for all time points and for CH₄ in August and October. If this relationship was shown to be consistent then it would be a useful relationship for upscaling from a subsample of trees at a site to all trees on the site by simply measuring their diameter and height. Mechanistically the relationship would appear logical for ER as a greater volume of tree would indicate a larger volume of respiratory active tree cells/greater volume of dead, decaying material. Tree volume may also be linearly related to sources of CH₄ production such as microbial production inside tree wood (Covey et al., 2012; Warner et al., 2016).

In some of the CH₄ measurements in May 2021 there was considerable variation in the rate of flux during the measurement itself, an observation which was not seen in August 2020 and October 2020 measurements. It is unclear from the data what might be driving this and why this was not observed at the other time points. However previous studies have identified that CH₄ fluxes can be temporally highly variable over short time scales (Barba et al., 2019b).

The data indicates that there is seasonal variation in CO₂ and CH₄ fluxes from the trees. The variation was more distinct for ER, NEE and GPP than it was for CH₄, but in most trees the magnitude of all fluxes was lowest in October. Similar to this study, stem flux studies have documented less pronounced seasonal change in CH₄ than CO₂ fluxes (Warner et al., 2016).

The CO₂ fluxes from the trees show roughly the same seasonal pattern as from the wooded plot ground collars in Chapter 4, with both features being a net source for CO₂ in the August and October measurements and a net sink for the May measurements. The seasonality of the CH₄ fluxes from the trees was proportionately similar to variation in the ground collars described in Chapter 4. However, the small number of sampling dates with an almost 8-month gap over the winter and early spring means some seasonal patterns could be missed. For example Pangala et al. (2015) observed that the proportion of CH₄ fluxes from downy birch trees relative to the vegetated peat surface was greater in winter and spring.

5.5.3. Applications of the tree chamber

This study developed and tested a new tool for studying CH₄ and CO₂ fluxes from trees. The design can therefore enhance understanding in this field in the following way:

1. It allows simultaneous measurement of CO₂ and CH₄ fluxes from all above-ground surfaces of stunted/juvenile trees and scrub. To date, a vast majority of research on gas fluxes from trees is based on measurements from small subcomponents. Not only do these techniques rely on simplistic upscaling approaches to estimate whole tree fluxes but they often neglect to account for multiple types of subcomponent (e.g. branches versus stems). This chamber allows a way for all above-ground surfaces to be easily and simultaneously measured.
2. A limitation of the design is that there is a restriction on the upper size of tree that can be enclosed. However, the basic chamber design could easily be modified to include a hole at the top of the chamber equivalent to that at its base, so that an inverted collar could be attached above the chamber as well as below. This modification would allow for flux measurements to be taken over a much larger section of tree trunk than would be possible with conventional techniques. Despite the fact that CH₄ fluxes can be very spatially variable, fluxes are usually measured over small surface areas (Barba et al., 2019a). A modified version of the chamber would allow measurements over larger areas and thus may give more representative measurements.
3. The chamber could be used in combination with sub-component surface measuring techniques to test the accuracy of upscaling methodologies compared to whole tree chamber measurements.

6. General Discussion

6.1. Thesis context

Large areas of northern peatlands were degraded by afforestation in the twentieth century (Sloan et al., 2018b). In Scotland, concerns about the ecological damage caused by peatland forestry led to a shift in regulation to prohibit the expansion of forestry onto peatlands (Stroud et al., 2015). However there remains scientific uncertainty and ideological conflict over what should be done to areas already degraded by forestry (Payne and Jessop, 2018). Peatlands have been increasingly valued for the large amount of carbon they store: depending on management, an area of peatland can continue to store this carbon and even be a weak carbon sink, but under poor management, such as artificial drainage, they have the potential to instead become large carbon sources. There is increasing political pressure to reduce carbon losses from peatlands. The international COP26 Glasgow Climate Pact agrees to limit global warming to 1.5°C and sees countries set Nationally Determined Contributions for reduction of greenhouse gas emissions. The UK aims to reduce national emissions by two thirds on a 1990 baseline by 2030 and ultimately have net zero emissions by 2050. These goals include the contributions of Land Use, Land-Use Change and Forestry (LULUCF). The most recent detailed management guidance on afforested peatlands accordingly has a focus on the greenhouse gas balance of different management options.

Nature based solutions – where the management of natural or modified ecosystems is improved to both enhance biodiversity and address societal challenges – can be a sustainable approach to address emissions relating to LULUCF. The new afforested peatland guidance supports forest-to-bog restoration in some situations; this is a type of nature-based solution which restores open bog habitats while also aiming to protect the peat carbon store and other peatland ecosystem services. The guidelines also recognise a need to maintain timber production, and so propose that productive forests are restocked when growth is believed to be sufficient to compensate for carbon loss from the peat by carbon sequestration in the trees. Finally, the guidelines propose a novel management option, Peatland Edge Woodland (PEW), envisaged as a low density predominantly native woodland established using low intensity methods (Forestry Commission Scotland, 2015). Peatland Edge Woodland can be seen as a nature-based solution as it is theorised to create valuable habitat while simultaneously providing human benefits such as being net carbon neutral. However, despite PEW being predicted to result in net zero greenhouse

gas emissions, there is limited scientific understanding as to what the impacts of such a habitat would be or whether it would be possible to establish this type of habitat.

6.2. Main findings and their implications for Peatland Edge Woodland policy and practice

The question posed by this thesis was 'Is Peatland Edge Woodland an Appropriate Management Option for Afforested Peatlands After Harvesting?'. The thesis explores this both in terms of whether stakeholders with a professional interest in afforested peatlands think PEW is an appropriate option, and in terms of whether PEW is an appropriate option for providing climate change mitigation benefits. This thesis uses social and natural science methodologies to enhance understanding of the context and environmental impacts of PEW. Chapter 1 introduced the historical, ecological and political background that gave rise to the conception of PEW as a policy idea. Chapter 2 then explored how this concept has been interpreted by stakeholders with a professional interest in afforested peatlands. Finally, Chapters 3-5 explored some the impacts of native woodland on above-ground carbon storage and ecosystem fluxes of methane (CH₄) and carbon dioxide (CO₂).

