Quantifying Carbon Accumulation and Loss in Afforested Peatlands

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Abstract

During the 20th Century, new tree planting techniques combined with tax incentives encouraged the establishment of forest plantations across large areas of peatland in the UK. As many of the stands reach harvesting age the question arises whether the bogs should be restored or restocked with new trees.

Bad a'Cheo Forest (Caithness, Scotland), which was afforested in 1968, is once such plantation. Ground elevation and peat depth surveys were compared against re-interpolated data from before afforestation, and across the full forestry rotation. Significant subsidence has taken place since drainage, with an average reduction of 53 cm (13 %) in peat depth under forest stands, suggesting a possible loss of carbon from the peat.

To confirm this Icelandic cryptotephra, most notably Hekla 4 (2310 ± 20 BCE), were used to define isochrones in peat cores, allowing for comparison of carbon stocks. ITRAX core scanning was used on 27 cores from Bad a'Cheo and on eight additional cores from four other sites around the Flow Country to rapidly identify the presence of elements indicative of tephra deposits. Hekla 4 was found significantly closer to the surface in afforested bogs (average 134.4 ± 16.3 cm) than in undrained bogs (244.0 ± 44.6 cm).

Forestry had caused an average loss of stored carbon from peat of 66.8 t C ha^{-1} at Bad a'Cheo, and an average loss of 103 t C ha^{-1} across all sites. At Bad a'Cheo carbon uptake in primarily aboveground forest biomass, partially compensated for this, producing a net loss of 19.49 t C ha^{-1} of afforested peatland over 50 years. High variation in the data produced uncertainty in the estimates, with possible outcomes ranging from a total net gain of carbon across peat and forestry of 266.72 t C ha^{-1} to a significant net loss of 305.69 t C ha^{-1}.

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Author's 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.

The co-authors of the papers presented in this thesis affirm that they are substantively the work of the PhD candidate, T. Sloan.

Primary data collection was planned and executed by T. Sloan. Field work was carried out by T. Sloan with assistance from P. Gilbert, R. Payne, R. Andersen, R. Anderson and others acknowledged in the individual papers. Laboratory work was carried out by T. Sloan with assistance from J. Serafin, P. Gilbert and others acknowledged in the individual papers. Data analysis was by T. Sloan. In chapter 4, ITRAX scanning was carried out by T. Bishop. All papers were written by the candidate with co-authors contributing edits and guidance.

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Chapters 2 and 3 have been amended at the request of the thesis examiners, and now differ slightly from the published versions.

Chapter 1 - Introduction

Research background and context

Peatland afforestation in the UK

In the second half of the twentieth century, large areas of UK peatlands were drained and afforested. This was made possible by changes in forestry practices and improvements in technology, especially in ploughing equipment (Pyatt 1990, Oosthoek 2013). Even with such developments these peatlands, especially deep peat blanket bogs such as those of the Flow Country of northern Scotland, remained marginal for forestry and were therefore likely to produce a wood crop of poor quality. Nevertheless, between 9 and 15 % of UK peatlands were planted between the 1960s and late 1980s (Hargreaves *et al.* 2003, Payne & Jessop 2018), which can be attributed to the tax regime that was in place (Mather & Murray 1988). This tax system provided massive incentives to invest in forestry as part of a longstanding national strategy to extend woodland cover across the UK (Marren 2002). However, these tax incentives did not specify the types of land that should be brought into cultivation, nor made requirements on the quality of the wood ultimately produced. They were therefore used by wealthy individuals and organisations to reap tax benefits whilst damaging areas of peatland through afforestation (Mather 1986).

The impact of afforestation was widespread, especially in the Flow Country where almost 17 % of the 400,000 ha region was planted (Stroud *et al.* 1987). At the time, public awareness and opposition to afforestation focused on the use of peatlands as a habitat for many species of bird and mammal (Thompson 1987, Lindsay *et al.* 1988). With increasing awareness of the importance of peatlands for carbon storage, the potential consequences of afforestation to climate change have become of more pressing concern (Warren 2000). This thesis investigates the extent to which the peatland forestry led to a loss of carbon from these landscapes.

Peatland carbon sequestration and storage

Globally, peatlands store between 400 and 600 Gt of carbon (Gorham 1991, Yu 2011) and cover approximately 3% of the earth surface (Dise 2009). This represents an annual average estimated net sink of carbon of 44GtC ka^{-1} (Yu 2011), and a consequent small net cooling of the climate over last 1000 years (Charman *et al.* 2013). There are suggestions that the pattern of uptake through the Holocene has been non-linear with higher accumulation in the early Holocene due to increased seasonality and maximum summer insolation (Yu *et al.* 2010). Other studies have argued that there remains a significant active uptake of greenhouse gases which has not decreased through the late Holocene (Lafleur *et al.* 2003, Roulet *et al.* 2007). While peatlands have accumulated large stocks of carbon over the Holocene, degradation of peatlands have shifted many to be a net source of carbon emissions (Joosten 2010), a process exacerbated by projected climate change (Gallego-Sala & Prentice 2013, Ferretto *et al.* 2019). In either case, peatlands represent massive terrestrial stores of carbon, the fate of which is significant to future global warming.

A peatland is an area of wetland that accumulates peat. Peat itself is an aggregate of semi-decayed organic matter which does not fully decay due to wet and therefore anaerobic conditions. While this leads to production of CH_4, the minimisation of loss of CO_2 via respiration produces a net accumulation of carbon. The blanket bogs that are the focus of this thesis are peat deposits which form in wet hollows in the landscape and spread in areas of high rainfall and low evapotranspiration. Over time the bog grows to 'blanket' the underlying topography, covering it with a characteristic 'dome' of peat. Such bogs are ombrotrophic if they derive their moisture solely from atmospheric sources. Blanket bogs are globally rare as a type of peatland because they form in high latitude, cool and oceanic climates (Loisel *et al.* 2014), and consequently the British Isles hold a relatively large proportion of the world's blanket bogs (Garnett *et al.* 2001). While the UK is estimated to hold between 17,000 – 22,000 km^2 of peatlands (0.43 – 0.55 % of overall global total of approximately 4 million km^2) (Kaat & Joosten 2008, Joosten 2020, Xu *et al.* 2018), it is estimated to hold up to 13 % of the global total of blanket bogs (Lindsay et al. 1988).

Extensive formation of modern peatlands coincided with the end of the last glacial maximum, beginning 16500 years ago, as ice coverage receded and water pooled to form anaerobic areas where there could be little decomposition. Peat formation increased greatly between 12,000 and 8000 years ago (Macdonald *et al.* 2006), although there is evidence for earlier peatland formation during the last interglacial 130,000 to 116,000 years ago (Treat *et al.* 2019) and throughout the Quaternary. While natural climatic conditions facilitate the formation of peatlands, throughout the Holocene peatlands were modified by humans (Billett *et al.* 2010). Mesolithic humans may have had a role in removing trees during initial peat formation through burning, felling small trees, and ringing large trees with axes. The purpose of this would be to create or expand clearings for animal grazing (Simmons 2003). The thinning of trees reduces water interception, increasing the wetness of the system.

Ecosystem services are the human benefits derived, directly and indirectly, from the natural environment (Fisher *et al.* 2009). The exploitation of peat has provided ecosystem services to humans throughout the Holocene. This has included small scale activities such as peat cutting and grazing through to commercial exploitation for fuel and leisure (Whitfield *et al.* 2011). Peatland forestry is a relatively new development, but is akin to other commercial uses of peat that dry peatlands and thus cause degradation through exposing greater depths to oxidative loss of carbon (Holden *et al.* 2007, Swindles *et al.* 2019). Put in this context, the afforestation of UK peat bogs fits with a pattern of peatland exploitation globally (Joosten *et al.* 2012) although more evidence is needed to quantify the damage caused.

Research into peatland afforestation

The development of peatland forestry in the UK had lagged behind that of northern peatlands in other countries up to the mid twentieth century (Zehetmayer 1954), in particular those of Fennoscandia where up to 25 % of the nation's exploited forests grow on peat (Laiho & Laine 1997). That these areas were afforested earlier, and with less mechanical disturbance, is attributable to a different climate and the prevalence of minerotrophic fens and naturally wooded bogs, which often required less drainage. These may have remained or become a greater net sink of carbon than previously (Silvola *et al.* 1996, Minkkinen *et al.* 2002, Maljanen *et al.* 2010, Ojanen *et al.* 2013) although subsequent re-sampling of some sites and new modelling have challenged some of these conclusions (Simola *et al.* 2012, He *et al.* 2016). Because of the differing climate and required level of drainage and other interventions in the UK meaning a more intensive effort is required to afforest UK bogs, the carbon balance is likely to be different from Fennoscandia.

The UK evidence base for the impact of peatland afforestation is limited, with very few studies directly assessing changes in peat carbon content (Hargreaves *et al.* 2003, Byrne & Farrell 2005, Yamulki *et al.* 2013), and in some cases there are serious criticisms of these studies (Lindsay 2010, Artz *et al.* 2013, Sloan *et al.* 2018). Many of the assumptions carried over from Fennoscandia studies about tree carbon uptake are also unsound as the wood produced on UK plantations which tend to produce lower yield, less economically viable (Laine *et al.* 2009, Payne *et al.* 2018). There is therefore an urgent need to address the lack of UK specific data.

Widespread afforestation of UK peat began in the second half of the twentieth century. The initial forestry rotations are now coming to an end, and many of the earliest plantations have already been logged. Bad a'Cheo forest, used in this study, was felled in 2017, while elsewhere in the Flow Country widespread restoration is underway at other sites such as the RSPB Forsinard Flows nature reserve. As the bulk of the peatland plantations finish their rotation, it is vital to know the exact balance of carbon losses and gains in order to make policy decisions as to whether these plantations should be commercially restocked with new trees, restored to open, undrained blanket bog habitats, or turned into novel ecosystems harbouring native tree communities on peat, on the edges of restoration areas (Forestry Commission Scotland 2015, 2016). These questions are central to this thesis.

Aims and objectives

This thesis seeks to explore the effects of afforestation on the carbon balance of UK peatlands with an emphasis on blanket bogs in the Flow Country of Scotland. It will do this by addressing several general objectives:

1. This study will assess the current evidence base for the effects of afforestation on peatlands, drawing comparisons between existing data in the UK and in other regions, in particular Fennoscandia.

2. It will quantify the changes at Bad a'Cheo, a peat bog which was drained and afforested in 1968 (see "Bad a'Cheo study site" section). This will include the physical alteration of the landscape (the widespread subsidence in the ground level) and the changes to carbon stored on within the bog.

3. It will further quantify the carbon storage provided by tree biomass at Bad a'Cheo through direct measurement of tree morphometrics.

4. Using the peat and tree carbon data for the previous two objectives it will produce a carbon budget for Bad a'Cheo, using a stock-based approach.

5. It is important to ascertain whether the results taken from our intensively studied site are applicable throughout the Flow Country. The results from Bad a'Cheo will therefore be placed into a regional context. This will be achieved by carrying out a simplified version of the Bad a'Cheo sampling protocol on several other sites that form a rough transect across the Flow Country.

6. This study uses tephrochronology to identify isochrones in peat (see "Methodology" section) (Lowe 2011). It therefore has the additional aim of further developing the methodology for rapid identification of cryptotephra deposits using ITRAX scanning (Dugmore & Newton 1992, Kylander *et al.* 2012). It aims to greatly enhance the evidence base for the geochemistry in tephra deposits in northern Scotland, using a level of replication uncommon in other studies.

Together this will meet the overarching aim of answering a pressing question in the management of UK peatlands: *Has peatland afforestation in the UK led to a net uptake or loss of carbon?* There is little data on the extent of the total effect on carbon storage of drainage of UK bogs for forestry (Sloan *et al.* 2018). This thesis will provide a comprehensive data set to that end and will contribute to the current policy discussions on whether UK afforested peatlands should be restocked for a new forestry rotation or restored through rewetting.

Methodology

Thesis structure

This thesis is presented as a sequence of four standalone papers, which initially describe the development of forestry and the current state of knowledge, develop novel methods for the rapid identification of isochrones in peat, and finally present a carbon budget for afforested UK peats.

Chapter 2, "Peatland afforestation in the UK and consequences for carbon storage", describes at length the development of the UK peatland forestry industry from its inception in the second half of the twentieth century, and the process through which peat is afforested. It outlines the theoretical framework of how carbon storage may be affected by drainage and afforestation, and outlines the existing evidence base. It develops the argument for more dedicated study of UK sites, and identifies some of the gaps in the research that this thesis seeks to answer.

Chapter 3, "Ground surface subsidence in an afforested peatland fifty years after drainage and plantation", uses a combination of interpolated historic data and modern ground level surveys to describe peat subsidence at Bad a'Cheo in Caithness, Scotland. Subsidence offers an indication that physical changes have taken place in the peat underlying afforested areas, as described in the previous chapter. This study indicates that widespread subsidence has occurred, which may be associated with carbon loss. As subsidence is only ever a loose proxy for carbon loss, the paper develops the argument that while afforestation has had an impact on the landscape, further detailed study is required to quantify carbon loss.

Chapter 4, "Local and regional-scale variability in tephra concentration and shard size in peatlands", develops one of the key methodologies which supports the carbon budget presented in chapter 5. Chapter 3 has demonstrated subsidence on site, something which may be a proxy indicating carbon loss. To quantify this properly, a stock-based approach to carbon using whole carbon inventories will give the most complete picture of net change. To accomplish this, stratigraphic markers are required to ensure that a comparison can be made between cores of peat of similar age. Tephrochronology is a useful tool for this. The chapter examines deposits of tephra across the sites, with particular emphasis on Hekla 4, dated to an eruption 2310 ± 20 BCE (Pilcher *et al.* 1995), which is a common cryptotephra in the highlands of Scotland. In addition to its utility as a stratigraphic marker, this paper describes tephra at a single site with a level of replication that, to our knowledge, has never been attempted.

Chapter 5, "A stock-based approach to quantifying carbon accumulation and loss in UK afforested peatlands", builds on the three previous chapters to provide a carbon budget for Bad a'Cheo, the intensively studied project site, as well as four other locations in the Flow Country. This includes the below ground peat carbon stock and the carbon stored in tree biomass, providing a complete carbon budget. This chapter provides the most comprehensive data set currently available for UK bog response to drainage and afforestation.

A concluding discussion chapter synthesises the results of the four papers and assesses the overall impact of peatland forestry on carbon storage, offering options for future management on this basis. This section explains how, when taken together, the papers that comprise this thesis meet the aims and objectives laid out above.

Bad a'Cheo study site

To fulfil the objectives outlined above, this thesis makes use of one of the most intensively studied afforested bog sites in the UK: The Forestry Commission Bad a'Cheo plantation in Rumster Forest, Caithness (Figure 1.1). The Bad a'Cheo bog covers an area of 50 ha and was initially surveyed (ground level and peat depth) in 1966, with trees planted two years later. The experiment contains blocks of monocultures of either Lodgepole pine (*Pinus contorta*) or Sitka spruce (*Picea sitchensis*), and mixed blocks of both Lodgepole pine and Sitka spruce planted in alternate rows. Multiple drainage and spacing techniques were trialled, with planted blocks subsequently replacing unplanted control blocks 1989 as new planting practices were developed (Miller *et al.* 1996, Anderson *et al.* 2000). Since the initial plantation, experimental work has been ongoing to assess the consequences of forestry and of drainage. The planting of Lodgepole pine led to a rapid drying of the peat, although this may not have had the anticipated beneficial effects on Sitka spruce growth (Ray & Schweizer 1994).

Pyatt *et al.* (1992) assessed the effects on the bog of afforestation on subsidence and water table levels after 20 years. Planting led to a fall in peat water content, leading to a 30 – 55 cm subsidence under the trees, with the effect tapering away 10 - 20m from the edge of the forest plots. The water table remained high beyond 20 m away (Pyatt *et al.* 1992). Subsequent research called into question the reliability of some of the initial ground level data but confirmed the overall pattern of subsidence on most of the site (Shotbolt 1997).

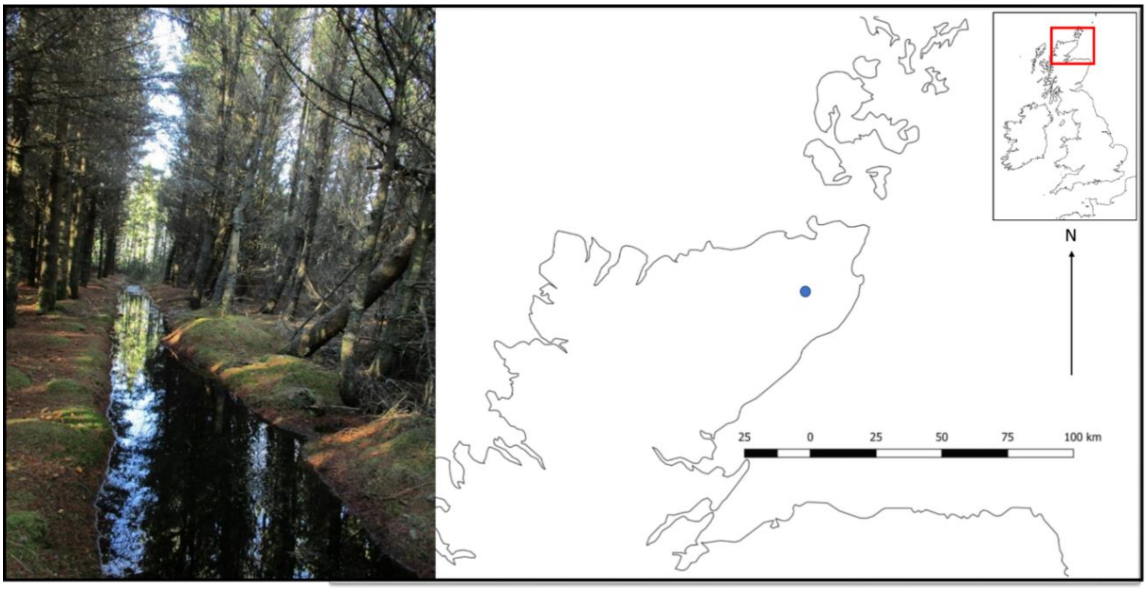

Figure 1.1. Location of the Bad a'Cheo plantation in Rumster Forest, Caithness, Scotland. An unmaintained drainage ditch has refilled, showing that the water table remains high. As a consequence the trees are poorly rooted, with some on the right of the picture having collapsed due to 'wind throw', a common occurrence in peatland plantations.