Chapter 2 highlighted awareness of PEW in a range of stakeholder groups with different professional interests in afforested peatlands. There were a range of interpretations of how PEW should be established/structured, and these were linked to the ideological viewpoints of the stakeholders, especially their conceptions of naturalness. Some participants were opposed to PEW and sceptical of the motivations of those who establish it. However, there were also a wide range of stakeholders who supported the idea of PEW, albeit with varying perceptions of what PEW should be like. Two main contexts in which PEW would be established were identified, the 'Ecotone model'; in which PEW would be established in areas determined by natural variation in the site (e.g. in areas with shallower peat, near flushes etc) and the 'Pragmatic model'; in which PEW would be established in areas where alternative management was considered too challenging due to anthropogenic constraints or damage to the site. Neither of the field sites used in this thesis (Chapter 3-5) were intentionally created as PEW since PEW is a recent concept, and it is therefore not possible to strictly apply either model to these sites. Nonetheless, both of the field sites containing PEW proxy habitat which were used in this study more closely match the pragmatic model. The Rumster Forest plots (Chapter 3) were established at a wide range of peat depths and not over any obvious ecotonal boundary. The Flanders Moss plots (Chapter 4-5) were

established in the main body of a raised bog where management decisions had unintentionally facilitated establishment of birch trees. The birch trees were not growing on any obvious ecotone and had not been removed by the site managers for various pragmatic reasons.

Chapter 2 identified a spectrum of opinion on how PEW should be established. There were low intensity perceptions of PEW, focused on areas where tree growth would be relatively slow and putting emphasis on drainage remediation to prevent peat carbon losses. In contrast, there were also higher intensity perceptions of PEW, where some carbon losses from the peat were considered acceptable and more emphasis was placed on promoting tree growth to compensate for peat carbon losses. The field sites identified in this study give insight into the diversity of this spectrum: the native woodland plots at Rumster Forest have key characteristics of high intensity PEW and the wooded plot at Flanders Moss has traits of an intermediate intensity PEW – combining elements of both high and low intensity. Table 6.1. compares these field sites with the theoretical types of PEW identified in Chapter 2.

Chapter 2 highlighted that many stakeholders who were interested in establishing PEW thought there was a lack of scientific understanding or case studies of the impacts of establishing PEW. Chapters 3-5 aimed to contribute data to inform discussions over the effects of PEW on greenhouse gas balances. Both greenhouse gas flux and carbon storage methodologies are employed in this thesis to quantify different aspects of the carbon storage and greenhouse gas balance of native woodlands.

In Chapter 3 the work at Rumster Forest explored the carbon-storage potential of different native tree planting treatments relative to non-native (commercial) restocking treatments. The non-native options stored substantially more carbon but most of the native woodland treatments did establish and contribute to plot carbon storage. There was some evidence of other benefits as well, such as suppression of non-native regeneration in some of the broadleaf native woodland plots and higher water tables in the native woodland plots relative to the non-native. However, no greenhouse gas flux measurements were taken and as such the study could not assess emissions or uptake of these gases from the vegetated peat surface or trees themselves and how this related to carbon storage in the trees.

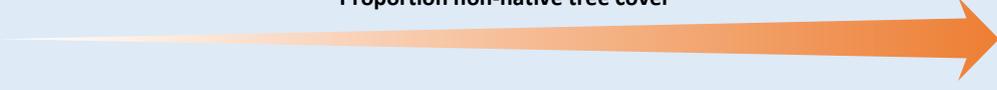
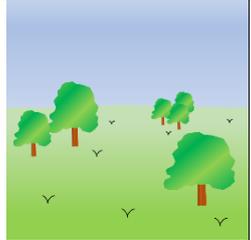
Chapter 4 showed that although a 22-year-old forest-to-bog restoration site was a net CO₂ sink, the large CH₄ fluxes meant the site remained a source of greenhouse gases. At this site the wooded plot had significantly reduced CH₄ fluxes, indicating that the trees are currently not adversely affecting CO₂ fluxes and may in fact be actively reducing CH₄ fluxes. Chapter 5 aimed to quantify the fluxes from the above-ground parts of the trees themselves. These measurements

required challenging technical developments and this chapter presents a new approach to tree flux measurements. The data collected showed that the measured silver birch trees were predominantly net sources of CH₄, and either net sinks or sources of CO₂ depending on the conditions and time of year. Overall, however, the trees were a minor component of CH₄ gas fluxes compared to fluxes from the vegetated peat surface. The trees in the wooded plot were estimated to have fluxes in the same order of magnitude as the vegetated peat surface with an indication that the trees exaggerate the extent to which the plot would be a sink or a source of CO₂ on any one day.

Chapter 2 identified different stakeholder attitudes on the intensity of PEW (table 2.1). Table 6.1 is a modified version of table 2.1 which highlights the ways in which the theoretical conceptions of high and low intensity PEW compare to the field sites studied in this thesis. The native woodland plots in Rumster Forest (the field site used in Chapter 3) are identified as a proxy for high-intensity PEW whereby carbon sequestration in the trees should be substantial in order to compensate for peat losses. Chapter 3 highlighted that although the PEW-like native woodlands studied can sequester carbon this is at a much-reduced rate than non-native commercial forestry. Chapter 3 gives only limited, indirect data on the relative effect of the tree treatments on emissions from the peat. However, given the fairly moderate difference in water tables between native and non-native plots discussed in Chapter 3, it would be unlikely that CO₂ emissions from the peat would be substantially less in the native plots (although more research on greenhouse gas fluxes from the peat would be required to be confident in this conclusion). It is therefore unlikely that the PEW-like habitats at this site are as effective for climate mitigation as the non-native plots; from a climate change mitigation perspective this is evidence against it being appropriate to establish higher intensity PEW.