Work by Shotbolt *et al*. (1998) followed around ten years later, showing an increase in bulk density under forest stands and observing more severe cracking on site. Again, ground level was measured along transects and at random points, and compared to levels extrapolated from the original site survey, with subsidence shown to be up to 80 cm under forest plots. Further subsidence in the years following plantation was limited to slow secondary compression and oxidative loss, highlighting the strong influence of initial drainage and early tree growth in stimulating rapid primary consolidation (Shotbolt *et al.* 1998). The lateral extent of subsidence beyond the plantation by the end of the first rotation was projected to be 40 m, a figure disputed elsewhere as an underestimation of between 10 and 40 m (Lindsay 2010).

Anderson *et al*. (1992) investigated subsidence across the gaps between 22-year-old forest stands (the forest 'rides'). They used short peatland transects between woodland edges to show that the ground surface was 0.5 m lower in forest stands compared to un-forested rides (Anderson *et al.* 1992). This lower ground level was presumed to indicate subsidence, possibly with associated carbon loss from the system. However, the rides measured in this study do not provide a sufficient control, as they are close to the stands and therefore not free of hydrological footprint of the plantations.

Recent work at the site has identified bands of cryptotephra (Ratcliffe *et al.* 2017), as is common across peatlands across northern Scotland (Dugmore *et al.* 1995). Ratcliffe *et al.* (2017) used tephra isochrones to show that carbon accumulation rates derived via stock-based studies were up to six times lower than carbon accumulation rates from flux-based eddy covariance studies in the region (15.4 g C m^{-2} yr^{-1} vs 99.37 g C m^{-2} yr^{-1} respectively). These and other data from around the region, including some taken from bad a'Cheo for this thesis, have further suggested a slowing of carbon accumulation rates over the Holocene (Ratcliffe *et al.* 2018).

As a research plantation rather than an area of commercial growth, Bad a'Cheo is atypical in some ways. Although it was planted using the system of drainage and ploughing widely used in peatland forestry at the time (Figure 1.2), it was also laid out in randomised blocks, and may have more open spaces more prone to 'wind throw' than might otherwise be the case. The drainage on parts of the plantation has also not been well maintained, although the standard of maintenance is variable throughout such forests. The site is, however, one of the best monitored peatland forest plantations in the UK and offers an amount of historical information and scientific data that would not be available on a commercial plantation. That said, when relying on a site like this the factors which make it not properly representative must be taken into consideration. We have therefore surveyed additional sites from around the Flow Country to validate and upscale the Bad a'Cheo findings.

How representative is Flow Country afforestation?

The Flow Country was the most significant of several UK peatlands that were afforested as a consequence of the tax breaks offered by the government (Mather & Murray 1988). Across the UK, an estimated 190,000 ha or 9 % of deep peatlands were planted, as were 315,000 ha of shallow peatlands (Cannell et al. 1993, Artz et al. 2014). The Flow Country accounted for 67,000 ha, or approximately a third, of the deep peat planting (Stroud et al. 1987, Lindsay et al. 1988).

It is important to understand how representative both the practice and impacts of Flow Country peatland afforestation was relative to the rest of the UK, as well as how important the afforestation of the Flow Country was in and of itself. The dominance of deep peat blanket bogs in the Flow Country means that it is not an ideal analogue of afforestation on shallow peats or of minerotrophic fens. Broadly speaking, the afforestation of fens and shallow peats requires less drainage and fertilisation and is more likely to produce higher tree yields to offset carbon loss. Such peats are more analogous to the Fennoscandian afforestation (Laine et al. 2009), the evidence for which is discussed in the next chapter.

The Flow Country represents a case study of the afforestation of the deep peats which were least practical for forestry. The deep peats afforested elsewhere in the UK are analogous to those planted in the Flow Country, with a key difference that tree species typically selected may have initially varied based on the original species assemblage at the planting site (Forestry Commission 1964), or other factors such as expected yield. For example, in Northern Ireland Sitka spruce monocultures were preferred to Lodgepole pine monocultures or mixed Lodgepole and Sitka stands during the initial forestry rotations. While the drainage and preparation of such sites would be similar, it is important to remember large and continued fertiliser use would have been employed to prevent such plantations from failing completely. Changes in forestry practice leading to the eventual development of the UK Forestry Standard (Forestry Commission 2017) ultimately ensured that trees planted should be appropriate for growing with a minimum input of fertiliser, which has meant that a mixed Lodgepole and Sitka approach of the sort adopted in the Flow Country has been the default approach on subsequent forestry rotations (Forestry Commission Scotland 2015). The impacts on carbon and upcoming management choices are therefore now also likely to be the similar to those of the Flow Country described in this thesis.

While the consequence to carbon storage of afforestation in the Flow Country reflect the likely responses of other such deep peatlands, the Flow Country is also worth considering in isolation. Scotland contains a disproportionately large 46 % of total British soil carbon (Milne & Brown 1997), much of which is concentrated in the deep peats which are characteristic of the Flow Country. As the Flow Country accounted for approximately one third of the total area of afforested deep peat ion the UK, the impact on carbon storage here is of significant national importance. The Flow Country is also Britain’s largest wilderness, so as well as carbon there are biodiversity and ecosystem storage considerations inherent in fragmenting one of the largest blanket bogs in Europe (Littlewood et al. 2010).

Summary

Peatlands hold a disproportionately large carbon content relative to the land they occupy, and the Flow Country of northern Scotland is a globally important blanket bog. The drainage and afforestation which took place there in the second half of the twentieth century has impacted the carbon storage in ways that are not yet fully understood. This thesis provides a data set to that end at a time when policy decisions must be made on the future of such plantations. It further contributes to a growing body of work about the consequences of drainage of peatlands for human exploitation.

Figure 1.2. Preparation of peat ridges for planting in 1979 using a double mould board plough mounted on a low ground pressure tractor at Rumster Forest, Caithness. Bad a'Cheo forms part of the wider Rumster Forest area, and although planted earlier it would have been prepared in a similar manner. Photograph by George Day, sourced from the Forestry Memories Image Library, https://www.forestry-memories.org.uk/.

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A comprehensive background of UK peatland afforestation

Chapter one has given a broad overview of the aims and structure of the thesis, as well as an introduction to the issue of peatland afforestation. Chapter two will expand on this by providing a further context on the history and likely impacts of deep peat afforestation.

Chapter two will provide a detailed history of the practice and drivers of peatland forestry in the UK up to the present day. It will expand on the processes by which afforestation affects the carbon balance of the peatlands on which trees are planted. It will also offer a review of the research undertaken in this country and the methods used. It will finally compare this to the knowledge base in Fennoscandia and elsewhere in Europe, where the majority of peatland afforestation studies originate.

Chapter two will therefore justify the questions asked in chapter one, and the identified gaps in knowledge will inform the approaches taken and questions asked throughout the subsequent research chapters of the thesis. In particular, the exploration of the available methods for measuring changes in carbon storage in bogs will underpin methodological approaches taken in the research papers presented in this thesis.

Chapter 2 - Peatland afforestation in the UK and consequences for carbon storage

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Summary

Peatlands are a globally significant store of carbon. During the second half of the 20th century new planting techniques combined with tax incentives encouraged commercial forestry across large areas of peat bog in the UK, particularly in the Flow Country of northern Scotland. Such planting was controversial and was ultimately halted by removal of the tax incentives, and policies to prevent new planting. Here we review the literature on UK peatland afforestation in relation to carbon and climate implications, and identify key issues for future research. The effects of conifer planting on peat bog carbon storage in the UK are poorly understood. A large body of research on peatland forestry exists, particularly from naturally forested fen peatlands in Fennoscandia and Russia, but the different conditions in the UK mean that results are not directly transferable. Data on the responses of UK peat bogs to afforestation are required to address this shortfall. Studies are required that quantify the loss of carbon from the peat and evaluate it against the accumulation of carbon above and below ground in trees, considering the likely residence time of carbon in wood products.

KEY WORDS: Flow Country, forestry, GHG, greenhouse gases, peat

Introduction

Carbon storage in peat

Peatlands are globally important stores of carbon. Covering about 3 % of the surface of the Earth (Dise 2009), they are believed to store between 500 and 700 Gt of carbon (Yu 2011, Page & Baird 2016, Loisel *et al.* 2017). This is of a similar order of magnitude to the 800+ Gt of carbon in the atmosphere (Batjes 1996, IPCC 2014). Northern peatlands are globally the most important stores of carbon, and are distributed primarily across Russia, North America, Fennoscandia, Eastern Europe and the British Isles (Mitsch & Gosselink 2015). These northern peatlands are estimated to contain approximately 90 % of the total peatland carbon pool (Yu *et al.* 2010). While recent exploration of peat deposits in Congo have increased the estimated extent of tropical peat deposits by as much as a third (Dargie *et al.* 2017), such deposits are dwarfed in scale by those of northern peatlands, which by some estimates may be even larger than previously thought (Nichols & Peteet 2019). Accumulation of this peat has provided a small negative feedback to climate (a net climatic cooling effect) over the last 1000 years (Charman *et al.* 2013), with an estimated net sink of carbon of 44 Gt ka^{-1} (Yu 2011). Peatlands also influence the climate system as a significant source of methane (CH_4), carbon dioxide (CO_2), aquatic carbon and to a less significant extent other greenhouse gases (N_2O, VOCs), and have a direct effect on radiation balance through albedo.

Development of the forest industry in peatlands

Peatlands have historically been viewed as barren and unproductive places, but in reality support many economic activities and provide often unnoticed ecosystem services. Economically, peatlands have important roles for agriculture (in particular grazing), water management, and leisure activities such as shooting and tourism (Whitfield *et al.* 2011). Ecosystem services include water and carbon storage (Joosten *et al.* 2012), and maintenance of biodiversity including specialised peatland species (Stroud *et al.* 1987, Lindsay *et al.* 1988, Littlewood *et al.* 2010).

It is estimated that around 20 % of European peatlands are currently drained for forestry (Drosler *et al.* 2008). Many peatlands, especially those in tropical and boreal regions, have natural tree cover and may be categorised as 'forest'. Other peatlands, for example many within the Arctic and temperate zone, are naturally treeless. In these landscapes, mixed wet scrub and low wet woodland are restricted to peat bog margins and along the courses of streams (Lindsay 2010). One such area is the United Kingdom, where an estimated 2300 Mt of carbon is stored in peatlands (Billett *et al.* 2010), of which blanket bog is the predominant type. While trees appearing naturally on ombrotrophic bog peat may have been more common in the past, today almost all UK bogs are open. This changed between the 1950s and the 1980s, when approximately 9 % (190,000 ha) of the UK's deep peats were drained for forestry (Hargreaves *et al.* 2003), although this figure may be an underestimate and may be as high as 17 % in Scotland (Vanguelova *et al.* 2018).

Early forestry trials in the UK

Numerous attempts at peatland afforestation have been made in the UK since the 18th century. For instance, in his history of the county of Peeblesshire, William Chambers (1864) records a drainage initiative by the Duke of Argyll in 1730, in which he made an "attempt to make a quagmire not only into a dry and arable land, but fitted by its amenity for the residence of a man of taste". This included an early and largely ineffective attempt at drain cutting, with trees being planted on any sufficiently dry areas. After poor results and the death of the Duke in 1761, the plan was abandoned (W. Chambers 1864). Such schemes, driven generally by individuals or individual estates, are typical of the small-scale and uncoordinated efforts common at the time.

Foresters in Britain were slow to take note of developments in continental Europe. Belgian reforestation of denigrated peatlands has begun in the late 18th century (Petit & Lambin 2002). By 1836, foresters in Belgium had developed a system of turf planting in which some of the peat was removed, upturned and laid over the remaining surface to give a deeper, drier substrate on which to plant, combining with intensive drainage to yield the first significant successes in planting forests on peat, a system not widely adopted in Britain until around 1907 (Zehetmayer 1954).

UK peatland afforestation in the 20th century

A critical moment in the history of British peatland forestry was the establishment of the Forestry Commission, a government body with responsibility for managing forestry. The Forestry Commission was founded under the Forestry Act of 1919, with a remit to increase forest coverage and timber production. As well as aiming to develop an economic resource, this was in part a response to concerns about depleted woodland stocks following the First World War, as a domestic supply of wooden pit props to support the mining industry was strategically important (Marren 2002). The establishment of the Forestry Commission led to a more coordinated and efficient approach to forestry. Broadly this was achieved through the state purchase of large tracts of non-woodland areas, followed by a systematic and centrally organised program of planting, and finally an ambitious initial target of planting 1.7 million acres of forest (Robertson 1943). It is a measure of the rapid expansion and success of these measures that within twenty years it had achieved over 80 % of its initial targets (Wightman 2015).

Expansion of forestry into the uplands during the inter-war years occurred mainly across organo-mineral soils. Deeper peat was considered too challenging for silviculture and unsuitable for the machinery then in use. It was not until after the Second World War that development and modification of the double mouldboard plough combined with efficient tractors with wide tracks allowed the Forestry Commission to commence more widespread trials on deeper peats (Wood 1974, Anderson 1997). The double mouldboard plough pushed cut peat into a ridge on either side of a drainage furrow, creating raised dry ridges typically two metres apart on which trees could be planted (Figure 2.1). This closely-spaced furrow ploughing was combined with collector drainage ditches at intervals to provide a sufficiently dry environment for tree growth (Harrison *et al.* 1994) and was supplemented by fertiliser application to overcome the paucity of nutrients, particularly phosphorous but also potassium, nitrogen and trace elements (Taylor 1991).

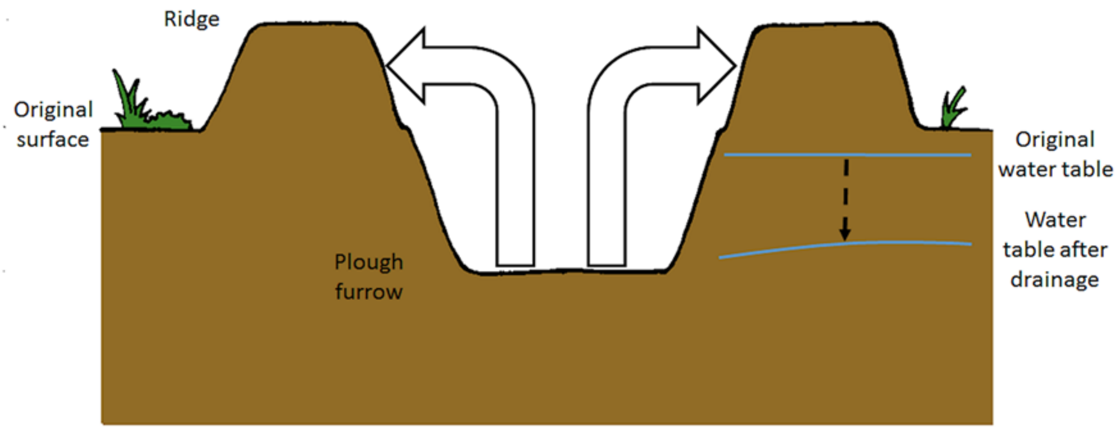

Figure 2.1. A double mould board plough creates a furrow in the peat and pushes excavated peat into ridges. These ridges are sufficiently raised above the drained water table that the survival chance of planted trees is substantially increased. Unlike the low tillage on Fennoscandian sites, such ploughing on UK forestry sites disturbs the peat and removes much of the bog vegetation.

Norway spruce (*Picea abies*), Scots pine (*Pinus sylvestris*), mountain pine (*Pinus mugo*) and species of larch (*Latrix decidua*, *Latrix kaempferi*) had been trialled for peatland afforestation in the UK by the early 20th century, but with limited success (Zehetmayer 1954). Ultimately, forestry in UK peat bogs became feasible with the adoption into European silviculture of trees native to North America, particularly some varieties of Lodgepole pine (*Pinus contorta*) and Sitka spruce (*Picea sitchensis*). 'South Coastal' varieties of Lodgepole pine were initially seen as good candidates for afforesting large areas of peat (Pyatt 1990). This species tolerates high water tables by creating gas pockets within the pericycle of the roots that allow continued oxygenation in waterlogged conditions by diffusion from the air (King *et al.* 1986). Consequently, it roots deeply, drying the peat. However, problems with curvature of the base of the trunk ('basal sweep'), low wood quality and occasional devastating outbreaks of Pine Beauty Moth (*Panolis flammea*) meant that Lodgepole pine was ultimately abandoned as a commercial crop.

Sitka spruce was introduced into the UK as an ornamental species in the late 1820s. Due to its rapid growth and excellent quality wood, it was adopted as a commercial crop in the early 20th century (Oosthoek 2013). Sitka spruce is a valuable timber-producing species and is the most widely grown conifer in the UK, covering 682,100 ha (50.3 % of the total conifer stock) (Forestry Commission 2011). This species alone accounts for 33 % of the total woodland coverage in the Highlands of Scotland (Smith & Gilbert 2003). Sitka spruce grows poorly in waterlogged conditions, so in peat bogs in many parts of the UK it was mostly planted in mixed stands with Lodgepole pine, which acted as a 'nurse species' (Pyatt 1993). It was hoped that the relative vigour of Lodgepole pine in wet conditions, and consequent water interception due to canopy closure, would in turn increase yields of other species during the first rotation (King *et al.* 1986). Ultimately, while Lodgepole pine was shown to have a drying effect on the peat, this did not always translate into an improvement in growth of the Sitka spruce (Ray & Schweizer 1994). For this reason the economic benefits of mixed planting were questioned and Sitka spruce monocultures became increasingly common as more stands were planted (Oosthoek 2013).

Sites across the UK were drained and planted by the Forestry Commission in the second half of the 20th century. At this time, forest planting was an industrial-scale operation involving extensive landscape change beyond simply planting trees including construction of roads, bridges and fences, and quarrying for building materials.

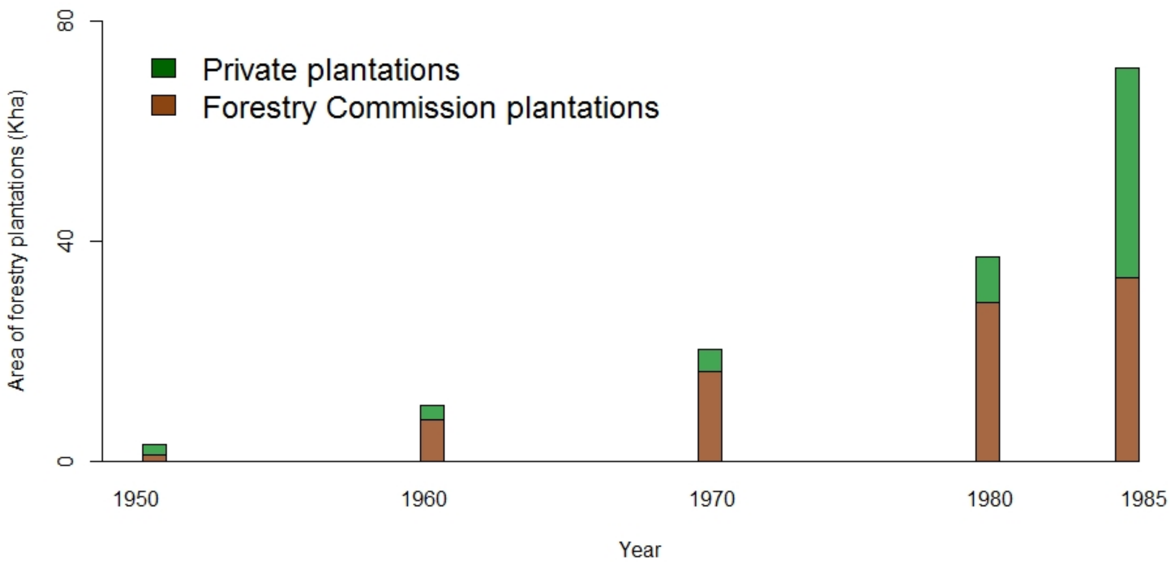

Figure 2.2. Extent and ownership of forest plantations in Caithness and Sutherland between 1950 and 1985. Adapted from Stroud *et al*. (2015).

The technological developments which permitted peat bog afforestation coincided with a tax and grant regime favourable to forest development in unsuitable areas. Government incentives proved popular as a mechanism for reducing tax liability (Mather 1986, Mather & Murray 1988). All expenses related to forestry were tax deductible, with loans available which could also be written off against tax. Companies such as Fountain Forestry managed large areas of land for wealthy individuals. Through the 1970s private planting overtook planting by the Forestry Commission (Figure 2.2), much of it concentrated in Scotland (Mather & Murray 1988). Tree growth was frequently poor and a large proportion of the forests planted during this period would not have been economically viable without tax relief.

Public and scientific reaction to afforestation

From the late 1960s the issue of peatland afforestation grew in prominence, with concerns raised over the loss of biodiversity and the risk of eutrophication of freshwaters and damage to fisheries (Moore & Bellamy 1974, Thompson 1987). Public awareness of the large-scale planting of the uplands and the economic factors underpinning it was raised with the revelation that well-known figures such as TV presenter Terry Wogan, singer Cliff Richard and snooker player Alex 'Hurricane' Higgins were using forestry-based tax avoidance schemes (Rosie 1986, Anon. 1995).

Between 1987 and 1988, the Nature Conservancy Council - the UK government statutory advisor on wildlife conservation matters at the time - published 'Birds, Bogs and Forestry' and 'The Flow Country - the Peatlands of Caithness and Sutherland', a linked pair of reports on the biodiversity of the Flow Country and the scale of forestry expansion (Stroud *et al.* 1987, Lindsay *et al.* 1988). The Flow Country is the UK's most extensive peatland region with over 400,000 ha of peat and wetland, of which around 67,000 ha (approximately 17 %) had by then been afforested. The reports highlighted the potential disruption that could be caused by forestry and, while the first report generated extensive political controversy, the detailed figures provided in the second report led the Secretary of State for Scotland to afford statutory protection to almost 200,000 ha of un-afforested peatland in the Flow Country as a composite Site of Special Scientific Interest (SSSI), the largest such site in the UK. The fallout from this controversial action is widely believed to have contributed to the subsequent decision of the government of the time to break up the Nature Conservancy Council (Warren 2000). Later, the SSSI was designated as the UK's largest terrestrial Special Area for Conservation (SAC) and Special Protection Area (SPA) within the European Commission's 'Natura 2000' nature protection network.

Controversy over tax avoidance in general, but particularly the schemes set up for forestry, led to legislative changes. With public outcry increasing, the then Chancellor of the Exchequer Nigel Lawson ended the tax breaks in his budget of 1988 (Oosthoek 2013). With the main financial incentive removed, new peat bog forestry planting has been limited since 1990 (Stroud *et al.* 2015) and was effectively halted by later Forestry Commission policy guidance (Patterson & Anderson 2000).

Should afforested deep peat be restocked or restored?

Following the intensive afforestation of the twentieth century around 9 % of the UK's deep peats, amounting to a total of approximately 190,000 ha, were drained for forestry (Hargreaves *et al.* 2003). This forestry is distributed across the UK but is particularly extensive in Scotland. Many plantations are approaching harvesting age and decisions must soon be taken on whether to restock the forests or restore drained bogs as far as possible to their previous state. While there is some debate as to whether restoration should aim to recreate a pre-drainage or pre-afforestation state, the process typically involves the removal of trees and blocking of drainage to raise and stabilise water tables and restore active peatland habitats.

While not the only factor (biodiversity considerations are especially important; Holden *et al.* 2007), the effect of afforestation on carbon stock and carbon cycling is an important issue in this decision-making given likely consequences for climate change, and has been acknowledged as such by Forestry Commission in guidance from 2000 onwards (Patterson & Anderson 2000). While peatland restoration was not originally specified by the Kyoto Protocol (beyond a general call for the protection of natural carbon stocks and sinks) or used as a mitigating factor in subsequent calculations of carbon emissions, the Protocol was ultimately amended to allow peatland rewetting to be considered in carbon accounting (Bain *et al.* 2012). Restoration of peatlands is recommended by several international bodies (Joosten *et al.* 2012) including, most recently, the International Union for the Conservation of Nature (Resolution 043; IUCN 2016).

In Scotland, the devolved government aims to restore 40 % of the estimated 600,000 ha of damaged peatlands by 2030 (Scottish Government 2017), which includes restoration of afforested peat bogs (Scottish Natural Heritage 2015). Generally, there is a presumption that any felled woodlands will be restocked, but allowances are made in the Scottish Government's Policy on the Control of Woodland Removal for not replanting on peatland sites that are a priority for restoration on ecological grounds, and on those peatlands that are not a priority for restoration when there would be a significant greenhouse gas benefit to restoring degraded peat (Forestry Commission Scotland 2009). Published guidance from the Forestry Commission

(Forestry Commission Scotland 2015, 2016) provides a decision framework for such restocking decisions, but the underpinning evidence is limited in some important areas. Therefore, the question of what effects the drainage and planting have had on peat bogs, and the likely effects of restoration, are issues of critical importance. There are extensive gaps in current knowledge that need to be filled. This article considers the likely effects of forestry on the peatland, and the applicability of currently available research data to the unique circumstances in which UK peatlands were afforested.

What effects has tree planting had on radiative forcing?

The climatic consequences of afforestation represent the net effect of several interacting processes on the peat bog ecosystem and wider supply-chain considerations. Changes to peatlands encompass physical changes to the peat itself, vegetation changes, changes to carbon sequestration, effluxes of carbon in gaseous and aquatic forms, and other more minor factors which may nevertheless contribute to the overall radiative forcing, such as albedo. This section reviews these processes.

Drainage and planting effects on carbon accumulation in peat

Undrained peatlands accumulate carbon through primary production, as plants (often non-vascular species such as *Sphagnum*) photosynthesise. Within an undrained natural bog, carbon sequestered in this way remains within the peat over long timescales (millennia) because dead material will not fully decay within the main body of peat (the catotelm). Approximately 50 % of dry peat mass is carbon (Chambers *et al.* 2011), with older peat compressed at the base of a profile having a higher density and therefore storing more carbon by fresh volume (Clymo *et al.* 1998). Drainage and the process of ploughing disrupts the existing vegetation, affecting the amount of carbon sequestered directly to the bog by the living layer (the acrotelm) (Figure 2.3). Afforestation essentially halts primary production by typical peat-forming bog species, so the ultimate capacity for radiative forcing then largely depends on the fate of carbon sequestered by trees and by the response of the peat stored in the catotelm.

Peat in a natural bog is divided between the aerated acrotelm and the deeper, constantly waterlogged, catotelm (Ingram 1978, 1983). The boundary between these two layers is the deepest point to which the water table falls under normal conditions. Undrained peat bogs typically have a high water table, commonly within 10–20 cm of the surface of the peat, but this is substantially lowered with afforestation. Lowering the water table through drainage is arguably the most important factor for successful afforestation, providing the aeration that is essential for growth of the roots of most tree species (Braekke 1983), changing the physical and chemical properties of the peat, and affecting hydrology (Braekke 1987, Holden 2004). Planted forests lower the water table further when canopy closure leads to increased interception and evapotranspiration (Sarkkola *et al.* 2010).

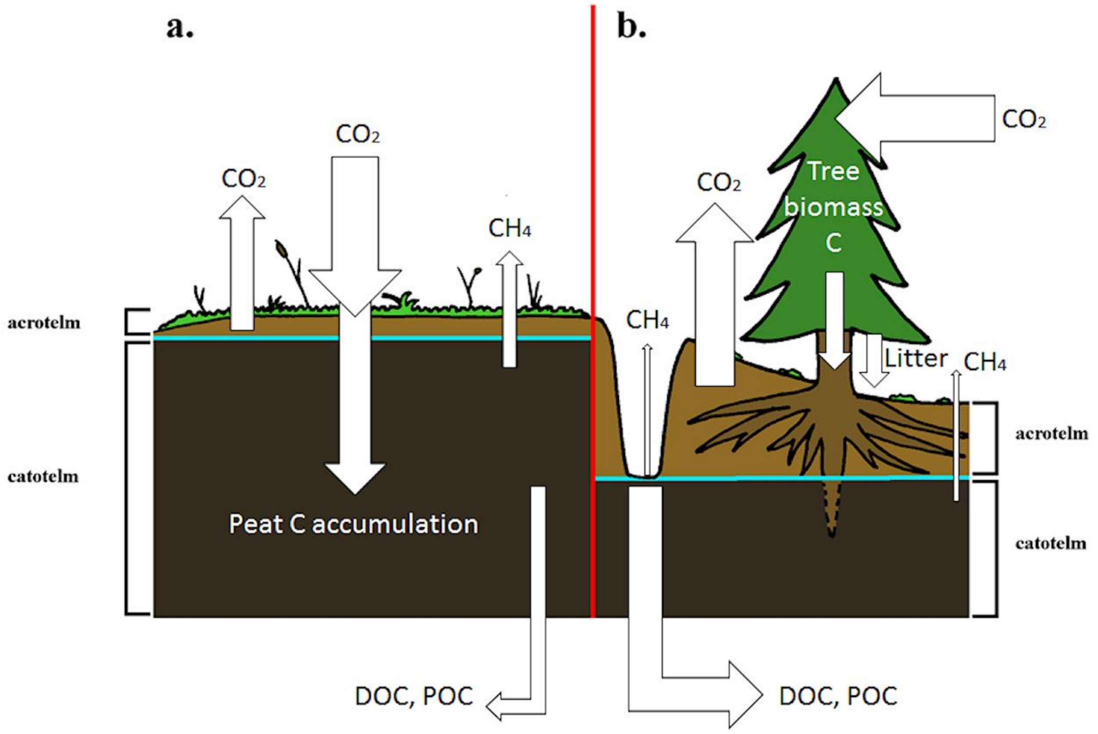

Figure 2.3. A lowered water table (blue line) gives rise to different rates of carbon loss and accumulation in peatland systems (a) before drainage and (b) after drainage and afforestation, which also cause subsidence. Carbon dioxide (CO_2) production will increase with aeration of the upper layer of peat, with a reduction of methane (CH_4) production from waterlogged peat. Loss of aquatic dissolved and particulate carbon (DOC and POC) may be increased through drainage. Carbon is taken up by vegetation in both scenarios. In an undrained bog some of this will go on to be stored in peat over long timescales, whereas in the drained system it will form tree biomass, eventually reaching the soil as litter and roots or being removed from the site as harvested timber. Unlike Sitka spruce, lodgepole pine is tolerant of waterlogging, and its roots can extend below the water table. The peat beneath the tree crop will have increased dry bulk density compared with the non-afforested peat bog. Flux magnitudes indicated by arrow widths are indicative and open to varying and different degrees of uncertainty and to variation with site conditions.

Drainage of a peat soil gives rise to three important processes: primary consolidation, secondary compression and oxidative loss (or peat 'wastage', discussed below). Primary consolidation occurs rapidly following drainage and is caused by loss of water from large pore spaces within the peat. Secondary compression occurs because more tightly-bound water is slowly squeezed from the peat matrix by the weight of peat material no longer supported by the bog water. In addition, the peat may be further compacted by the weight of growing trees (Hobbs 1986). These various processes cause subsidence of the ground surface and ultimately cracking of the upper peat, which can lead to deeper aeration (Pyatt & John 1989, Pyatt *et al.* 1992). Furthermore, any clearing or re-grading of the drainage system will stimulate a new round of primary consolidation before the slower, steady processes of secondary compression and oxidative loss resume (Wold 1976).

The horizontal 'zone of impact' associated with forest blocks on adjacent peatlands has yet to be determined for carbon, hydrology and bog vegetation, though effects on peatland birds are well-established (Wilson *et al.* 2014). There has been only limited monitoring of long-term changes in surface morphology, vegetation assemblages, hydrology and peatland microtopography, meaning that current estimates are based largely on relatively short-term studies, often of hydrology (generally water table depth and hydraulic connectivity). These estimates currently range from 2–3 metres up to 50–60 metres, but some hydrological models suggest that drainage effects may extend for several hundred metres in some circumstances (Holden 2005).

The net increase in radiative forcing caused by the effect of physical changes in the peat on carbon storage may be added to by the direct radiative effect through changed surface albedo of forest plantations. Trees can affect snow cover and where trees are felled, the surface environment can have a very high albedo leading to a cooling effect (Lohila *et al.* 2010). Such effects are rarely considered but may be significant.

Greenhouse gases

The depth of the water table below the ground surface is a key driver of greenhouse gas (GHG) balance (typically measured as parts per million equivalent carbon dioxide; ppmv CO_2eq), as this determines the volume of peat exposed to aeration and consequently microbial production of both CO_2 and CH_4 (Drosler *et al.* 2008). Lowering the water table during afforestation has the potential to significantly affect the fluxes of both of these GHGs from peat to the atmosphere (Figure 2.3).

In the permanently waterlogged catotelm, bacterial decomposition is inhibited by low temperature, pH, and oxygen availability (Freeman *et al.* 2001b). In these anoxic conditions CH_4 is an end product of anaerobic decomposition through several pathways (Lai 2009). As it moves up through the acrotelm a large proportion of this CH_4 is oxidised by methanotrophic bacteria (this varies greatly depending on the type and acidity of peat, but a lowering of the water table from the surface has been shown to reduce methane flux by 90 – 100 %). Lowering the water table in peatland afforestation increases the depth of air penetration into the normally-waterlogged catotelm peat and thereby the space in which CH_4 can be oxidised, and thus typically leads to a linear decrease in CH_4 efflux (Moore and Knowles 1989).

Simultaneous with the reduction in CH_4 efflux, lowering the water table with peatland afforestation leads to increased efflux of CO_2 through oxidative loss, or peat 'wastage'. Drainage enables oxygen to penetrate into (what had previously been) catotelm peat, exposing the long-term carbon store to oxidative decomposition by bacteria and fungi, leading to increased production of CO_2 (Eggelsmann 1975, Hobbs 1986). The increase in CO_2 flux with drainage is highly variable based on a range of factors, making a single estimate of this number impossible to make, however a doubling of CO_2 emissions from a well drained site would not be unusual (Silvola *et al.* 1996). The loss of the unique structure and function of the aerated acrotelm may lead to the bog becoming a single-layered haplotelmic bog (Ingram & Bragg 1984).

Peatland drainage is, therefore, likely to have opposing effects on these two GHGs, increasing CO_2 and reducing CH_4 effluxes. While more carbon is lost to the environment as CO_2, CH_4 has a global warming potential over 100 yr (GWP100) 34 times greater than CO_2 when climate-carbon feedbacks are considered (IPCC 2013). In terms of fluxes from peat, it is likely that the CO_2 increase outweighs the CH_4 decrease and the net effect is to promote climate warming (Martikainen *et al.* 1995, Alm *et al.* 1999).

CO_2 and CH_4 are the most important GHGs arising from peatlands, but nitrous oxide (N_2O) may also be significant in some situations. The GWP100 of N_2O is 298 times that of CO_2 when climate-carbon feedbacks are considered (IPCC 2013). Fluxes of N_2O in peatlands are typically small but can become substantial in fens or when peatlands are exposed to N in fertiliser, as in some peatland afforestation. However there are few studies which directly consider the effect of afforestation on N_2O flux (Maljanen *et al.* 2010).

Beyond the direct effect of afforestation on the carbon balance of peat there are other factors which may also result in GHG production. Emissions from vehicles and machinery, as well as road and steel fence construction, also have significant GHG implications for the initial ploughing, planting, interim management and final harvesting of any forestry site (Morison *et al.* 2012).

Aquatic carbon

Aquatic carbon is exported from peatlands via watercourses, principally as dissolved organic carbon (DOC) and particulate organic carbon (POC). Both of these fluxes may be affected by afforestation. DOC concentration in streams correlates positively with the presence of organic soils and peats in a catchment (Hope *et al.* 1997, Aitkenhead *et al.* 1999). Higher outflow of water either drained from the system or lost through consolidation and compression carries with it more aquatic carbon. This process will continue slowly but indefinitely in a drained system. DOC may enter the atmosphere downstream through other degradative pathways, usually through rapid emission as CO_2, and may be a significant GHG source in upland areas (Freeman *et al.* 2001a). The pathways of POC to the atmosphere are less certain (Rowson *et al.* 2010), although there is increasing evidence that such carbon will ultimately be released to the atmosphere as CO_2 (Evans *et al.* 2016).