In contrast to the native woodland plots at the Rumster Forest field site, the wooded plot at Flanders Moss (the field site used in Chapters 4-5) is identified as being analogous to a lower or intermediate intensity PEW site, including featuring substantial drainage remediation and with trees being established through regeneration rather than planting. Consequently, the trees at Flanders Moss were typically smaller and less vigorous than the native trees at Rumster Forest. The wooded plot at Flanders Moss remained a CO₂ sink and experienced reduction in CH₄ emissions relative to the open plot. Birch scrub is often viewed as problematic in the context of peatland restoration, but this thesis finds evidence that it does not have an adverse effect on greenhouse gas balance (over the timescales studied).

Table 6.1: Compares the theoretical types of PEW identified in Chapter 2 with observations and data from the field sites used in Chapter 3-5.

Character Spectrum	Increasing: Tree cover Proportion non-native tree cover			
				
Category	Low Intensity PEW Chapter 2 (theoretical)	Intermediate Intensity Chapters 4 & 5 (field site)	High intensity PEW: Chapter 2 (theoretical)	High Intensity PEW Chapter 3 (field site)
Visual rendering	 Representation	 Flanders Moss, wooded plot	 Representation	 Rumster Forest, native plots
Intention of management approach	Restore to a state considered to have once been more prevalent	To test different forest-to-bog restoration approaches	Restore to a system without an obvious analogue	To test the productivity of different native planting mixes
Level of drainage	Drainage fully or partially remediated	Main drains blocked	Drainage partially or un-remediated	No drainage remediation
Native tree cover	Low density of native trees	Very high (0.51 trees/m ²), but stunted	Relatively high density of native and non-native trees	Variable but high (up to 0.25 trees/m ²), with variable growth
Establishing trees	Self-seeded regeneration or low intensity planting	Self-seeded regeneration	Planted and self-seeded regeneration	Planted and self-seeded regeneration
Non-native tree cover	No non-native trees	No non-native trees	Non-native trees might be a component of tree cover	Present, variable coverage
Ground flora vegetation	Open peatland vegetation well restored	A mix of open peatland and woodland vegetation	A mix of open peatland and woodland vegetation	Woodland vegetation more dominant than open peatland
Carbon balance: theoretical strategy compared with observed effects	<u>Theoretical strategy:</u> <ul style="list-style-type: none"> Prevent carbon losses from peat carbon store 	<u>Chapter 4:</u> <ul style="list-style-type: none"> Vegetated peat surface has CO₂ sink function restored even with tree growth Presence of trees reduces CH₄ emission relative to treeless areas <u>Chapter 5:</u> <ul style="list-style-type: none"> Tree contributions to CH₄ emissions are negligible 	<u>Theoretical strategy:</u> <ul style="list-style-type: none"> Some carbon loss from peat carbon store acceptable if compensated for by sequestration through growth of trees 	<u>Chapter 3:</u> <ul style="list-style-type: none"> Native trees sequestered approximately 15% of the carbon sequestered by commercial non-native plantings. Indirect evidence that carbon loss from peat may be less than that lost under non-native forestry but this is not quantified

This thesis only compares two field sites and so is not exhaustive. However collectively the data indicate that high intensity PEW may not be an effective climate change mitigation strategy: using native trees to compensate for carbon losses may be relatively ineffective compared to planting non-native trees, while failing to provide substantial reductions in below-ground carbon losses. Conversely, allowing the development of lower intensity PEW habitats can restore the net sink function of a peatland area (over the timescales studied) and therefore may be an effective climate change mitigation strategy.

Scots pine is a species that can naturally grow on peatlands in Scotland and internationally, so could potentially be suitable for PEW. However, Chapter 2 highlighted that many stakeholders had concerns about Scots pine as a PEW tree due to misgivings about whether it could establish in wet conditions, its susceptibility to disease and whether it would support much biodiversity benefit. Chapter 3 showed that Scots pine grew less successfully than native broadleaf trees in terms of survival rate, carbon storage and non-native regeneration prevention. Chapters 4-5 focused on a PEW proxy site dominated by *Betula pendula* with some *Betula pubescens* so no comparison to Scots pine could be made. The results from this thesis indicate both that native broadleaved species such as *Betula* and *Salix* species may be more likely to be used in PEW creation and also that these species may be more likely to yield climate benefits than Scots pine.

The thesis provides evidence that lower-intensity broadleaf PEW types may have the best potential for establishing and preventing non-native regeneration while being carbon neutral or resulting in net carbon sequestration. That native woodland on peatland might have the capacity to have neutral or positive carbon effects runs contrary to studies on similar habitats (Friggens et al., 2020; Mazzola et al., 2022), and as such caution should be taken around concluding that every low intensity PEW would necessarily have climate change mitigation benefits. Instead, this thesis highlights that in some situations PEW may provide climate change mitigation benefits.

In the afforested peatland guidelines PEW is proposed to have multiple benefits, and participants in Chapter 2 were often motivated to establish PEW primarily for biodiversity reasons. From this perspective PEW doesn't necessarily need to provide substantial climate change mitigation benefits. Instead, PEW habitats not having detrimental effects on climate change mitigations would be a subsidiary factor to support the establishment of PEW for other primary motivations.

The thesis shows that there are contexts where PEW may be an appropriate management option. From a land management perspective, although some stakeholders with a professional

interest in afforested peatlands were entirely opposed to PEW, a majority thought there were at least some contexts in which it could be an appropriate option. Chapter 3 provides evidence to caution against relying on trees growing in PEW to sequester carbon, as this may not be appropriate if the intention is to maximise the climate change mitigation potential of an area. However, Chapters 4-5 illustrate a context in which lower intensity PEWs can restore vegetated peat that is a net sink for CO₂ while reducing CH₄ emissions; highlighting, at least over the timescales studied, that PEW can be an appropriate option for providing climate benefits.

6.3. Future research

Chapter 2 highlighted that many stakeholders identify the lack of scientific knowledge and case studies as a major issue in making decisions about PEW. This thesis addresses some of this uncertainty but large uncertainties still remain.