Disruption caused by on-site activity such as ploughing, tree planting and the continuing maintenance of drains is also associated with increased concentrations of DOC and POC in streams draining the forest stand. Later, disruption to the peat surface caused by tree thinning or felling can lead to further aquatic carbon loss for several years after the trees are removed (Cummins & Farrell 2003). This loss of carbon through aquatic pathways may depend on variables including nutrient content of the peat (Nieminen *et al.* 2015), catchment properties (Holden 2005) and weather patterns (Koehler *et al.* 2009).

Carbon accumulation in tree biomass

Any loss of carbon from peat soils may be offset by gains of carbon stored in tree biomass, litter and new soil organic matter. The true carbon balance then depends partly on the fate of the wood produced (Minkkinen *et al.* 2002). The quality and longevity of the wood products that arise from forestry will determine whether or not the harvested portion of the carbon captured by the trees is sequestered over long timescales (Laine *et al.* 1992, Ojanen *et al.* 2013). In areas with high yield and high-quality wood this timber may be used for long-lifespan uses such as construction, effectively storing the carbon for many decades or even centuries. However, forestry crops on bogs in the UK are often of such poor quality that much of the wood goes for pulp, fuel and other low-grade uses, returning carbon to the atmosphere much more quickly (Thompson & Matthews 1989, Artz *et al.* 2013). The portion of the carbon captured by the trees that is left below ground when they are felled consists of roots, litter (root, needle, branch, etc.) and soil organic matter derived from these. In addition, the stumps, branches and top parts of the stems are normally left on the ground after harvesting. The fate of these below-ground and surface components containing tree-derived carbon also influences the true carbon balance (Vanguelova *et al.* 2017). The true climate consequences of peatland forestry are further complicated by the role of the wood produced in the supply chain and the potential for timber to replace alternative materials with high carbon footprints such as plastics and concrete.

The wetness of naturally treeless British bogs may contribute to an increase in trees lost to wind-throw (Figure 2.4). Many peat bogs used for forestry remain wet even after drainage, leading to the development of shallow and often uni-directional root plates confined by cracks beneath the ploughing furrows (Lindsay & Bragg 2004). This, combined with the very windy climate of many UK peatland forest regions, makes trees more prone to toppling (Ray & Nicoll 1998). Wind-throw will reduce timber yields, and may force earlier harvesting (Gardiner & Quine 2000), reducing the quantity and quality of wood products and so reducing the residence time of carbon in the tree biomass. Lodgepole pine is especially prone to wind-throw (Nicoll *et al.* 2006).

Figure 2.4. A wind-thrown Lodgepole pine with exposed root plate at Bad a'Cheo, Rumster Forest, Caithness.

Approaches to measuring carbon loss from peatlands

From the above discussion, it will be clear that afforestation can affect the peatland carbon budget and radiative forcing more generally through many mechanisms. Studies have taken several approaches to quantifying these effects (Table 2.1). Many studies attempt to assess peat carbon balance by directly measuring the key fluxes of GHGs and aquatic carbon (although aqueous carbon is considered less often in the literature). Methods such as cover boxes ('chambers') or eddy covariance towers use infra-red gas analysis (IRGA) to measure GHG fluxes in real time in the field, replacing older methods using gas sampling for chromatography or quadrupole mass spectrometry (QMS), or recording weight change in soda lime. Using these methods to understand the way in which carbon is imported to or exported from peatlands can help to understand processes over short timescales. Typically, forestry on bogs requires a programme of site drainage followed by over forty years of tree growth. As a result, short-term studies of carbon fluxes in the system may not accurately describe the carbon change in the system over longer timescales. This is important as GHG emissions can be highly variable over time (Klemedtsson *et al.* 2008).

Table 2.1. Methods used for determining carbon budgets in peatlands.

Assessment type	Methods	Timescale	Advantages	Disadvantages	Example paper
Carbon flux	Cover box (GHG)	Usually between days and months	Continuous, accurate data	High cost equipment, flux from trees not measured accurately	(Hermans *et al.* 2019)
	Eddy covariance tower (GHG)		Continuous, accurate data	High cost equipment, ground level processes missed	(Schrier-Uijl *et al.* 2010)
	Gas sampling, gas chromatography (GHG)		Accurate data	Data not continuous, analysis can be expensive	(Pihlatie *et al.* 2010)
	Soda lime measurement (GHG)		Low cost	Only measures CO_2, inaccurate, prone to underestimates	(Byrne & Farrell 2005)
	Water sampling, elemental analysis (DOC/POC)		Accurate	Data not continuous, high cost	(Webb *et al.* 2019)
Carbon stock	Coring, bulk density, carbon analysis	The whole age of the peat	Provides complete picture of carbon loss or gain, no long-term monitoring	No information about fine scale processes, only total carbon, reliable stratigraphic markers required	(Ratcliffe *et al.* 2017)
	Optical or satellite surveys of subsidence	Decades, depending on age of original records	Low cost, quick	Subsidence an unreliable proxy for carbon loss, original surveys may be of poor quality	(Lees *et al.* 2018)

Another approach to measuring changes in soil carbon is to use a whole-column inventory of the carbon stock in the peat (Pitkanen *et al.* 2013). This typically involves coring a column of peat, then determining the carbon content through the measurement of dry bulk density followed by either direct elemental analysis or deriving a value from the amount of organic material lost on ignition and an assumption of about 50 % as the proportion of carbon in the organic matter (Chambers *et al.* 2011). Such carbon analysis allows an assessment of the net exchange of carbon with the environment over long timescales, although this does not identify the specific gas and aqueous components. The use of stratigraphic markers in the peat allows the age of a sample to be identified (Pitkanen *et al.* 2013), meaning that direct comparisons can be made between peat of the same age. This analysis can be paired with analysis of carbon in the trees to determine net balance. Laiho and Pearson (2016) highlight a number of issues which must, nevertheless, be considered when using such an approach.

A less exact method of determining loss of carbon stock on sites that have historical ground level surveys is to use subsidence as a proxy for loss of carbon. While this method is relatively low-cost where historical records of ground levels exist, subsidence is an unreliable indicator of carbon loss as it is often based on initial surveys which can be decades old and of poor quality, with estimates produced in this way "roughly determined" at best (Hommeltenberg *et al.* 2014). In addition, it ignores the compaction and compression that usually occurs.

Carbon flux and stock measurements are both useful tools in determining the effect of management decisions on the carbon held in peat bogs. Each set of techniques have advantages and weaknesses and will provide different types of data (Table 2.1). Modern flux techniques (other than the now less common soda lime method) allow for accurate continuous recording of gaseous GHG movement in and out of peat, which can now be easily watched and logged in real time in the field (Sterk *et al.* 2019), as well as incremental measures of aqueous carbon. This provides key data on the fine scale pathways through which carbon is lost. Such techniques are especially useful when monitoring carbon flux in the immediate aftermath of a change in land management (Ward *et al.* 2007). Over longer time periods, or where there was no baseline data from before a management intervention for comparison, flux data can be less useful. Conversely, stock based approached do little to reveal the precise pathways through which carbon is gained or lost from a system. Such data does however provide a net assessment of how total carbon stock has changed over longer periods and can provide details on broad accumulation rates (Ratcliffe *et al.* 2017, Marrs *et al.* 2019).

As with any form of field science, these techniques have different practical considerations and difficulties associated with them. As each series of flux measurements provides a data set for a short period of time, multiple sampling campaigns are usually required to obtain a representative data set, especially as seasonality is so important to water tables and carbon flux in peatlands (Holden 2005). This is less of a consideration for carbon stock techniques using coring (although optical and satellite surveys may need to be performed more than once if possible due to 'bog breathing'). In addition, several flux data sets (for different gasses or sources of aqueous carbon) are usually required to produce a total budget (Worrall *et al.* 2003), although this has the advantage of also accounting for other non-carbon greenhouse gasses such as N2O (Wilson *et al.* 2016) which will not typically be included in carbon stock assessments. In addition, the cost of analytical equipment for flux measurements may be high, while stock approaches can usually be performed with relatively basic equipment (Aaby & Digerfeldt 1986), although accuracy is greatly improved if more sophisticated elemental analysis techniques to determine carbon content replaces simple loss on ignition. Core samples will however often need additional supporting data, for example stratigraphic markers such radiocarbon dates (Piotrowska *et al.* 2011) or tephra (Davies 2015), which can be an expensive or time consuming to process.

The data produced by each type of technique also have separate caveats and limitations. The repeated visits to field sites required for flux sampling may lead to the disturbance of bogs, and so such study sites may require the addition of extra semi-permeant infrastructure such as duck boards. Such disruption is less of a concern in stock sampling, although coring does involve the removal and destructive sampling of peat, which may raise issues when working in protected areas. There is also a growing awareness in the literature that some complicating factors with these methods exist and should be more widely considered when planning studies. For example, in flux-based studies the growth of vegetation and changing species assemblages may significantly alter the internal volume of cover boxes over the duration of a study (Morton & Heinemeyer 2018). In measuring aqueous carbon fluxes, water samples may degrade over time, so storage is a further consideration (Cook *et al.* 2016). In stock-based approaches there is now an acknowledgement of the unreliability of carbon accumulation calculations that include younger acrotelm peat (Young *et al.* 2019). Each of these issues must be factored in when adopting these approaches.

The choice of technique to use to determine carbon budgets in peatlands depends on the questions being asked, and the conditions at a given field site. Where long term monitoring is possible, where baseline data can be gathered before a management change, and where a process level understanding of the fate of carbon is required, flux studies may be more appropriate. Where land use change occurred prior to the beginning of a study, where long term monitoring is not possible, or where a figure for an overall net change in carbon is sought, stock-based approaches should be considered.

A review of the available research

Current understanding of the impact of peatland forestry in Fennoscandia

Much of the work on the effects of peatland afforestation on carbon storage and GHG production has been carried out in Fennoscandia. Forestry is particularly widespread on drained peat in Finland, with up to 25 % of the nation's commercial forests growing on peat (Laiho &

Laine 1997). Across Fennoscandia, drained and afforested peat occupies approximately 5,700,000 ha in Finland, 5,000,000 ha in Sweden, 420,000 ha in Norway, with additional smaller plantations in Russia and Iceland (Kløve *et al.* 2017). There is a broad body of work over several decades describing effects of this forestry on carbon storage.

Initially, drainage was predicted to lead to a loss of stored carbon, through the mechanisms described earlier in this paper (Armentano & Menges 1986). However early studies in the 1990s and 2000s demonstrated that afforestation is generally less detrimental than drainage for agricultural use (Nykänen *et al.* 1995). Indeed, the afforestation of a cultivated or cut peatland is often seen as an ideal way to reduce the overall GWP of an area of degraded peat (Alm *et al.* 2007).

That the balance of evidence is that peat afforestation is carbon neutral or associated with a net gain of carbon is attributable to the carbon sequestered by tree biomass in drained areas exceeding the loss of carbon from peat (Sakovets & Germanova 1992). Net ecosystem productivity in afforested peat has been projected to remain positive for 300 years, largely due to increased carbon storage in above ground biomass (Laiho & Laine 1997). A typical study shows that the carbon accumulation rate at an undrained site over 30 years was 21 g C m^{-2} yr^{-1}, against a loss of 14 g C m^{-2} yr^{-1} in drained sites, but that this loss was less than the carbon accumulated in tree biomass (Laine & Minkkinen 1996).

Other studies have suggested that nutrient poor peats may even continue to act as a CO_2 sink after drainage (Ojanen *et al.* 2013), which is unexpected given the relationship between drainage and oxidation of peat. This is explained by the relatively limited amount of drainage (around 30 cm was typical; Braekke 1983) usually required to convert naturally forested Fennoscandian peats into commercial wood production. A classic study by Lohila *et al.* (2001) illustrates this well: on a dwarf-shrub pine bog that had been ditched for 35 years (only producing a relatively minor drop in the water table), eddy covariance measures showed an accumulation of 870 g CO_2 m^{-2} yr^{-1}. As the site was also a net sink for CH_4, the overall annual peat accumulation of 240 g C m^{-2} was higher even than the 175 g C m^{-2} accumulated in tree biomass (Lohila *et al.* 2011). The depth of drainage may therefore be a critical factor: increasing drainage will lead to higher rates of carbon release from peat, and increased vegetation net primary productivity is less likely to compensate for this if the water table is lowered by more than 30 cm (Silvola *et al.* 1996). Conversely it is possible that if a site is poorly drained enough to still sequester CO_2 they may also remain net sources of CH_4 (Ojanen *et al.* 2010). Indeed forest volume (usually dependent on drainage) is strongly correlated with a reduction in CH_4 efflux (Minkkinen et al. 2007b).

Perhaps the most comprehensive review of the evidence is by Maljanen *et al.* (2010), who synthesise around 100 Fennoscandian flux studies to suggest that afforested peatlands remain a net sink for carbon (a mean sink of 790 g CO_2eq m^{-2} yr^{-1}). This is variable dependent on hydrology, with afforested peat switching to net sources of carbon in dry years. Similarly, a wider review of European peatland emissions by Drosler *et al.* (2008) found that the balance of evidence was that afforested peatlands were greenhouse gas neutral, or sequestered carbon if wood biomass was included.

While these annual figures are useful, they are highly variable through a whole forestry rotation. It is therefore valuable to consider the net change in carbon storage over the lifetime of plantations. Such studies have produced contrasting conclusions about whether there has been a gain or loss of carbon from peat, but tend to agree on a net carbon accumulation when carbon sequestered in tree biomass is included. Two contrasting studies of long term accumulation in Finnish peat (of which 55 – 60 % was drained for forestry) illustrate this well. Minkkinen *at al.* (2002) estimate that between 1900 and 2000 an additional 50 Mt C was stored in Finnish peat (due largely to a decrease in CH_4 emissions combined with a lower than expected increase in CO_2 production), with an additional approximate 100 Mt C stored in tree biomass. Conversely Turunen (2008) finds that between 1950 and 2000 the carbon stored in peat in Finland may have decreased by as much as 73 Mt C. However, when the increase in carbon sequestered in vegetation biomass is included, peatlands have had a net gain of 52 Mt C.

Within the last decade, long term studies have led to some reappraisal of the effects of peatland forestry. Measuring carbon stock through whole column inventories have shown losses of surface peat as a result of drainage (Pitkanen *et al.* 2013), and consequently higher losses of carbon than would have been expected from flux studies. In a resurvey of peatland sites first monitored in the 1980s as part of initial work into afforested peatland carbon, coring peat profiles estimated a loss of 150 g C m^{-2} y^{-1} from peat drained for forestry in Finland, a larger estimate than is conventionally given, and one that would equate to average annual efflux of 31 Mt CO_2 yr^{-1} (Simola *et al.* 2012). This carbon loss is considerably larger than in earlier calculations estimated on the basis of flux studies alone.

Questions have been raised about the methodological approaches used in earlier work, and these may account for some of the differences. Chamber methods used for CO_2 flux studies have been shown to be associated with high levels of uncertainty (Ojanen *et al.* 2014). There is also an increasing awareness that emissions may be highly variable over time based on seasonal controls of drainage and temperature (Klemedtsson *et al.* 2008). Over winter, forestry drained bog releases CO_2, while well drained fens are a net sink (Alm *et al.* 1999). Therefore, high time resolution or continuous measurements are needed to capture peak emissions, as there will be a CO_2 pulse after thaws (Pihlatie *et al.* 2010). Greenhouse gas emissions can also vary spatially in a plantation, for example in drainage ditches which occupy a relatively small area but may produce efflux of CH_4 higher than found by measuring the drained peat (Minkkinen *et al.* 1997, Minkkinen & Laine 2006). In addition, the high decomposition rates of litter (95% over 30 years) suggests that assumptions about the speed at which carbon in some of the tree biomass is returned to the atmosphere should be reconsidered (Pitkänen *et al.* 2012).

Predicted future rates of carbon sequestration should also be re-evaluated with consideration of climate change. While forests on mineral soils are predicted to remain a sink, peatland forestry is likely to become a source of net emissions as warming and increasing water table draw down increases CO_2 production in exposed peat (Minkkinen *et al.* 2007a, Sievanen *et al.* 2014). Other contributors to radiative forcing should also be more fully considered in examining Fennoscandian data. Albedo, while probably not as important a driving factor as carbon sequestration (Lohila *et al.* 2010), is often missed from studies. Similarly, the incorporation of other greenhouse gasses, particularly N_2O, may prove to be a significant and underappreciated factor that may shift the greenhouse gas balance from sink to source (Von Arnold *et al.* 2005, Maljanen *et al.* 2010). Modelling work using "CoupModel" and based around the Skogaryd research station (in peat initially drained for agriculture then converted to forestry) in Sweden showed that when only carbon emissions were considered drained and forested peatlands were a net sink for greenhouse gas (16.0 kg C m^{-2} over 60 years). However, when the emission of N_2O was included the site become a small net source (equivalent to 26.4 kg C m^{-2} over 60 years). Including emissions derived from thinning the felling, the site becomes a massive source of greenhouse gas (He *et al.* 2016).

Other sources of carbon from forestry may also be poorly accounted for, particularly those resulting from the erosion of peat which is inherent in drainage (Marttila & Kløve 2010). Clear cutting increases exports of DOC (Nieminen 2004, Nieminen *et al.* 2015) and may cause a pulse of GHG production in the immediate aftermath of felling as tree photosynthesis is removed and CO_2 is produced by decomposition of tree residues (Korkiakoski *et al.* 2019). There is an increasing awareness that problems of the quality of water exported from drained peat must be addressed (Nieminen *et al.* 2020). Such considerations have led to the examination of other forest management options and to the possibility of restoration (Vasander *et al.* 2017).

Continuous cover forestry, using natural regeneration and avoiding variation in water table, has been suggested as a possible future forestry management option in Fennoscandia (Nieminen *et al.* 2018), and to be encouraged through the use of “carbon payments” (Pukkala 2020). Restoration, through tree removal and rewetting of peat, has been shown to revert greenhouse gas fluxes to levels found in undrained sites (Laine *et al.* 2019). Such studies are at an early stage, and long term structural changes to the peat are still poorly understood (Haapalehto *et al.* 2011) meaning there may yet be unforeseen difficulties in rewetting land (Kløve *et al.* 2017).