Long term effects of PEW

A major concern highlighted in Chapter 2 was the long-term stability of PEW. Peatland Edge Woodland stability relies on two main components: how vigorously trees will grow and how rate of tree establishment will compare with rate of tree death. This is especially relevant because forest-to-bog restoration is often depicted as providing the best benefits over long timescales (Lindsay, 2010; Hermans et al., 2019). Both field sites studied in this thesis were over twenty years old, however their long-term trajectories are unclear. At both sites there was little evidence of native tree regeneration within the sites, suggesting that the native tree density is currently stable, however the growth and death rate of the established trees is unknown, as is the potential impact these growth and death rates could have on ecosystem services. Long-term monitoring of the field sites would be useful for determining this. If more PEW and PEW-like sites could be found that are of different ages but similar histories then chronosequence comparisons would be possible, such as have been done for forest-to-bog restoration (Creedy et al., 2017, 2020; Gaffney et al., 2018; Hermans, 2018). However, it is unlikely there will be many PEW proxy sites (sites where native trees have established after at least one commercial rotation of forestry) with trees much older than those used in this study due to the relative recency of afforestation on peatlands in Scotland.

There are many examples where mature native woodlands have established on areas of deep peat over long timescales due to other types of historical disturbance. The author knows of no study which investigates the greenhouse gas balance of these habitats and study of these may provide more general insight into what effects well-established, mature native woodland have on deep peat.

It would be useful to improve understanding of factors which affect if a wooded peatland area is a source or a sink of greenhouse gases and at what tree density/biomass this switch may occur. Chapters 4-5 indicated that birch scrub can exist on rewetted peatland sites without having a detrimental effect on net greenhouse gas emissions. With an average height of 2.54 m and a density of 0.51 trees/m² the plot studied contained what would be considered a substantial amount of scrub. However, the data, although not significant, suggested that the area where trees had established was a weaker CO₂ sink than the open plot. It is unclear at what level of tree cover/biomass that the site might become a net source. More research on the effects of birch scrub on peatlands in different contexts (e.g. different management backgrounds, different density/size of trees) may be useful to gain a better understanding of if and when self-seeded trees have an adverse effect on greenhouse flux.

At the Flanders Moss wooded plot further research could measure fluxes from a control area and additional area both before and after carrying out partial or complete tree removal, in order to assess the effects of this management.

Purposely created Peatland Edge Woodland

The PEW guidance explicitly suggests that the concept of PEW could be refined in light of improvements in understanding (Forestry Commission Scotland, 2015). Chapter 2 identified a range of interpretations of PEW and highlighted that some land managers are starting to create PEW at various sites and in various ways. Forestry and peatland management requires relatively long timescales for the effects of management options to be fully understood. The fact that land managers are already trying to establish PEW indicates that practitioner understanding of establishing and managing PEW should now be developing. As more sites are established and more time passes there will be greater scope to carry out scientific investigation of purposely established PEW sites.

The wooded plot discussed in Chapters 4-5 was identified as an intermediate intensity PEW-like site. It had many features of a low intensity PEW site, but a high density of the trees and

a relatively substantial presence of floral species associated with woodland. It would have been interesting to be able to include a PEW proxy site that fully represented low intensity PEW. However, the intermediate PEW-like site studied was a net sink for CO₂ which indicates that even lower intensity PEWs might also be net sinks for CO₂.

None of the field sites in this study matched the ecotone model for PEW. This is therefore a key type of site for future study. Some participants in Chapter 2 who advocated for an ecotone model of PEW imagined PEW as being part of a long-term, large scale approach to managing the landscape. This may mean that an understanding of this approach might develop slowly over time as land managers allow the gradual development of PEW habitats. Such sites could be used as case studies for the ecotone model.

The thesis highlights how native and non-native trees can establish on degraded peatlands without human intervention. In Chapter 2 some participants were concerned that PEW is a way to legitimise land managers not investing in forest-to-bog restoration and instead letting disused afforested peatlands become inundated with trees. Other participants saw PEW as a valuable development in policy which allows working with the natural process of tree colonisation on degraded peatland sites to create valuable habitats. More time will allow assessment of the extent to which PEW can be established by allowing natural regeneration, what sort of tree composition and densities this creates, and how much management is required to maintain a specific tree density or composition.

Establishing Peatland Edge Woodland in combination with high intensity forest-to-bog restoration approaches

Both of the PEW proxy field sites were established on sites either not using any form of forest-to-bog restoration technique (Chapter 3) or using low intensity forest-to-bog restoration techniques such as leaving brash on site and drain blocking. Since PEW was proposed and since research for this thesis began, new approaches to forest-to-bog restoration have become more prominent and priorities have been shifting. For example, fire prevention is an increasing concern to land managers and this would be a factor in the consideration of leaving woody remains on site. At both of the field sites trees were at least partly able to grow due to the presence of the old ploughing micro-topography which created relatively dry ridges where trees could grow. Ground smoothing techniques are now a well-established way to restore peatlands. It is unclear if or how well trees might be able to establish on peatlands restored in this way. If these higher intensity

restoration techniques entirely replace lower intensity methods, then there may be less scope for PEW to establish or be established. Higher intensity restoration techniques are still relatively new and the effects they have on the speed and extent of the recovery of ecosystem services (such as being a net carbon sink) are relatively poorly studied compared to older, lower intensity approaches. It is hoped that higher intensity methods will facilitate more rapid recovery of sites being net carbon sinks. Chapter 4 compared an area which was a PEW proxy with an area of low intensity forest-to-bog restoration. It would be helpful to have comparable research on the greenhouse gas balance of areas which are PEW or PEW proxies as contrasted to adjacent areas which have been restored using high intensity methods.