Applicability of previous research to UK peatlands

The majority of available data from Fennoscandia have been obtained from minerotrophic fen sites or naturally wooded bog sites, both of which tend to have greater timber production than do ombrotrophic bog sites (Minkkinen *et al.* 2002, Drosler *et al.* 2008, Maljanen *et al.* 2010). These issues, combined with the differences in climate between Fennoscandia and the UK, mean that any comparison between forestry on peatlands in Nordic countries and afforestation of peat bogs in the UK and Ireland must be made with considerable care and may sometimes be inappropriate.

UK blanket bogs are naturally treeless, at least in their broad central expanses, requiring cultivation and more fertiliser than would be used elsewhere (Laine *et al.* 1995). The natural or pre-existing conditions on many of the Finnish peatlands are very different from the UK, and typically may include dwarf trees and scrub (Laiho & Laine 1997) or even a significant pre-existing tree cover (Minkkinen *et al.* 2002). Fennoscandian bogs typically have peat with inherently very low hydraulic conductivity (Päivänen 1973), so that water table drawdown in response to drainage is probably more limited in depth and extent, with resulting aeration of the peat more limited. In addition, many peatland sites drained for forestry in Fennoscandia are minerotrophic fens and thus the required site treatments, planting methods and suitability for silviculture differ significantly from blanket bog peatlands of the UK and Ireland (Minkkinen *et al.* 2002, Maljanen *et al.* 2010). In Fennoscandia trees are often not actively planted. Drains are instead dug across peatland systems in order to encourage growth of existing trees which grow sparsely or in a variety of growth forms prior to drainage. While work on site is often required to cope with forest regeneration, forestry activities are generally restricted to deep ditching and fertiliser application, along with appropriate thinning as the forest develops (Päivänen & Hånell 2012). In consequence, such peatlands suffer less direct disruption during site preparation than in the UK, and this allows much of the original vegetation to remain and leaves the peat relatively undisturbed (Laine *et al.* 2009).

The use of closely-spaced plough furrows between the deeper drainage systems is, thus, almost unique to the UK and Ireland and this may explain many of the observed differences between peatland forestry responses here compared with those reported from the rest of northern Europe. These differences are worth considering in detail before Fennoscandian evidence is used to inform UK policy, as emissions from UK peatlands are likely to be much greater.

Evidence from Ireland and the UK

There has been limited work on afforested peatlands in the UK and only a few studies have considered the consequences of peat bog forestry for carbon storage (Table 2.2). Reviews and carbon accounting studies have often integrated Fennoscandian data to argue that planting on peat would produce a net carbon accumulation in UK peatlands over the first 100 years (Cannell 1999, Worrall *et al.* 2010).

Table 2.2. Published empirical studies on carbon in afforested peat bogs in the UK and Ireland.

Authors	Year	Location	Type of peatland	Timescale of study	Measurement	Method
Byrne & Farrell	2005	Cloosh Forest, County Galway, Ireland	Ombrotrophic blanket bog	Two 24 hour measurements, repeated 13 times	CO_2	Soda lime
Hargreaves et al.	2003	Auchencorth Moss, Midlothian, Scotland	Extensively drained ombrotrophic blanket bog	22 months, continuous	CO_2	Eddy covariance
		Bealach Burn, Sutherland, Scotland	Ombrotrophic blanket bog	Month long continuous measurements, repeated at different aged forest stands		
		Channain Forest, Sutherland, Scotland	Peat of 1m depth			
		Mindork Moss, Newton Stewart, Scotland	Peat of 2m depth			
Yamulki et al.	2013	Flanders Moss Forest, Scotland	Ombrotrophic raised bog	Two years, 2 - 4 week intervals	CO_2, CH_4, N_2O	Chamber flux

In County Galway, Ireland, Byrne & Farrell (2005) examined CO_2 fluxes from afforested blanket peat. They found that CO_2 loss from drained and planted peat was similar to estimated uptake of carbon by the tree stands, suggesting that there would be no net loss of carbon (Byrne & Farrell 2005). However, DOC and POC export from the site were not considered, meaning that total carbon loss was likely to be greater than uptake by the trees. Furthermore, the 'soda lime' method was used to measure the CO_2 flux from the peat; an approach that the authors acknowledge underestimates CO_2 relative to direct instrumental measurements. And the sampling was for relatively short time periods only.

Hargreaves *et al.* (2003) conducted a study of carbon flux from three afforested bogs in Scotland representing different tree maturities, and one unplanted control site. This study had continuous eddy covariance assessment for over a year in the control site, but only extrapolated from shorter periods of measurements in the afforested sites. The article concluded that afforested peatlands will accumulate more carbon due to forestry than would be lost because of planting and drainage. This was believed to hold true for 90–190 years, after which restoration should take place because the amount of carbon in the tree biomass and peat would fall below that which would have been sequestered by undrained peat (Hargreaves *et al.* 2003). However, these conclusions are questionable. In fact, the site used to provide the baseline control had previously been drained extensively, and so provided unusually low carbon accumulation values. Also, in the 'mature' afforested site, canopy closure was not complete and the stand was up to 30 yr from a full rotation, meaning that the carbon loss from a large proportion of the life of the forest stand was not properly accounted for, nor were DOC and POC losses considered (Lindsay 2010).

Another important UK-based study is that of Yamulki *et al.* (2013), who studied gas fluxes and DOC loss from sites at West Flanders Moss in Scotland. While suggesting that drainage increases GHG emissions by 33 %, they concluded that increased CH_4 emissions from rewetted bogs would outweigh the reduced CO_2 emissions, meaning that restoring forest to bog is likely to increase potential warming effects on climate (Yamulki *et al.* 2013). This article has been criticised by Artz *et al.* (2013), who pointed out that there were problems with the control being unrepresentative of undrained bog, with higher than expected CH_4 fluxes, and that there were calculation errors. Further investigation revealed that the control was in an area that had been dug out as a reservoir for flushing cut peat into a nearby river around 100 years previously. The flux work also ignored above-ground tree respiration, comparing below-ground CO_2 flux under forest stands to total above- and below-ground flux in the control. In addition it was noted that carbon budgets of restored sites change over time, and as restored sites mature the vegetation cover becomes less 'patchy', producing a stronger CO_2 uptake which would make restoration seem more beneficial (Artz *et al.* 2013).

Summary and recommendations

The evidence base for the effects of afforestation on UK peat bog carbon is weak, and research is often underpinned by data taken from other regions, particularly Fennoscandia. Such studies rely on assumptions that may not hold for conditions in the UK. There is also a bias within the research towards measurement of gas flux without considering other pathways of carbon loss from the system (particularly aquatic pathways). At present it cannot be reliably determined whether afforestation of open UK peatlands exacerbates or ameliorates climate change.

As existing forests on peat come to harvesting age, decisions must be taken to either restock trees or, where possible, to restore bog habitats. The benefits of restoration on biodiversity are well understood. This is chiefly through a reduction in habitat fragmentation, which is beneficial to the specialist bog species. In particular, ground nesting birds and waders are much less likely to suffer predation from mammals hunting on the margins of forestry (Wilson *et al.* 2014). As the effects on carbon are more uncertain, work is urgently required to plug gaps in current knowledge (IUCN 2014).

Better data on the yields, quality and ultimate use of peat bog forests in the UK are needed. There must also be a proper quantification of other aspects of climate effects including fossil fuel use in ploughing, planting, fencing, fertilising, drain maintenance, road building and the effects on albedo, emissions from transport, and the fate of the wood products. Further use of whole-column inventories should be made to provide peat carbon budgets over the life of a plantation, particularly if the ground is to undergo restocking. Such carbon stock research must be integrated with flux studies to provide a complete long-term picture of total changes in carbon storage and the processes by which these changes occur, which will determine the loss of carbon to the atmosphere relative to accumulation in tree biomass and quantify any resulting global warming potential.

A wide range of organisations (government, academic, charity and non-government) are now addressing the effects of peatland forestry. A coordinated effort is required to plan and share peatland forestry research, to provide a sound body of evidence for approaching policy decisions. Work on the carbon effects of forestry needs to be understood in relation to research on the economic and ecosystem services provided by peatlands. This is a particular priority in the Flow Country, the UK's most extensively afforested peatland region.

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The value of an initial assessment of subsidence and peat depth

Chapter two described the history of UK peatland forestry and explored some of the mechanisms through which drainage and afforestation may alter the carbon balance of peat. The following chapters will quantify the physical changes in UK afforested peat, beginning with subsidence at the intensively studied site at Bad a'Cheo.

Ground level subsidence and a thinning of the peat are some of the more physically obvious signs of the changes caused by drainage and afforestation and may indicate possible loss of carbon. The type of surveying work presented in chapter three is therefore an important first step in assessing such sites and provides a context for the quantification of the change in carbon storage described in subsequent chapters.

This chapter will also discuss in further detail the history of work on the Bad a'Cheo experimental forest. It will integrate and update some of the research that has been undertaken since 1966 and will provide the final data set prior to the felling of the plantation in 2017. The survey design seeks to be as robust as possible while integrating several disparate sources of previous data. This has been achieved through interpolations of old data sets and in resurveying old transects, all on a site that has been made partly inaccessible by extensive wind-throw. The considerations which led to the selection of the experimental design will be further elaborated in chapter three.

Chapter 3 - Ground surface subsidence in an afforested peatland fifty years after drainage and planting

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Summary

In the UK, large areas of peatland were drained for forestry in the second half of the twentieth century. Ground surface subsidence and diminishing peat depth can indicate compaction of the peat and/or carbon loss, but there are few long-term datasets from afforested UK peatlands. Here we present an unprecedented fifty-year time series of surface subsidence from Bad a'Cheo Forest (Caithness, Scotland). This site was initially surveyed for ground level and peat depth in 1966, prior to drainage and plantation, with further surveys repeated roughly twenty and thirty years after drainage. We re-surveyed the site fifty years after initial drainage, producing a unique long-term time-series to assess change since these historical studies. Significant subsidence has taken place since drainage, with an average reduction of 56.8 cm (or 13 %) in peat depth under forest stands. Subsidence was rapid in the initial phase after drainage and planting but has progressively slowed with relatively little change between the surveys in 1996 and 2016. These results imply the potential for carbon loss, but do not demonstrate this directly as compaction of the peat is also probable. These subsidence data demonstrate that drainage and afforestation leads to considerable loss of peat depth and show how this evolved through time.

KEYWORDS: afforestation, Flow Country, peat, subsidence

Introduction

Globally, peatlands store up to 600 Gt of carbon (Yu 2011); the fate of this large carbon stock is of considerable importance to climate change (Charman *et al.* 2013). In addition, peatlands provide a wide variety of other ecosystem services, including water purification and storage, cultural ecosystem services, and biodiversity (Littlewood *et al.* 2010). Peatlands are also managed for a variety of commercial purposes, which often involve drainage, a key driver of carbon loss (Whitfield *et al.* 2011; Joosten *et al.* 2012).

Most UK peatlands are not naturally forested but large areas have been planted with trees in the last century. Initially forestry on deep peat presented huge challenges due to high water tables and low nutrient availability (Anderson 1997). However, development of new ploughing techniques, application of fertiliser and the adoption of North American conifer species in the second half of the twentieth century made afforesting UK bogs a technical possibility (Oosthoek 2013). While there was doubt in some quarters that many sites would produce a commercially viable yield within the first rotation, large tax breaks were exploited, leading to afforestation of extensive areas of deep peat (Mather & Murray 1988). Around 15% of the UK's deep peats were drained for forestry, amounting to a total of approximately 190,000 hectares (Cannell *et al.* 1993). The practice only ended in the late 1980s, after controversy led to a change in tax law and an increase in protected areas (Stroud *et al.* 2015). The most extensive afforestation of deep peat in this period was in the Flow Country of northern Scotland; the UK's most extensive area of blanket bog, with around 400,000 ha of peat and wetlands, of which around 67,000 ha (approximately 16.8 %) has been afforested (Stroud *et al.* 1987).

Drainage of a peat bog gives rise to several important processes. Primary consolidation is caused by loss of water from large pore spaces within the peat, as drainage directly removes water. Secondary compression occurs because more tightly-bound water is then gradually squeezed from the bottom layers of peat by the weight of overlying peat no longer supported by the water table. Thirdly, drainage enables oxygen to penetrate the upper catotelm, exposing the long-term carbon store to oxidative decomposition by bacteria and fungi, leading to increased production and efflux of CO_2 (Eggelsmann 1975). Water drained from the system also directly exports a large quantity of dissolved and particulate organic carbon (Freeman *et al.* 2001). If peatland is planted for forestry the peat may be further compacted over time as the weight of growing trees increases (Hobbs 1986). These processes are likely to be associated with subsidence of the ground surface and diminishing peat depth, as well as an increase in bulk density (Holden *et al.* 2004). While subsidence and reduction in peat depth may simply reflect compaction of the peat body without carbon loss, the loss of carbon stock remains a distinct possibility. Loss of peat depth can be cautiously considered as an indirect indicator that peat carbon loss may have occurred, and has been used to infer carbon loss in some previous studies (Leifeld *et al.* 2011; Hommeltenberg *et al.* 2014).

Some subsidence studies have been carried out on afforested peatlands with the largest quantity of data from Fennoscandia. A survey of 273 forestry-drained peatland sites (primarily pine fens) across Finland found an average 22 cm subsidence over 60 years, a figure low enough to suggest increased carbon storage in the system when tree biomass is included (Minkkinen &

Laine 1998). Similarly, fen sites in Latvia have shown similar mean subsidence of 26 cm over 54 years (Lupikis & Lazdins 2017). Data is rarer from the naturally-open bogs which are typical sites used for forestry in the UK. On a bog in Southern Norway, 70 cm of subsidence was recorded 26 years after drainage, fertilisation and afforestation via naturally seeded Scots Pine (*Pinus sylvestris*) (Braekke 1987). Despite widespread recognition that subsidence is important, and the relative simplicity of data collection, there is remarkably little long-term data from afforested UK bogs (Lindsay *et al.* 2010). Indeed, the site we discuss here is, to our knowledge, the only UK afforested blanket bog site with any long-term subsidence series.

Many UK peat bog plantations from the 1960/70s are now ready to be harvested, and as such, decisions must be made as to whether to restock plantations or to restore bogs. Restoration attempts to reinstate the hydrology and species composition of bogs through tree felling and rewetting (Forestry Commission Scotland 2015; Forestry Commission Scotland 2016). To inform decisions about which option is most appropriate, quantitative evidence for how afforestation has affected the functioning of peatbogs and the ecosystem services they provide is required. Biodiversity, and the economic value of forest plantations are important concerns, but carbon loss is a particularly important factor due to the large amount of carbon stored in these systems (Yu 2011; Billett *et al.* 2010). This study combines ground level and peat depth surveys with historic data sets, to demonstrate the effects of drainage and plantation on a blanket bog over fifty years.

Methods

Study area

One of the most intensively studied afforested sites within the UK has been the Forestry Commission experimental plantation at Bad a'Cheo in Rumster forest, Caithness. Bad a'Cheo (58°25′49.28″N, 003°25′41.30″W) covers an area of approximately 50 ha of ombrotrophic blanket bog, at an elevation of approximately 90 m above sea level. Humification measured on the Von Post scale during this survey is in the range of H3 – H8 throughout the site, typically increasing deeper into the catotelm. In the top 3 meters of the peat, bulk density averages 0.07 g/cm^3 in the wetter open bog areas, rising to an average of 0.1 g/cm^3 towards the centre of the forest stands. Bad a'Cheo was initially surveyed for ground elevation and peat depth in 1966, with the site drained, ploughed with a double mould board plough, and afforested in 1968 in a randomised block design as a Forestry Commission experiment. The experiment contains individual blocks of Sitka spruce (*Picea sitchensis*) and Lodgepole pine (*Pinus contorta*), and mixed stands of both species planted in alternate rows. Plough furrows are around 30 cm deep across the site, with drainage ditches of up to 1 m deep spaced at approximately 20 m intervals. Towards the edge of the site where there has been no drainage the water table is close to or on the surface but is considerably lower in drained forest stands. In 1989, a second randomised block experiment was set up, occupying the unplanted control plots of the first experiment and testing the performance and immediate hydrological impact of the then-current afforestation options (Miller *et al.* 1996; Anderson *et al.* 2000). The entire plantation was felled in 2017 prior to wind-farm construction and our study was conducted in a brief window before this work.

The Bad a'Cheo site presents a unique opportunity to assess long-term change in peat depth with subsidence over a full growth cycle, from before planting to immediately prior to felling. The site has the further advantage that planting was conducted earlier than the majority of UK peatland plantations, offering an insight into future trajectories of change elsewhere. This study compares a new survey of elevation and peat depths with previous surveys conducted in 1966, 1987, and 1996, to produce a unique long-term time series.

Previous studies

Since the initial drainage and plantation, several studies have been undertaken to assess the impact of forestry and drainage. These previous studies constitute an important long-term data resource, although as we discuss below, their reanalysis is not straightforward. Prior to planting, in 1966, ground elevation across the whole site was measured to the nearest 0.025 m using optical levelling with a contour map produced with intervals at 0.15 m. Peat depth was also measured in 1966 on a grid of approximately 55 m intervals and recorded to the nearest 0.3 m.

In 1987, Pyatt *et al.* (1992) assessed the effects on the bog of conifer planting after 20 years, in a study which included measurement of ground elevation along three transects (Table 3.1) using optical levelling equipment. Further surveys by Shotbolt *et al.* (1998) were completed in 1996. Again, the ground elevation was measured along one of the original Pyatt transects, a new short transect, and at 101 random points on site, and compared to ground levels estimated by interpolation between the original site survey grid points.

With this history of data collection in mind it is worth discussing some of the practical considerations of working on a mature deep peat plantation, and how these considerations influenced the experimental design. As discussed in chapter 1 and above, Bad a'Cheo is a uniquely well studied afforested peat site with data stretching back over fifty years. This provides an excellent opportunity to reanalyse existing datasets, particularly concerning the changing physical characteristics of the peatland.

Due to the growth of the forest it was impossible for Shotbolt *et al.* (1998) to repeat the original 1966 peat depth sampling points, and it remained impossible for this survey. No detailed notes or primary data remain for the 1966 ground level survey, only the finished elevation map. As the original methods could not be repeated two alternative methods, random sampling and transects, were considered. Random sampling elevation and peat depth would have the advantage of producing a widely distributed data set distributed fully representative of the site. Sampling along transects would have the advantage of replicating some of the previous work, and of providing series of high-resolution profiles showing the extent of the impact of plantations. Transects are arguably not as representative as a truly random design but could be placed across the site to mitigate this as far as possible.