In the eight years since PEW was first proposed in policy there have been many developments in forest-to-bog restoration approaches. Furthermore, the focus of governments on climate change mitigation, and specifically on the role of peatland management as relevant to that goal, has intensified. This has led to increasing investment in forest-to-bog restoration. The original PEW guidelines suggest it being established on areas not suitable for either conventional restocking or forest-to-bog restoration, with the implication being that some sites are too challenging to be worthwhile to restore to open bog. However, the increasing investment and focus on peatland restoration may mean the attitudes that led to the proposal of PEW in 2014 are becoming less relevant. It should be noted that even the data for Chapter 2 is no longer completely contemporary for a rapidly developing policy area, having been collected in 2018/2019.

Other impacts

The greenhouse gas effects of management options are just one amongst many considerations in managing peatland areas. In particular, Chapter 2 highlighted biodiversity as a major factor in stakeholders' thoughts about establishing PEW, with stakeholders showing excitement about which species PEW habitats might be able to support, and about the idea of recreating a low-density ecotonal woodland which has largely been lost from the Scottish landscape. Thus, although it may be important to land managers that PEW does not have adverse effects on greenhouse gas balance the ultimate motivation may be more often focused on the biodiversity impact of PEW. The biodiversity impacts of PEW are therefore an important research area, making more research on a range of taxonomic groups needful. The Flanders Moss data set recorded some differences between vegetation in the plots, but this data was not assessed in terms of its implications for biodiversity or compared to a near-natural control on the site. Stunted birch trees

have been identified by the former Flanders Moss NNR reserve's manager as providing important habitats for several rare invertebrate species such as the Rannoch brindled beauty (*Lycia lapponaria*) (Pickett, 2004). It would be valuable to establish more thorough understanding of these species and others supported by PEW and PEW-like habitats, in order to further assess the biodiversity benefit of PEW.

Communication

The PEW concept originated within the forestry sector, but interest in PEW was also found in a range of other sectors such as conservation NGOs and statutory bodies. Chapter 2 highlights that regardless of improvement in scientific understanding, ideological and priority differences between different land managers and policy makers will always exist. However, Chapter 2 also highlighted a degree of antagonism between stakeholder groups when the analysis showed that there were similarities between stakeholder groups. In order for the concept of PEW to be more fully developed there may be a need to promote further dialogue between stakeholder groups. Given the current scarcity of PEW there are limited case studies of PEW to learn from; greater cooperation and dialogue is therefore likely to facilitate improved understanding of PEW in the land-managing community.

This thesis provides new evidence on the management options available for afforested peatlands and is the first piece of research specifically looking at the effects of Peatland Edge Woodland. It is hoped that this work and its published outcomes will prompt further discussion and understanding of Peatland Edge Woodland as a management option.

Appendices

Summary:

- **Appendix 1:** The interview schedule for the qualitative survey (Chapter 2)
- **Appendix 2:** The quantitative survey, originally presented on Qualtrics (Provo, UT) (Chapter 2)
- **Appendix 3:** Data tables augmenting the data presented from Rumster Forest (Chapter 3)
- **Appendix 4:** Data loggers on Flanders Moss (Chapter 4)
- **Appendix 5:** Long term monitoring at Flanders Moss (Chapters 4-5)

Appendix 1. The interview schedule for the qualitative survey (pertains to Chapter 2)

1. Could you briefly describe your job role?
Could you tell me a little bit more about [your institution?]
How did you come to take up this role?
2. Do you know when you first heard the term Peatland Edge Woodland?
Who from?
Why?
3. What do you understand by Peatland Edge Woodland?
What sort of tree species composition?
How would it be structured?
What sort of peat depths may it be suitable to be established on?

How have you reached that understanding?
What do you think of the Forestry Commission guidelines on peatland edge woodland?
How influential are they for your interpretation of Peatland Edge Woodland?
Would you modify them?

Have you encountered the Forestry Commissions guidance and practice guides which discuss peatland edge woodland?
4. How would you advise that a Peatland Edge Woodland should be created and maintained on a site?
Would you attempt to block drains? ("little or no artificial drainage")
How would you establish trees? (planting or regeneration)
Do you see it as a low cost management strategy with little management or one with more intensive management? (is this supposed to be cheap and easy or a good habitat?)
5. What considerations do you think are important when determining where to establish Peatland Edge Woodland
How did you decide which areas to restore, keep at conventional density or convert to Peatland Edge Woodland?
Would you only consider it for sites which have already been afforested?
Is the type of bog important? Just on ex plantation?
Would the human uses of the area affect decisions about peatland edge woodland?
Would you expect peatland edge woodland to have differing structure/compositions across Scotland?
How much peatland edge woodland do you think it might be sensible to establish across Scotland? --- why?
6. How well known would you say that the notion of peatland edge woodland is in the [your organisation]
What overall sort of feeling would you say these organisations had towards peatland edge woodland?
Is the creation of peatland edge woodland something that [your organisation] is involved in?

If so, could probe for extent. If not, is there a reason why? Might want to do it but unable for some reason. Could a Peatland Edge Woodland Project qualify for funding under the Peatland Code?

7. What do you see as the potential benefits or opportunities of establishing peatland edge woodland?

Ask about if they see it as a natural or artificial habitat

8. What do you see as the risks or challenges of establishing peatland edge woodland?

9. To what extent do you think there are uncertainties around the effects of peatland edge woodland

10. What do you see as the long-term outcomes of creating peatland edge woodland?

What should happen to peatland edge woodland if future research results shows that they result in greater net greenhouse gases release than production forestry plantations would?

11. Do you know of any other peatland edge woodland sites or other groups that are creating peatland edge woodland? Or, anyone else you think would be useful to the research

Appendix 2. The quantitative survey, originally presented on Qualtrics - Provo, UT (pertains to Chapter 2)

Peatland Edge Woodland Survey

Start of Block: Consent

Participant's Consent (required to participate)

I give consent for my answers to be used by this project including in publications.

I am aware that my participation in the survey is entirely voluntary and that my responses will be anonymous.

I am aware that once submitted I cannot withdraw my answers since there will be no way of identifying which answers are mine.

I agree to only answer this survey once.

I consent to taking part in this survey (1)

End of Block: Consent

Start of Block: Participants Details (1/6)

Participant's Details (1/6)

* Required

Q1. I confirm that I have a professional interest in Scottish afforested peatlands or forest-to-bog restoration. *

I confirm (1)

Q2. Who is your employer/what is your professional interest in afforested peatlands? (please pick the one most relevant to you) *

Conservation charity (1)

Research organisation or university (2)

Forest Research (3)

Public sector forestry (e.g. Forestry Commission Scotland) (4)

Other governmental or statutory bodies (5)

Private sector forestry (e.g. private forestry company, forestry consultants) (6)

Private land owner or land manager (7)

Other private sector organisation (e.g. utilities companies) (8)

Other (9) _____

End of Block: Participants Details (1/6)

Start of Block: Awareness and Reception (2/6)

Awareness and Reception (2/6)

Introduction

Please read the following definition of Peatland Edge Woodland:

Peatland Edge Woodland is a future management option for afforested deep peat (>50cm) defined in Forestry Commission Scotland's Practice Guide: [Deciding Future Management Options for Afforested Deep Peatland](#) (2015) as:

“a low-density woodland which avoids the net carbon loss that would result from conventional restocking on unsuitable land and combines some of the biodiversity and visual benefits of woodland and peatland. It is designed for afforested land after first rotation which is not a presumption to restore and is considered, following this assessment process, to be neither suitable for conventional restocking nor a good candidate for restoration.” It is envisaged as predominately native woodland, but an element of non-native regeneration may be allowed if it helps “secure a positive carbon balance, provided this regeneration does not compromise the growth of native planting on the site.”

It is proposed as a "low density, low intensity" woodland for sites that "are neither a good candidate for conventional restocking nor for restoration (because the peat is significantly damaged by the first rotation)"

For all questions in this survey Peatland Edge Woodland will be defined according to this definition (as a low-density, predominately native woodland established on deep peat after a first rotation of forestry).

3. Are you familiar with the above definition of the term Peatland Edge Woodland?

- No (1)
 - I have heard of the term but was not aware of how it has been defined (2)
 - I have heard of the term and was aware of this definition (3)
-

4. Have you been involved in managing or planning the creation of Peatland Edge Woodland?

Yes (1)

No (2)

(Optional) If yes, where?

5. Do you think there are situations where creating Peatland Edge Woodland could be a good option for the management of deep peat sites after commercial forestry?

I foresee no situation where it could be a good option (1)

Could be a good option in exceptional circumstances (2)

Could occasionally be a good option (3)

Could be a good option at several sites (4)

Could be a good option at many sites (5)

6. In your experience how well known is the idea of Peatland Edge Woodland in your organisation and others working in peatlands and forestry.

Poorly known (1)

A bit known (2)

Well known (3)

7. To what extent do you think the popularity of Peatland Edge Woodland might be limited by the following issues?

	Strong limiting factor (1)	Slight limiting factor (2)	Not a limiting factor (3)
Insufficient time elapsed since its original proposal in 2014/2015 (1)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Doubts that it will deliver benefits (2)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Lack of guidance or case studies on creating Peatland Edge Woodland (3)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Too different from current expectations for how peatlands should be managed (4)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

End of Block: Awareness and Reception (2/6)

Start of Block: Establishment and Management (3/6)

Establishment and Management (3/6)

8. How much tree growth would you consider ideal in a typical Peatland Edge Woodland?

- No growth (I am against Peatland Edge Woodland) (1)
 - Scattered stunted trees (2)
 - Mixture of open patches and patches of stunted trees (3)
 - Continuous cover of stunted trees (4)
 - Mixture of open patches and patches of reasonably well growing trees (5)
 - Continuous cover of reasonably well growing trees (6)
-

9. In a typical Peatland Edge Woodland what proportion of the canopy do you think could be accepted as non-native?

- I am against any Peatland Edge Woodland regardless of the level of non-native tree species control (1)
 - Less than 1% (2)
 - Less than 5% (3)
 - Less than 20% (4)
 - Less than 49% (5)
 - No limit (6)
-

10. Which of the following approaches would you consider using to establish Peatland Edge Woodland?

	Yes (1)	Possibly (2)	No (3)
Allowing natural regeneration of native trees on a deep peat site (1)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Actively encourage native tree regeneration on a deep peat site by planting adjacent shallow peat or mineral peat with desired tree species (2)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Actively establish Peatland Edge Woodland by planting trees on the deep peat (3)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

11. How important do you think it would be to remediate historic artificial drainage on a Peatland Edge Woodland creation site?

- No remediation would typically be required (1)
- It would typically be important to block drains (2)
- It would typically be important to block drains and reduce drainage via the original plough furrows (for example by blocking, ground smoothing or stump flipping) (3)

12. How expensive do you think Peatland Edge Woodland would typically be as a management option relative to a typical open-bog restoration project at the same scale?

- A lot cheaper (1)
 - A bit cheaper (2)
 - About the same price (3)
 - A bit more expensive (4)
 - A lot more expensive (5)
-

13. **(Optional)** What work would you do to establish a Peatland Edge Woodland site, and how much would you expect it to cost?

14. Please rate the following images according to how good a model for Peatland Edge Woodland they appear to you (if you wish you can enter comments on your ratings in Question 15)

- 1 star = Bad model
 - 2 stars = Quite a bad model
 - 3 stars = Okay model
 - 4 stars = Quite a good model
 - 5 stars = Good model
-

Image A
(below)



Image B
(below)





Image C
(below)



Image D
(below)



Image E
(below)





15. **(Optional)** Why did you give these ratings?

End of Block: Establishment and Management (3/6)

Start of Block: Motivations (4/6)

Motivations (4/6)

16. Do you agree or disagree that Peatland Edge Woodland "offers the best benefits" when a site is "neither a good candidate for conventional restocking nor for restoration"?