Outside of the advantages in allowing direct comparisons to be made to older data transects were found to be the only practical way of producing meaningful results, given the constraints inherent to working on deep peat forestry plantations. Initial site surveys and scoping work revealed that there had been a deterioration on the site in the decades since the last full survey was performed, with a high incidence of wind-throw throughout, and with many sections where there had not been brashing work to remove branches now completely impassable.

In the roughly 20 years since Shotbolt *et al.* (1998) found many of the original sampling points to be inaccessible trees have matured and the wind-throw has become more extensive, exacerbating this further. Many points could simply not have been reached or would be impossible to survey for fallen wood. As data collection would therefore have been channelled into a few areas a truly randomised sampling design would probably be impossible. Other areas would be accessible, but the specific sampling point would have been difficult to accurately identify within the time available on site (due to the imminent felling of the forest) as GPS functionality under forest canopies is low. Measures of ground elevation would have been especially impacted as optical survey requires back-sighting to benchmark markers (see section "Measuring ground elevation"). In back-sighting the poor visibility in forests outside of the brashed corridors and the consequent repeated movement of the optical levels would have produced data with unusably high levels of error. Overall, the use of transects was the only practical approach given the condition of the site, and transects were spread across the site as widely as was practical to ensure they are as representative as possible of the wider system.

Establishing transects

We conducted ground level and peat depth surveys in 2016 and 2017 respectively, immediately prior to the final felling of the forest stand. We conducted three phases of data collection: i) a re-survey of transects surveyed previously by Pyatt *et al.* (1992) and Shotbolt *et al.* (1998), ii) a survey of two new transects across the site margins and iii) a site-wide survey of peat depth for the first time since 1966.

Previous transects were re-located using site maps, information on starting locations and markers originally installed along the transects (primarily dipwells and wooden posts). Peatlands are highly variable heterogenous landscapes where different microtopographies can occur over short distances. The transects were recovered with enough accuracy (estimated ± 10 cm) that a sampling point is unlikely stray into a different microtopography (e.g. a sampling point originally on a hummock would not be resampled in a hollow), but there may be some small scale difference from the original sampling point. While any differences in elevation or depth arising from this is likely to be several orders of magnitude less than the changes caused by drainage, it is nevertheless important to be aware of such variations. Transects are referred to following the name of the author and the number referred to in the original publication (e.g. Pyatt 1 is the first transect surveyed by Pyatt *et al.* 1992, Figure 3.1). Transects Pyatt 1 and Shotbolt 2 were re-located with a relatively high degree of accuracy (estimated ± 10 cm) using on-site markers, although heavy wind-throw meant that only 326 m of the original 430 m of Pyatt 1 could be re-surveyed. The exact starting point and approximate route of Pyatt 2 was identified, primarily using brashed avenues though the forest stand, although no markers were found. As such this transect is an approximate re-creation, and the sampling points could not be recreated with the same level of accuracy as in the other transects (Table 3.1).

Table 3.1. Length and measurement history of transects at Bad a'Cheo. Data collection for the current study was in 2016, when all transects surveyed contained forest stands, internal open ground and undrained bog. 1966 surveys (marked*) are based on results interpolated from the initial site survey and have not been directly measured along the transects.

Transect Name	Date established	Original length (m)	Previous survey dates	2016 resurvey length (m)	Accuracy of resurvey to original transect line
Pyatt 1	1987	430	1966*, 1987, 1996, 2016	326	Line resurveyed, last 104 m lost to windthrow
Pyatt 1a	1987	87	1966*, 1987, 1996	NA	Transect could not be located
Pyatt 2	1987	326	1966*, 1987, 2016	350	Approximate line resurveyed
Shotbolt 2	1996	75	1966*, 1996, 2016	70	Line resurveyed, last 5 m lost to windthrow
New 1	2016	NA	1966*, 2016	265	NA
New 2	2016	NA	1966*, 2016	193	NA

To increase coverage across the site, two further transects (New 1 and 2) were added at the southern end of the plantation (Figure 3.1). Like the original transects they spanned a length of open, undrained bog as well as afforested plantation. Some previous survey points could not be relocated or were not considered suitable for resurvey. The 101 random points surveyed by Shotbolt *et al.* (1998) were not resurveyed as detailed records of their location were unavailable, while a final short transect from Pyatt *et al.* (1992) could not be re-located.

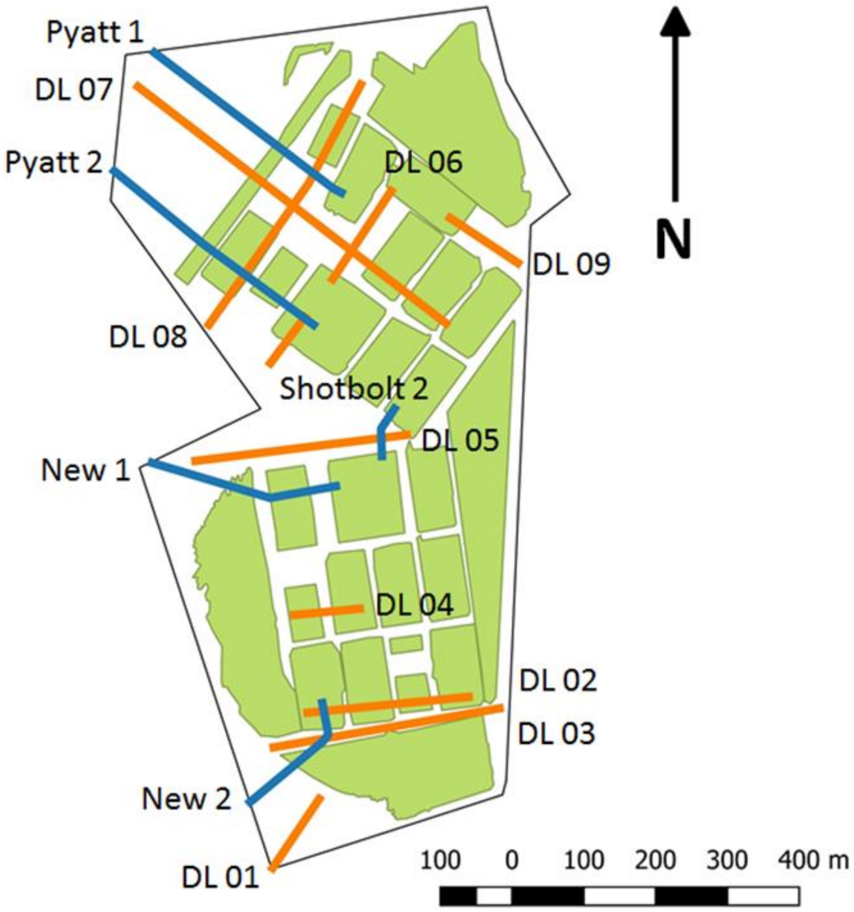

Figure 3.1: Bad a'Cheo peat depth and elevation survey locations. Blue lines indicate transects surveyed for ground elevation and peat depth. Orange lines indicate transects surveyed for depth only.

During the initial 1966 survey, five metal markers were drilled into the mineral layer underlying the peat at the corners of the site. These pipes were of known elevation and were used as benchmarks for the original ground level surveys. The pipes were re-located and used as benchmarks in this study with locations recorded using DGPS (Trimble, R8 GNSS/R6/5800). As the DGPS system could not be used accurately under the forest canopy, only open bog sections were recorded, with the remaining points derived by interpolation

Measuring ground elevation

Ground elevation was surveyed along the previously established transects (Pyatt *et al.* 1992; Shotbolt *et al.* 1998), and along the two newly-established transects. Previous surveys were carried out with optical levels, so similar equipment was used to repeat the measurements in this study (Level Mark, AL10-32). Optical levelling techniques must be approached with caution on bogs, as the relative ease of disturbing the peat surface can lead to high accumulated errors after repeatedly moving the tripod. Nevertheless, when used with care optical levels can return good quality data and have the benefit of replicating the methods of previous work at Bad a'Cheo. It is worth noting that of the possible alternative methods DGPS was already deployed as part of this work, and conceivably could have been used to make an elevation survey. DGPS was not used in this way as the method is unreliable in forested areas, and experimentation on site demonstrated that the signal between the base station and mobile unit cannot be maintained. On the original transects the same sampling intervals as those used in 1987 and 1996 were measured, which gave 10 m intervals in the open bog, and narrower more erratically spaced intervals in the forest stands (typically 1.0 – 0.3 m). In the new transects, open bog was surveyed at intervals of 5 – 10 m depending on the variability of the microtopography, and at 0.5 m resolution under the forest stand to capture the increased variability of these ploughed areas. DGPS was again used to accurately record the location of sampling points along the transects in open bog.

Measuring peat depth

Peat depth was surveyed along all the ground elevation transects, as above. In addition, we undertook a site-wide re-survey of peat depth for the first time since 1966. This survey was conducted along nine new transects established in 2017 (Figure 3.1, Table 3.2) and recorded at 5 or 10 m intervals (Figure 3.1). Coring established that basal peat directly overlies sand, clay and rock in the site so peat depth was measured by inserting a sectional peat probe into the ground until contact was made. This is the standard methodology for any survey of peat depth. Due to limited availability of DGPS, and its unsuitability for use across a large forested plantation, a GPS (Garmin, GPSmap 62s) was used to mark transect locations for later plotting against the 1966 depths.

Table 3.2. Description of additional transects added for depth surveys. 'Undrained bog' refers to areas extending from the edge of the plantation, where the hydrological impact of drainage will be diminished. 'IOG' refers to Internal Open Ground areas between forest stands which have not been drained for plantation, but may be impacted by the hydrological changes caused by nearby trees.

Name	Length (m)	Number of points	Areas covered
DL 01	110	11	undrained bog
DL 02	320	33	IOG
DL 03	250	26	forest stand
DL 04	100	11	forest stand
DL 05	300	61	IOG
DL 06	230	46	IOG, forest stand
DL 07	550	55	undrained bog, IOG, forest stand
DL 08	400	41	IOG, forest stand
DL 09	120	13	IOG, forest stand
Pyatt 1	326	36	Undrained bog, IOG, forest stand
Pyatt 2	350	33	Undrained bog, IOG, forest stand
Shotbolt 2	70	16	Undrained bog, IOG, forest stand
New 1	265	26	Undrained bog, IOG, forest stand
New 2	193	17	Undrained bog, IOG, forest stand

Data analysis

Data was digitised to compare ground elevations and peat depths between the various surveys. Interpolated values from before drainage and planting (1966 survey) could then be compared to subsequent data gathered over the following fifty years. The methods used for this interpolation are described below, but it is important to remember that interpolated data does not represent a "real" physically measured value, but rather a probable value extrapolated from several nearby measurements. There are therefore two types of data considered here: an interpolated value for 1966, and a field measured actual value for all subsequent surveys. Each of these sources of data have different considerations attached.

Measured data are subject to some seasonal variation arising from factors such as 'bog breathing' (Morton & Heinemeyer 2019). Interpolated data are derived from directly measured values and so are also affected by these considerations. In addition, interpolation does not provide any information on microtopography and small-scale variation and therefore represents something of a 'smoothing out' of the data. It is, however, extremely rare to have survey data over a fifty year time scale, and even with these considerations the novelty of such data means that it is worth exploring. This is especially so as previous interpolations of these data (Shotbolt *et al.* 1998) were through a 'pencil and paper' calculation, rather than the more statistically rigorous GIS techniques described below.

Much of the raw data from the original 1966 surveys has been lost. No notebooks containing original elevation or peat depth measurements could be located in Forestry Commission archives. A single paper survey map produced from these measurements was the only remaining data source. Scans of the original notebooks from the more recent Pyatt *et al.* (1992) study were available, along with raw data from Shotbolt *et al.* (1998) in the form of an appendix to her thesis (Shotbolt 1997). These disparate data needed to be digitised and combined for reanalysis.

A high-resolution scan of the 1966 survey map was produced (Figure 3.2). GPS and DGPS data including corner points and benchmark locations (DGPS and GPS), peat depth survey transect locations (GPS), and ground level transect locations (DGPS) were uploaded to QGIS (version 2.14.21, 'Essen'). Where tree cover had prevented accurate use of DGPS, missing survey points were added to the transect lines at the intervals at which they had been surveyed.

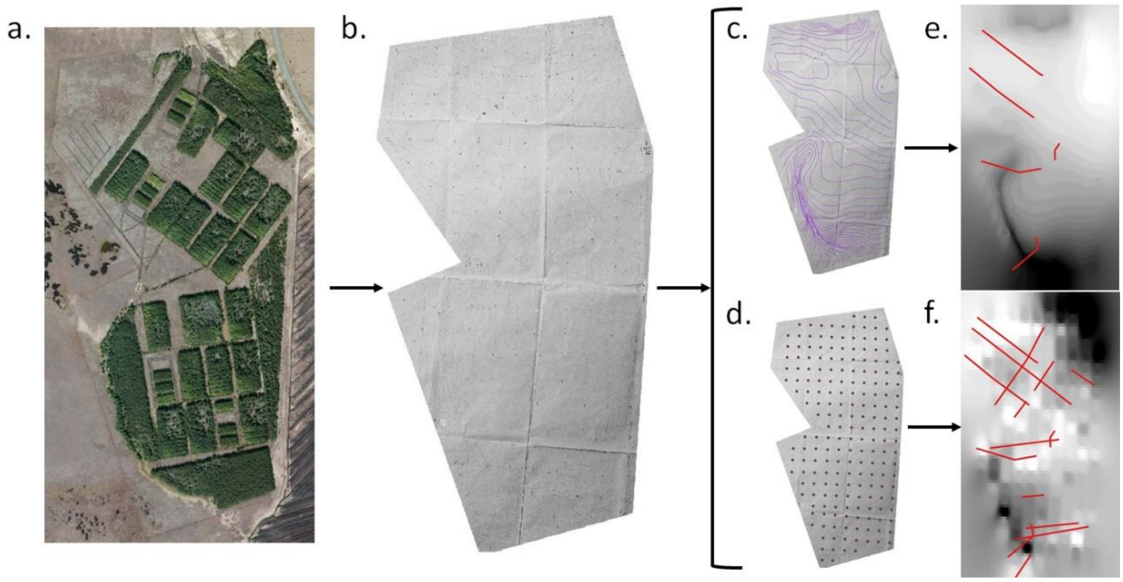

Figure 3.2. Data from Bad a'Cheo (a) were georeferenced to the 1966 Bad a'Cheo survey map (b). The values from the 1966 survey were used to determine likely surface elevation and peat depth. Contour lines (c) and peat depths (d) were digitised and interpolated in GIS to generate elevation and depth maps from which data for each of the transect sampling points could be extracted. Darker areas represent (e) lower elevation and (f) shallower peat, respectively.

Using the benchmarks and additional corner points, the 1966 survey was georeferenced to the transect data. The contour lines of the 1966 survey were digitised and assigned elevation values converted to metres. These levels were used to produce an interpolated map of projected ground levels in 1966, using inverse distance weighting on a grid of 100 x 100 cells. Depth was also interpolated using inverse distance weighting from the 1966 map values, using the grid of survey points converted to meters and rendered in a grid of 100 x 100 cells.

Differences between interpolated values for the 1966 survey were tested to determine the change in elevation and peat depth since planting. Transects were divided and analysed separately in three classes based on the likely hydrological impact of plantation: forest, undrained bog and internal open ground (IOG) defined as the areas between forest stands which have not been drained for plantation, but may be impacted by the hydrological changes caused by nearby trees. Forest and IOG were hypothesised to have subsided following drainage, with the impact being greatest on the directly drained forest sections. The undrained bog was hypothesised to have not subsided or thinned. In this way measurements in areas of undrained bog at the margins of the site further away from forestry were distinguished from the measurements taken in the rides and larger areas of internal open ground between the drained, planted blocks. This is a slightly different classification scheme than had been used in some previous work on the site (Pyatt *et al.* 1992), which had distinguished unplanted bog more than 10 m away from trees from unplanted rides within 10 m of trees and a third category of unplanted plot and ride 10 – 40 m from trees. The reclassification used in this study aimed to simplify the categories used, and avoid categories with small numbers of data points.

Measured elevation was then compared to the previous Pyatt *et al.* (1992) and Shotbolt *et al.* (1998) measurements to determine the extent of subsidence over the final twenty years of the fifty-year forest rotation. During the exploration of the notebooks that contained the data from these previous surveys, it became apparent that while data for Pyatt 1 and Shotbolt 2 were usable, benchmark data for Pyatt 2 had been lost, rendering them unusable.

In each instance, Wilcoxon signed rank tests were used to compare pairs of measurements taken at the different sampling times (e.g. 1966, 2016 etc.) at each point (IBM SPSS Version 24).

Results

Data quality and limitations

Before considering the results of this study, some possible sources of error should be acknowledged. Error during levelling surveys was found to average 4 cm, a figure comparable to previous surveys and relatively minor considering the difficulty of using optical levels on soft and wet peat surfaces. The transect specific errors from closing the survey loop were 5.5 cm on Pyatt 1, 4 cm on Pyatt 2, 1 cm on Shotbolt 2, 6 cm on New 1, and 3.5 cm on New 2, all to the nearest half centimetre. It is important to clarify that this error does not represent a statistical calculation of uncertainty, but rather directly measured value for the 'drift' of the optical level that describes the difference between the initial measurement of the benchmark and the final measurement of the same marker after closing the survey loop. Therefore, the uncertainty is not attached to each individual data point, but rather represents a cumulative error across the survey. This error is more likely to accrue as the survey progresses along the length of the transect, and this has two implications. Firstly, part or all of this error may have been accrued in the final back-sighting, following primary data collection, and so the actual error on each measurement may be lower than stated. Secondly, the need to begin the transect near to a benchmark meant that the transects were surveyed from the outer, undrained section into the drained afforested section, perhaps introducing a source of systematic bias in the reliability of the results. This means that in where there was no drainage and consequently the least change to detect the data is likely to be more accurate, while there is a greater possibility of error in the forest where the greatest effects are observed, although it is important to note that the magnitude of these effects are greatly in excess of the possible error.