- Strongly agree (1)
- Somewhat agree (2)
- Neither agree nor disagree (3)
- Somewhat disagree (4)
- Strongly disagree (5)

17. How important would the following be in influencing your judgement of whether a site is unsuitable for open bog restoration?

	Important (1)	A bit important (2)	Not important (3)
Practically unrestorable - site too badly damaged to be possible to restore (1)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Financial unrestorable - too costly to restore and keep free of trees (2)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Surrounding land-use constraints - e.g. adjacent land-uses prevent blocking of drains, adjacent afforested land would be a persistent seed source etc. (3)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Social/aesthetic constraints - local people/land-users support the presence of trees (4)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Restoration would take too long to deliver desired benefits (5)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

18. Which of the following benefits do you think Peatland Edge Woodland creation could provide at appropriate sites?

	Yes (1)	Maybe (2)	No (3)
Conservation of biodiversity (1)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Climate change mitigation (2)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Improved habitat connectivity (3)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Reduced spread of non-native tree species out of adjacent plantations (4)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Provide wood (e.g. for fuel) (5)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Create a more natural mosaic of habitats within a landscape (6)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Recreate habitats analogous to natural bog woodlands found in Scotland and internationally (7)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Improved landscape aesthetics (8)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

End of Block: Motivations (4/6)

Start of Block: Concerns (5/6)

Concerns (5/6)

19. How important to you are the following concerns about the creation of Peatland Edge Woodland?

	A concern (1)	A bit of a concern (2)	Not a concern (3)
Being able to get native trees established and growing (1)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Being able to control excessive non-native tree regeneration (2)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Being able to control excessive native tree regeneration within Peatland Edge Woodland (3)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Being able to control native tree regeneration onto surrounding open peatland (4)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Whether it won't be beneficial for climate change mitigation (5)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Whether it won't deliver biodiversity benefits (6)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Whether forest expansion targets will lead to its creation on sites that could be restored to open bog (7)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
It being an artificial habitat (8)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Lack of research and case studies identifying potential benefits and how to achieve them (9)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>
Being costly to create and maintain (10)	<input type="radio"/>	<input type="radio"/>	<input type="radio"/>

End of Block: Concerns (5/6)

Start of Block: Additional Comments (6/6)

Additional Comments (6/6)

20. **(Optional)** We welcome you to share any other thoughts you might have on Peatland Edge Woodland or this survey. Remember this survey is anonymous so we will be unable to respond to any comments. If you would rather you can email me directly with comments (Will Jessop at wj573@york.ac.uk).

To submit your survey please click the right-hand arrow below. Thank you

End of Block: Additional Comments (6/6)

Appendix 3. Data tables for the seasonal and plot comparison (pertains to Chapter 4)

Appendix 3a. F and p values for significance tests of within-subjects contrasts for sequential pairwise comparisons of the month

Seasonal time point comparisons	CH ₄	ER	GPP	NEE	Pore water [DOC]	Pore water [NH ₄ ⁺]	Pore water [PO ₄ ⁻]	Soil moisture	Soil temp_5cm	Soil temp_10cm	Water table depth
October 2019 vs. January 2020	146.572 (<0.001)	132.713 (<0.001)	13.675 (0.001)	0.544 (0.468)	542.625 (<0.001)	9.109 (0.006)	42.217 (<0.001)	3.324 (0.081)	643.776 (<0.001)	1085.405 (<0.001)	62.623 (<0.001)
January 2020 vs. February 2020	3.144 (0.089)	31.749 (<0.001)	0.147 (0.704)	0.727 (0.402)	104.643 (<0.001)	1.091 (0.307)	1.674 (0.208)	1.354 (0.256)	309.588 (<0.001)	219.546 (<0.001)	246.721 (<0.001)
February 2020 vs. July 2020	26.072 (<0.002)	554.012 (<0.001)	41.230 (<0.001)	3.325 (0.081)	128.141 (<0.001)	5.580 (0.027)	10.266 (0.004)	0.354 (0.558)	2976.691 (<0.001)	6464.584 (<0.001)	6.776 (0.016)
July 2020 vs August 2020	8.739 (0.007)	36.934 (<0.001)	11.825 (0.002)	0.017 (0.896)	16.988 (<0.001)	2.769 (0.109)	31.884 (<0.001)	6.291 (0.019)	41.354 (<0.001)	40.886 (<0.001)	2.875 (0.103)
August 2020 vs September 2020	11.888 (0.002)	100.282 (<0.001)	29.310 (<0.001)	2.765 (0.109)	18.893 (<0.001)	11.319 (0.003)	64.351 (<0.001)	6.974 (0.014)	302.320 (<0.001)	629.247 (<0.001)	150.230 (<0.001)
September 2020 vs October 2020	5.774 (0.024)	7.627 (<0.011)	37.141 (<0.001)	23.235 (<0.001)	0.150 (0.702)	0.958 (0.337)	2.144 (0.156)	5.815 (0.024)	0.152 (0.700)	0.041 (0.842)	504.906 (<0.001)
October 2020 – May 2021	9.827 (0.004)	11.820 (<0.002)	25.811 (<0.001)	12.091 (0.002)	95.185 (<0.001)	16.010 (<0.001)	0.521 (0.477)	0.013 (0.909)	0.632 (0.435)	92.808 (<0.001)	41.798 (<0.001)

Appendix 3b. Means and 95% confidence intervals for different fluxes, pore water nutrient and DOC concentrations, and environmental variables which were measured at the 30 sampling points at 8 time points. Means and confidence intervals are split according to plot and topographic level. Means are the first value in each cell and 95% confidence intervals are the second value, in brackets.

Dependent variable	All collars		Ridge collars		Intermediate collars		Furrow collars	
	Wooded	Open	Wooded	Open	Wooded	Open	Wooded	Open
CH ₄ (μmol m ⁻² s ⁻¹)	0.0050 (0.0015)	0.0123 (0.0026)	0.0020 (0.0011)	0.0048 (0.0017)	0.0069 (0.0037)	0.0164 (0.0045)	0.0061 (0.0024)	0.01581 (0.00527)
ER (μmol m ⁻² s ⁻¹)	0.4776 (0.0901)	0.4717 (0.0759)	0.5323 (0.1763)	0.5330 (0.1477)	0.5731 (0.1709)	0.4978 (0.1224)	0.3275 (0.1032)	0.3842 (0.1213)
GPP (μmol m ⁻² s ⁻¹)	-0.7947 (0.1491)	-0.8986 (0.1563)	-0.7539 (0.2698)	-0.6477 (0.1786)	-0.9391 (0.2975)	-1.1532 (0.2674)	-0.6911 (0.1972)	-0.8947 (0.3282)
NEE (μmol m ⁻² s ⁻¹)	-0.3171 (0.0822)	-0.4269 (0.1120)	-0.2216 (0.1607)	-0.1147 (0.1036)	-0.3660 (0.1481)	-0.6554 (0.1792)	-0.3637 (0.1132)	-0.5106 (0.2376)
Pore water [DOC] (mg/L)	44.0367 (4.6354)	45.8297 (4.6312)	42.5724 (5.6764)	34.1362 (4.7642)	60.2526 (9.9564)	44.6185 (6.6466)	29.2852 (4.2177)	58.7343 (9.9720)
Pore water [NH ₄ ⁺] (mg/L)	0.1004 (0.0166)	0.1124 (0.0302)	0.1239 (0.0300)	0.0881 (0.0202)	0.0885 (0.0310)	0.1745 (0.0850)	0.0889 (0.0239)	0.0745 (0.0116)
Pore water [PO ₄ ³⁻] (mg/L)	0.0450 (0.0045)	0.0432 (0.0036)	0.0626 (0.0106)	0.0412 (0.0048)	0.0379 (0.0038)	0.0519 (0.0080)	0.0344 (0.0039)	0.0366 (0.0041)
Soil moisture (%)	0.6091 (0.0395)	0.7475 (0.0217)	0.5118 (0.0663)	0.6453 (0.0458)	0.5576 (0.0655)	0.7700 (0.0215)	0.7581 (0.0471)	0.8270 (0.0065)
Soil temp_5cm (°C)	10.8033 (0.8513)	10.8942 (0.7958)	11.1225 (1.5732)	11.3800 (1.4957)	10.7300 (1.4226)	10.7225 (1.3579)	10.5575 (1.4554)	10.5800 (1.2962)
Soil temp_10cm (°C)	9.9200 (0.6975)	10.0008 (0.7070)	10.0825 (1.3162)	10.1650 (1.2760)	9.9325 (1.1792)	9.9825 (1.2354)	9.7450 (1.1512)	9.8550 (1.1903)
Water table depth (cm)	6.0733 (1.6311)	2.0317 (1.6069)	16.5683 (0.7852)	13.0125 (1.1011)	4.8133 (1.6959)	-0.1275 (1.2976)	-3.1617 (1.2064)	-6.7900 (0.8604)

Appendix 4. Dataloggers on Flanders Moss (pertains to Chapter 4)

From late August 2019 to late May 2021 at my Flanders Moss field site water table depth, PAR, soil temperature at 10cm depth, and air temperature were continuously measured with dataloggers installed in both plots described in Chapters 4 and 5. Wind speed was also measured from February 2020. The variables measured can impact on gas fluxes and some of these variables may also affect peat pore water DOC/nutrient concentration. This data could be used for more complex models to estimate annual CH₄ and CO₂ gas fluxes.

PAR was measured every half hour with a single Quantum Sensor (Delta-T Devices Ltd) connected to a GP1 datalogger (Delta-T Devices Ltd) and positioned in an unshaded location between the two plots. Wind speed was measured at one location between the two plots at 1 meter height with a Type A100R Ser. 2995 Switching Anemometer (Vector Instruments/Windspeed Ltd.) connected to a GP1 datalogger (Delta-T Devices Ltd.). The other variables were measured with sensors and dataloggers for the entirety of the study period at each sampling point in 1 block from each plot (a total of 6 sampling points). However, some equipment failure means there are some gaps where data is not available for all 6 sampling points. Soil moisture was measured with ML2x Theta probes (Delta-T Devices Ltd.) connected to GP1 datalogger (Delta-T Devices Ltd.), soil and air temperature were measured with HOBO U23-003 Pro V2 (Onset Computer Corporation) and water table depth was measured with TROLL500 (In-Situ Inc.).

Since the PAR Quantum Sensors were old the calibration of the PAR sensors was checked against a brand-new, more advanced PAR sensor (a QS5). The PAR sensors used did not give significantly different results to this new sensor. The temperature and moisture sensors were calibrated against the handheld sensors, as result some sensors were replaced, but the final selection of sensors showed close agreement between each other.

Appendix 5. Long term monitoring at Flanders Moss (pertains to Chapters 4-5)

For the two plots used in Flanders Moss there were long-term vegetation data, gathered primarily by Russell Anderson. Data was collected at 6 points in each plot. Data was collected on three variables; vegetation percentage cover by species, water table depth, and bulk density at 10-20 cm depth and 70-80cm depth. Data was collected on all variables before restoration work was carried out in the summer of 1998. Water table depth was collected in December 1997 and then repeatedly in the years 1998, 1999, 2000, 2001, 2008, and 2009. Bulk density was measured in 1997 and 2000. Vegetation cover was measured in 1997, 1999, 2000, 2003, and 2008.

I followed the methodology of this previous data collection to create a new time point for vegetation cover and bulk density in 2020, and measured water table depth four times in August 2020, September 2020, October 2020, and May 2021. The originally sampling points used in the previous data could not be found due to overgrowth of vegetation and decay of the old markers but their approximate location were estimated following the old experiments notes. There was some evidence of the success of this approach as decayed evidence of the original sampling points was found at some of the new sampling points.

This combined data set was not analysed as part of this thesis due to time constraints but could be used in future work to give a better understanding of the developing trajectory of the sites as discussed in the general discussion.

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