Possible instrumental errors in the 1987 measurements of the Pyatt 1 transect were identified by Shotbolt (1997), suggesting that the ground surface over the IOG section would have risen by an improbable degree after afforestation (Shotbolt 1997). The difficulty of using levelling equipment on very soft bog surfaces means that the likelihood of recording error is high, and the marked disparity between the 1987 and both 1996 and 2016 recordings suggest possible problems with the 1987 survey across the ride. Ground elevation

Widespread and significant changes were found to have occurred across the site as a response to drainage and afforestation. Ground elevation was compared between the unplanted site in 1966 and the mature plantation in 2016 (Figure 3.3) using points along the transect lines (Table 3.3). Generally, drainage and planting led to a statistically significant fall in ground elevation in all but one (New 2) of the forested sections of the transects, with mean subsidence of 44.9 cm observed on the afforested section of Pyatt 1 (Table 3.3). Most transects showed no significant change in ground elevation in the undrained bog sections of the transect, except for New 1, which was significantly higher than the 1966 interpolation (Table 3.3). Most IOG sections showed a highly significant drop in ground elevation since 1966, except for Shotbolt 2 which was unchanged ($n = 60$, $Z = -1.966$, $P = 0.49$) and New 2 which showed significantly higher elevation ($n = 23$, $Z = -4.197$, $P < 0.001$) (Table 3.3). An average reduction in ground elevation, measured as the mean difference between the interpolated 1966 elevation and measured 2016/17 elevation, of 22.1 ± 1.96 cm was observed across all afforested portions of the site (an average

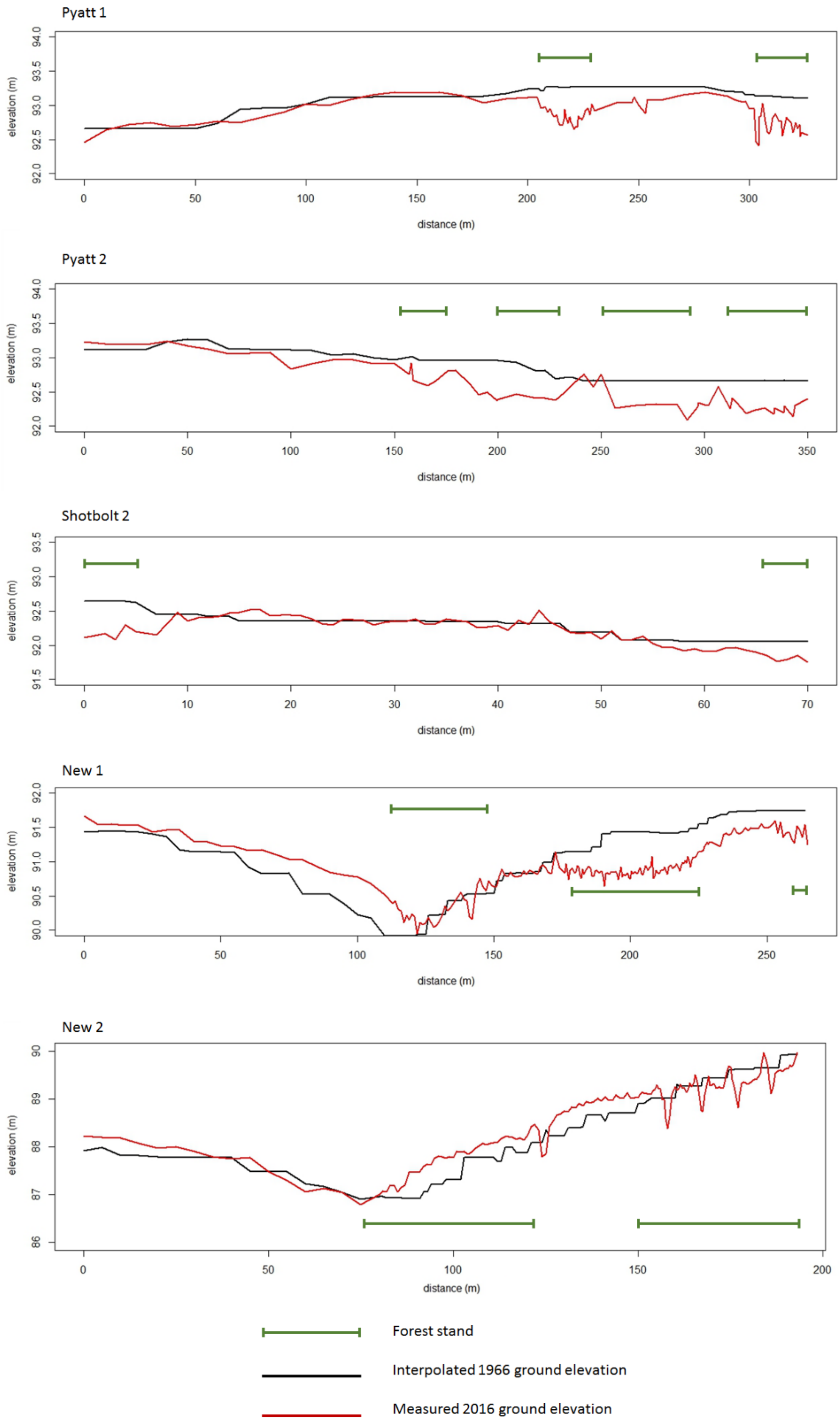

Figure 3.3. Ground surface elevation across Pyatt 1, Pyatt 2, Shotbolt 2, New 1 and New 2. The black lines indicate 1966 interpolated elevations and the red lines 2016 survey results. Green bars indicate the positions of forest stands.

that drops to 35.7 ± 2.33 cm if New 2 is excluded), with an average drop of 7.6 ± 1.94 cm in IOG and an increase of 6.5 ± 1.05 cm in undrained bog (Figure 3.4). Values for ± refer to standard deviations and this convention will be used throughout this thesis.

Where intermediate surveys were available, the ground levels from 1987 and 1996 were compared to the most recent data. In Pyatt 1, the elevation was not found to have changed significantly between 1996 and 2016 in either the undrained bog (n= 25, Z= - 0.525, P= 0.600) or forest sections (n= 45, Z= - 1.801, P= 0.072). A significant rise in ground level, on average 14.8 cm had taken place in the IOG sections (n= 31, Z= - 2.274, P= 0.023). Ground level subsidence since 1987 was found to have been significant in both the IOG (n= 9, Z= - 2.666, P= 0.008) and undrained bog sections (n= 22, Z= - 2.419, P= 0.016), with an average fall of 21.3 cm and 5.4 cm respectively. Data from 1987 suggests a rise in elevation in the IOG following afforestation, a finding which is surprising and calls into question the accuracy of the 1987 measurements, as previously highlighted by Shotbolt *et al.* (1997) (see above). There were insufficient data points to analyse changes in Pyatt 1 between 1987 and 2016.

The Shotbolt 2 transect showed no significant change under forest stands since 1996 (n= 10, Z= - 1.786, P= 0.074) but a significant fall across the IOG (n= 60, Z= - 2.075 P= 0.038) of on average 0.9 cm.

Peat depth

Mean peat depth under forest plantations decreased significantly between 1966 and 2016/17 (n= 121, Z= - 8.646, P < 0.001). Peat depth of IOG also decreased to a significant degree (n= 193, Z= - 4.820 P < 0.001) while external undrained bog peat was not significantly changed (n= 111, Z= - 1.015, P= 0.310). This represented an average loss of peat depth of 56.7 cm (13 %) in forest stands, 24.7 cm in IOG (5.5 %) and 3.1 cm (0.6 %) in undrained bog (Figure 3.5). Overall average peat depth in afforested areas dropped from 435.3 cm to 378.6 cm, 446.7 to 422.0 cm in IOG, and 475.1 cm to 472.0 cm in undrained bog. Overall, measuring peat depth indicated more extreme subsidence against the interpolated 1966 values than did measurements of ground elevation. Peat depth measurements showed an average reduction in peat depth 9.6 cm, 17.1 cm and 34.6 cm greater than that indicated by the ground elevation measurements in undrained bog, IOG and forest stands respectively.

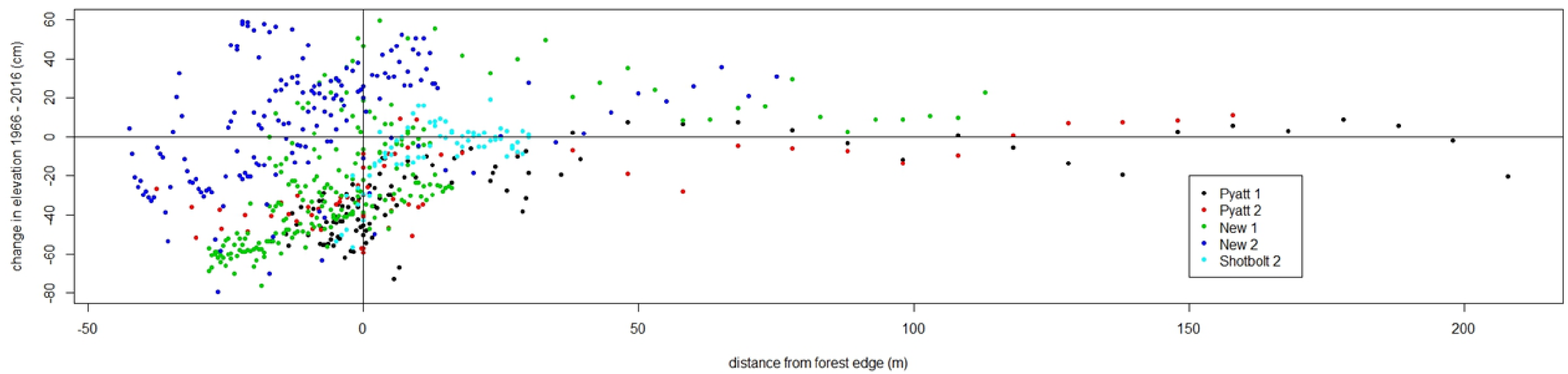

Figure 3.4. Change in ground elevation (cm) between interpolated 1966 points and measured 2016 plots, based on distance from plantation edge. Negative distance values reflect sampling points within the forest stand.

Table 3.3. Results of Wilcoxon signed rank tests to determine change in elevation between 1966 interpolations and 2016 survey, with the average mean change in ground level between 1966 and 2016. The un-forested area is further divided into 'undrained bog' (extending from the edge of the plantation, where the hydrological impact of drainage will be diminished) and 'internal open ground' (between forest stands which have not been drained for plantation, but may be impacted by nearby hydrological changes). N refers to number of points, Z is the test statistic, P is the significance, and GL refers to ground level.

	Undrained bog				Internal open ground				Forest stand			
Transect	N	Z	P	GL change cm (SD)	N	Z	P	GL change cm (SD)	N	Z	P	GL change cm (SD)
Pyatt 1	25	-1.951	0.051	-5.08 (1.08)	32	-4.937	< 0.001	-28.66 (1.64)	46	-5.905	< 0.001	-44.86 (0.93)
Pyatt 2	17	-1.728	0.084	-6.47 (1.09)	13	-2.691	0.007	-23.08 (1.91)	31	-4.86	< 0.001	-37.94 (1.13)
Shotbolt 2	NA	NA	NA	NA	60	-1.966	0.49	-2.26 (0.87)	10	-2.803	0.005	-39.2 (2.77)
New 1	23	-4.197	< 0.001	25.86 (1.68)	53	-5.219	< 0.001	-16.43 (1.60)	148	-9.152	< 0.001	-32.45 (1.04)
New 2	15	-1.931	0.053	10.86 (1.68)	27	-3.772	< 0.001	30.07 (2.18)	126	-1.09	0.276	3.24 (3.03)

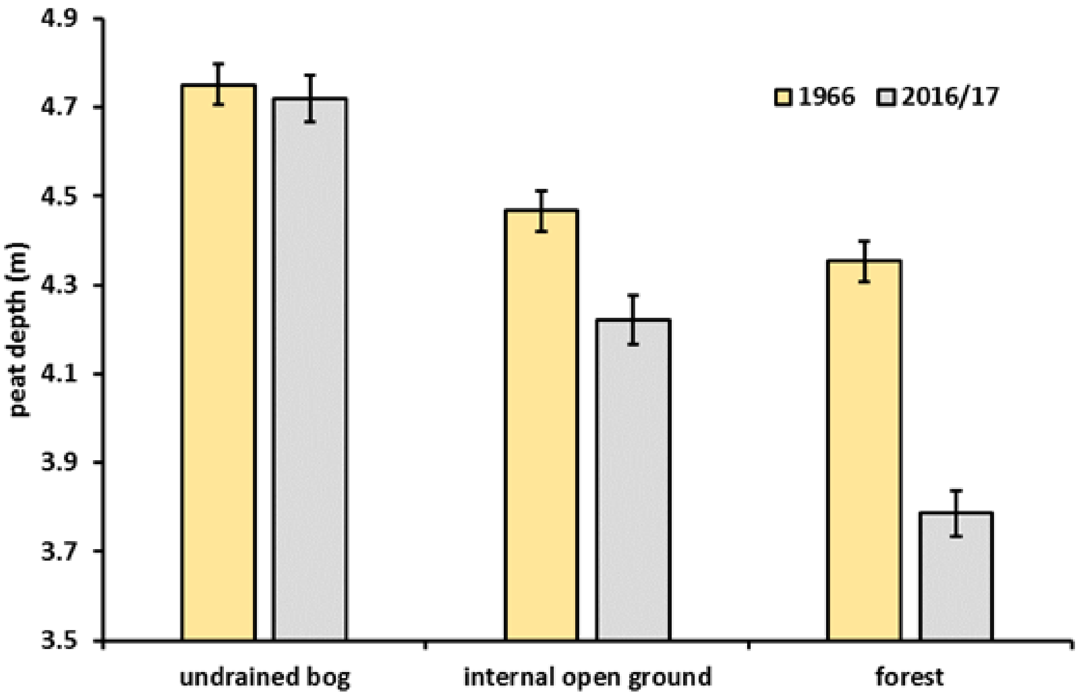

Figure 3.5. Mean peat depth across Bad a'Cheo: 1966 interpolated values and 2016/17 measurements, with standard error, n = 425.

Discussion

Comparison of interpolated undrained ground levels and peat depths with those fifty years after the drainage and afforestation of Bad a'Cheo reveals large changes on the site. Significant reductions in both peat depth and ground elevation are observed under afforested areas. This subsidence and reduction in peat depth is also seen throughout the IOG, even though these areas have not been directly drained. This suggests that afforestation impacts the areas of peat immediately surrounding plantations, as well as within, which may be important in estimating carbon loss and hydrological change in peatland forestry plantations.

The mean subsidence in afforested transects is of a similar order of magnitude to the limited pool of other datasets for bogs, for instance the 70 cm of subsidence recorded by Braekke (1987). The average reduction in peat depth of 56.7 cm under afforested areas is larger than has been observed in other drained peat systems (Dawson *et al.* 2010), and underlines the need for more data from afforested UK bogs.

Analysis suggested a significantly lower elevation and reduced peat depth since 1966 in both afforested peat and IOG. The extent of the subsidence and reduction in peat depth suggests large changes in the density and volume of peat, possibly accompanied by a loss of carbon. The lowering of the water table likely to have been associated with the subsidence reported here may have led to changes in other peat soil properties and features such as peat cracking, which can make restoration more difficult (Holden 2004). While the strongest subsidence and loss of peat depth occur soon after planting in the drained forest stands, we also find that the effects of drainage and afforestation have spread to the adjoining areas of peat, with peat significantly shallower than the 1966 interpolation in the IOG. In the specific case of the Shotbolt 2 transect, which contains one of the longest continuous stretches of IOG considered in this study, continuing subsidence across the IOG suggests that in some areas the impacts of afforestation on adjoining open peatland reported by Shotbolt *et al*. (1998) may continue over the life of the plantation.

In the Pyatt *et al.* (1992) study of the site, subsidence had been found to taper away 10 - 20 m from the edge of the forest plots, with the water table remaining unchanged beyond 20 m away (Pyatt *et al.* 1992). The IOG sections surveyed were located 0 - 45 m from the nearest plantation edge, and the significant subsidence associated with four out of the five elevation transects supports the interpretation that the lateral extent of the influence of subsidence at fifty years since afforestation is more in line with the 40 m observed by Shotbolt *et al.* (1998) after thirty years (Figure 3.4). This may then represent the extent to which the surrounding bog is affected by drainage and plantation, which is achieved after roughly 30 years, although this figure was disputed elsewhere as an underestimation of up to 40 m (Lindsay 2010). Other factors, such as the poor maintenance of drains on site, may have also played a part in limiting the spread of subsidence.

In the external undrained bog, subsidence in the Pyatt 1 transect was close to being significant (Table 3.3). This may be a result of how sections of the transect were designated as 'external undrained bog'. We have used a loose definition which designates any section of transect at the edge of a stand, not enclosed on any other sides as undrained bog. While ultimately this categorisation was felt to be best way to differentiate types of bog, earlier studies on site have suggested that an alternative approach which differentiates bog adjacent to plantations may also be also valid (Shotbolt *et al.* 1998; Lindsay 2010).

In the forest stands, no significant difference was found between the intermediate (1987 and 1996) and recent measurements. This suggests that the bulk of subsidence is related to initial drainage and planting, with relatively little change thereafter. More mixed results occurred in comparisons between 1996 and 2016 in the IOG sections of Pyatt 1 and Shotbolt 2 with the former increasing in elevation and the latter decreasing. This confusing picture, which contradicts the otherwise strong effects shown in the comparisons between 1966 and 2016, may be down to the difficulty of recreating the transects on the site. While the discovery of some markers allowed these transects to be recreated closely, the high variation in microtopography could mean that an error of even a few cm could lead to a very different elevation from the original sampling point. As such, these recreated transects are perhaps not as useful as the data extrapolated from the 1966 survey maps.

The ground elevation transects sampled for this study were established with the aim of replicating the previous interim data sets, and to distribute the data collection around Bad a'Cheo to be as representative as possible, given that a truly randomised design was not feasible due primarily to high wind-throw. This design allowed meaningful comparisons to be made between the 1966 interpolated data and the 2016 measures, as well as allowing some comparisons with interim data from 1987 and 1996. As the mean change and statistical significance of each category of each elevation transect has been reported (Table 3.3) it is worth comparing the transects in greater detail and asking whether they are broadly representative of Bad a'Cheo. The peat depth transects are not included in this comparison as they included a large number of data points (425) that were distributed across a greater number of transects giving a wider coverage over the site (Table 3.2), and were analysed together rather than on an individual transect basis (Figure 3.5)

Where available, supporting data on bulk density at peat coring points on these transects is presented in chapter 5, giving an indication of the variation in peat compaction under the plantations. While much of Pyatt 1 was recreated with a high degree of accuracy, it was incomplete as wind-throw had collapsed most of the afforested sections of the transect. It is therefore representative of open bog and IOG (both the narrow 'rides' between forest blocks and wider areas of bog adjacent to forestry that are hydrologically impacted but not directly drained), but only of the edges of the forestry plantations and the less intensively afforested 'shelter belts'. Pyatt 2 was more complete but reconstructed with less accuracy and with less intermediary data, having no results from 1996. We can therefore be relatively confident of differences between 1966 and 2016, but should be more cautious in drawing comparisons from the 1987 data set. Pyatt 2 contained long sections of undrained bog, IOG and plantation, although the IOG were mainly 'rides' and not wider sections. The maintenance of drainage was mixed throughout the individual forest blocks (as it was across the site as a whole), and this transect can therefore be taken as the most broadly representative of those surveyed. Shotbolt 2 recreates a 1996 transect with a high degree of accuracy and represents a data set with low accumulated instrumental error covering a wide area of IOG, although it does not sample any undrained bog and only the edge sections of forestry. The two new transects were established to expand the cover of evaluation transects to the south of Bad a'Cheo. New 1 captured a relatively waterlogged part of the site with poor drainage ditch maintenance and lower than average tree growth (data on bulk density and tree biomass presented in chapter 5). Conversely, New 2 covered a well-drained and maintained area with relatively good tree growth (data again in chapter 5). Both New 1 and 2 contained open bog, but in both cases the IOG was again restricted to 'rides'. The two Pyatt transects and Shotbolt 2 covered relatively flat gradients, as was typical in the north and centre of the site, while the two New transects incorporated steeper inclines as was typical for the south (Figure 3.3).

The transects covered a range of slopes and included both well drained sections of forest where tree growth had been good, as well as sections where drainage ditches had not been maintained and consequently the water table was high and tree growth was poor. In addition, the transects captured both forest 'rides' and wider sections of unplanted ground further from forest edges. The results across the transects are broadly consistent (barring the anomalies discussed earlier in this section) in finding no significant change or net peat growth in the undrained bog since 1966, moderate subsidence in the IOG and strong subsidence in the forest stands. The exception to this is the IOG and forestry components of New 2, which will be discussed separately below. That the results were relatively consistent despite the variation in the transects described above suggests that drainage and afforestation, even when drainage is poorly maintained, are responsible for significant subsidence in peat. The only major component of Bad a'Cheo not adequately represented by the sampling design was the wind-thrown sections of forestry. It is an inherent problem of any survey of ground elevation and peat depth, either by transect or fully randomised, that access to the peat surface is required. Such access is impossible either because fallen trees are directly blocking the peat surface or through wider disruption caused by overturned root plates (Figure 2.4). A consideration of how wind-throw might influence estimates of losses of carbon from Bad a'Cheo is made in chapter 5.

New 2 exhibited some unusual changes. While the undrained bog and forest stand were not significantly changed, the ride was significantly higher than in 1966. Errors in the original survey map, or in the interpolation may be a factor, especially as the transect has some of the steepest gradients on the site. Much of the forest stand in this section is shelter belt (an area on plantation edges designed to protect the main forest stands from high winds), which was planted on ploughed ground but without any drains, thus differing from normal commercial stands. A lack of proper drainage would have meant that the water table remained relatively high, reducing consolidation and compression of the peat.

As with other studies examining long term subsidence, the quality of old data sets will determine the reliability of conclusions. While this analysis used GIS techniques to interpolate data from the 1966 survey, replacing work that had been done by hand in previous studies, problems remain. The original survey used relief lines to map the ground surface, but these data would be too coarse to show small scale changes in the ground topography. Peat depth can be even more variable, and the 55 m x 55 m resolution of the original survey is insufficiently high to capture fine-scale variability, nor does it provide information on accrued errors revealed by closing the loop following levelling, and as such cannot reflect the original microtopography of the site. This may be reflected in the few improbable instances of larger peat accumulation since 1966 suggested in Figure 3.4. Such difficulties in interpolation may also explain the differences in subsidence in the measurements of ground elevation and peat depth. However, considering the limitations of the older data sets available, this study represents the most thorough analysis possible. Other previous study sites on the bog were not re-surveyed. In particular Anderson *et al.* (1992), which investigated subsidence across the gaps (or 'rides') between plantation blocks, could not be repeated due to tree encroachment into these areas.

While this survey has described changes in peat depth caused by human intervention in peatlands it is important to acknowledge that there are other drivers of changes. Seasonal fluctuation may be important, in particular by mooratmung (or 'bog breathing'). Felling at the site began on March 2017, which left a small window for a campaign of data collection on site. As a result, no account was made for seasonal variation in ground levels driven by mooratmung, although the variation is unlikely to be significant (Morton & Heinemeyer 2019). Perhaps more significantly, the slow natural accumulation of peat will have continued over the fifty-year study period in the undrained bog. Due to the aerobic conditions associated with a lowered water table and a reduction in the cover of *Sphagnum* moss (the primary peat forming species) which is typical of afforested peat, this is less likely to have been a factor in the planted sections of the transect. On the undrained bog, the rate of peat formation is associated with the plant assemblage and decomposition rate (a factor which in turn is dependent on the nutrient status and water table level of the bog). There is good practical and modelled evidence that microtopography variation influences these factors, and as a result we may expect areas of hollow to accumulate peat at a greater rate than hummocks (Chaudhary *et al*. 2018). While the interpolation techniques used were sufficient to capture the major changes in peat depth and elevation caused by the drainage and afforestation, the resolution of the data may not entirely account for the natural growth. Although it is important to consider it as a possible source of variation, the resulting change in depth resulting from peat growth is likely to have been significantly smaller (millimetres over fifty years) than the drainage-mediated losses (typically tens of centimetres).Bad a'Cheo has been monitored for over fifty years, and has been the subject of studies concerning many aspects of peat bog afforestation (Ray & Schweizer 1994; Miller *et al.* 1996; Anderson *et al.* 2000). The site has been managed and records maintained by the Forestry Commission, and the archives of raw data were invaluable to completing this research. Other long-term experiments of this type are rare, and it is vital that data from such sites are properly archived and accessible so that use can be made of these invaluable resources.

The large subsidence on site and changes to the depth of the peat suggest that carbon is likely to have been lost from the system. It also suggests changes to the function and character of the bog, which may impede restoration. Work to quantify the exact nature of the change to the bog is required, as it may differ from the well reported well reported Fennoscandian drained peatland forests which often require less drainage.

Acknowledgements

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Developing supporting data sets to quantify carbon loss

Chapter two has described the mechanism through which drainage and afforestation physically changes the peat. This is most obviously manifested through ground subsidence as the peat is compacted and compressed. This subsidence may also be associated with an oxidative loss of peat carbon, which may in turn lead to a net loss of carbon from drained peat forest. Chapter three has demonstrated that such subsidence has taken place at the Bad a'Cheo plantation over fifty years, suggesting a potential loss of carbon, although this cannot be accurately quantified through ground elevation and peat thickness data alone. As subsidence is indicative of carbon loss, but is otherwise a poor proxy, an alternative method is needed to determine the change in carbon on the site.

Chapter two further established that a whole column inventory should be used in order to measure the total change in peat carbon stock. For this to be effective stratigraphic markers are required to ensure that when undrained and afforested peat columns are compared the sampled material will have accumulated over the same time period. To this end cryptotephra can be a useful tool as major deposits, most prominently Hekla 4, are widely reported throughout Scottish peat. Chapter four therefore attempts to identify Hekla 4 and other Icelandic tephra deposits at Bad a'Cheo and other afforested peatland sites in the Flow Country. The data set presented in this chapter will make possible the full carbon stock analysis reported in chapter five.

Chapter 4 - Local and regional-scale variability in tephra concentration and shard size in peatlands.

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Summary

Throughout the Holocene, microscopic layers of volcanic ash (cryptotephra) have been deposited repeatedly in peatlands, at distances of up to several thousand kilometres from source eruptions. (Crypto)tephrochronology is a valuable tool to understand the distribution of volcanic products, and to correlate and date stratigraphic records. The presence and the abundance of tephra shards from a specific eruption vary over fine spatial scales, dependent on several poorly understood pre- and post- depositional factors. Understanding these factors is important both for interpreting the stratigraphic record and for constraining models of tephra plume dispersal. In this study we compared the distribution of Hekla 4 (2310 ± 20 BCE), a widely described Holocene tephra layer, at uniquely high spatial resolution within a single peatland site and with cores from the wider region. Each peatland used in the study incorporated samples from a drained, afforested bog and from an undrained bog. ITRAX core-scanning allowed rapid tephra identification, demonstrating the value of this novel use of the ITRAX technique for thicker cryptotephra deposits. Drainage and afforestation are associated with compaction, compression and oxidative loss of peat, which was found to significantly influence tephra layers. Hekla 4 was found significantly closer to the surface in drained afforested bogs than in undrained bogs. Microtopography variation in afforested areas also significantly impacted on the depth of Hekla 4 layers, although this was not observed in undrained sites, where absolute peat depth was associated with variation in Hekla 4 depth. Shard abundance and size was not significantly influenced by the drainage status or microtopography of the peat, although shard abundance showed a high degree of variation not attributable to any measured factor. These results show that greater consideration of heterogeneity in peatlands is needed in planning tephra research, and that single deposits cannot be taken to be representative of a wide geographical area.

Keywords: peat, tephrochronology, cryptotephra, Holocene, Hekla 4

Introduction

Tephra and peatland research

Tephrochronology is an important tool in the study of the natural and archaeological history of the Holocene, and peat in particular forms a vital part of the tephra archive (Davies 2015, Watson *et al.* 2016a).

Studying the distribution of tephra is useful for correlating and dating stratigraphic records. Geochemical analyses can be used to identify distinctive characteristics of multiple tephra layers found in profiles across a region and aid in their identification and correlation (Lowe 2011). In some regions major deposits are relatively well described, with some such as AD 860 B, Hekla 4 and Lairg A widespread throughout north-west Europe (Lawson et al. 2012). As tephra from any given eruption will be deposited over a geologically short timescale, the existence of tephra in the stratigraphic record can be used to match cores to these well-dated eruptions.

Tephra also provides insight into processes that may have directly impacted peatlands. In thicker deposits, the direct environmental impacts of palaeo-volcanism can be directly inferred (Payne & Egan 2017). In peatlands proximal to an eruption (within kilometres) there are strong direct impacts on vegetation, ranging from tephra deposited directly onto leaf surfaces, to burying of whole plant layers and impacts on vegetation assemblages (Ayris & Delmelle 2012, Hughes *et al.* 2013). As peatlands become more distal, deposition of tephra may be associated with climatic changes (in the case of large eruptions), changes in hydrology and increased acidity, the release of elements through tephra leaching and in some cases direct damage to plant tissue (Payne & Blackford 2008). Even at the ultra-distal range of tephra deposition, where deposits form cryptotephra layers up to several thousand kilometres away from an eruption, environmental impacts can be inferred. One such example is the possible link between the deposition of Hekla 4 tephra and the decline of Scots Pine (*Pinus sylvestris*) in northern Scotland (Blackford *et al.* 1992, Hall *et al.* 1994, Charman *et al.* 1995).

The concentration of tephra shards can vary markedly between and within peatland sites in the same geographical region (Newton *et al.* 2007). The processes that lead to this variation are poorly understood, and within site variation has not been widely studied. Differences in shard concentration between sites can be significantly affected by numerous natural processes (Watson *et al.* 2015), although the relative importance of each is uncertain.

Processes of dispersal, deposition and reworking

Ultra-distal cryptotephra have been found globally in deposits up to thousands of kilometres away from the source volcano. Numerous factors influence the direction and distance of the spread of ash plumes. Dispersal is highly dependent on the magnitude and type of eruption, and the resulting duration and height of the ash plume (Mastin *et al.* 2009). Atmospheric conditions, in particular the prevailing wind direction, are then key drivers in determining the geographical spread of the plume, although modelling such spread remains difficult (Folch 2012). Factors including precipitation, particle size, particle shape and the interaction between particles determine how far the particles travel before deposition. With the right conditions the distances involved can be great, as shown, for instance, by recent research in Poland were tephra from both Icelandic and Alaskan eruptions has been identified (Watson *et al.* 2017a).

Local conditions may give rise to further variability during the process of deposition. Often this will be driven by wind, which may remobilise and redistribute surface tephra (Liu *et al.* 2014). Snow cover may be a major factor across many high latitude sites (Cutler *et al.* 2018). Where tephra falls onto persistent snow cover and remains exposed, wind or snow melt is more likely to redistribute material before it can settle into the peat (Blong *et al.* 2017), and so tephra variability within single sites may be greater in areas with more persistent snow cover. Conversely, the presence of vegetation may have the opposite effect, buffering tephra from wind until it is subsumed into the peat proper (Cutler *et al.* 2016a b).

Shards of tephra do not remain stationary in peat after deposition (Dugmore & Newton 1992). Subsequent factors may lead to shards being 'reworked' to different positions in the peat profile though several natural processes (Figure 4.1). Firstly, tephra will simply naturally sink through peat over time. Experimental studies have shown that while most tephra will initially settle within the top 1 cm of peat during tephra deposition, small amounts of tephra may penetrate as much a 6 cm into the peat surface over a two year period (Payne *et al.* 2005), with shards sinking as much as 15 cm over six years (Payne & Gehrels 2010). It is possible that tephra will be concentrated into narrower bands as the tephra moves through the surface vegetation and settles into the upper surface of the peat, with tephra concentrated further through future compaction as more peat is accumulated.

Secondly, hydrology acts on deposited tephra in many ways, often driven by precipitation. Peat bogs are wet landscapes, characterised by high water tables (Damman 1978, Verry 1984), where seasonal and climate-driven variation in water table depth is likely to impact the rate at which tephra is integrated into the peat column (Watson *et al.* 2015) and redistribute tephra vertically.

Flows of water across the peat surface may also lead to a lateral redistribution of tephra. This process will be partially dependent on the peat microtopography, with exposed hummocks potentially losing material to more sheltered hollows (Dugmore *et al.* 2020), a process further aided by wind.

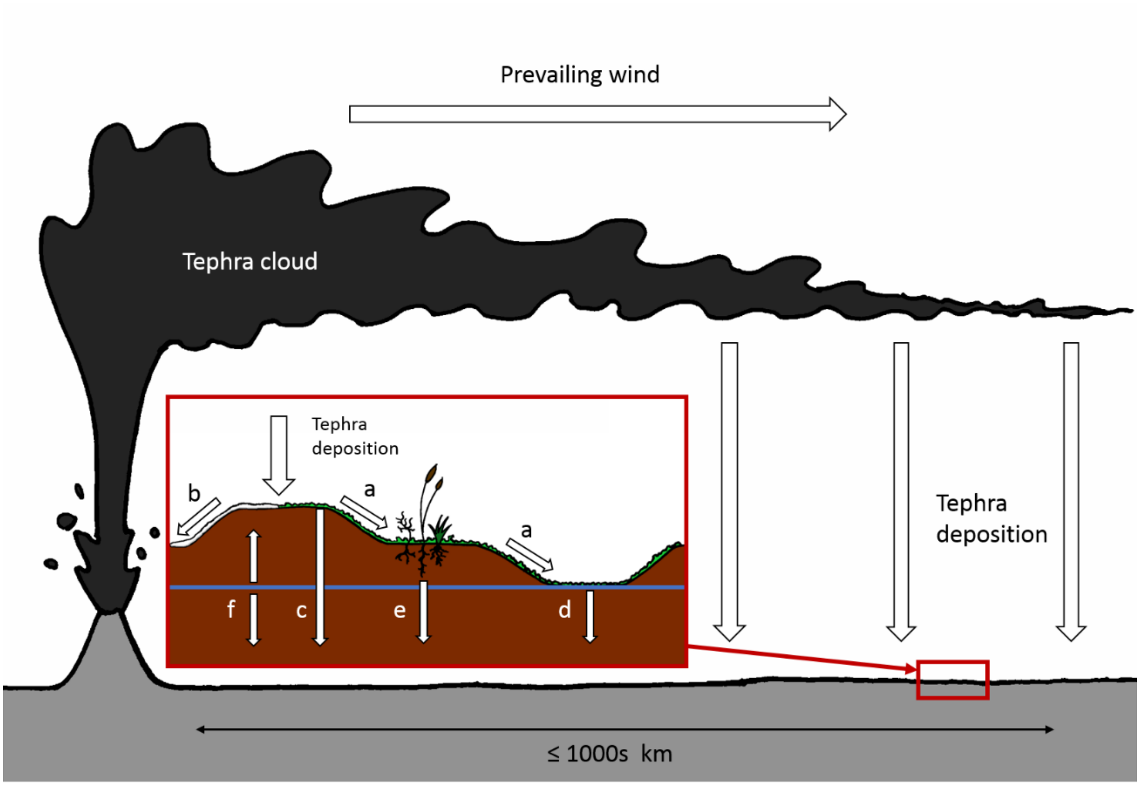

Figure 4.1. Factors affecting cryptotephra deposition and post-deposition reworking in peatlands. Cryptotephra may be dispersed up to thousands of kilometres away from an eruption, depending on the prevailing wind and particle size. Once deposited, tephra may move across the peat surface from exposed hummocks to sheltered hollows due to the action of wind and water (a). Tephra is more likely to be redistributed in this way if snow cover at high latitude sites prevents tephra from settling (b). Ultimately, tephra will naturally settle in the peat and move down the peat profile over years and as new peat forms (c), this is especially the case in sheltered parts of the microtopography such as hollows where it may have aggregated (d). The presence of vegetation on the peat may reduce distribution across the surface, and bioturbation through vegetation roots or disturbance by animals may influence the movement of tephra through the peat profile (e). Variation in the water table (blue line), which will change daily and seasonally, may also rework the tephra (f).

Thirdly, the presence of vegetation may lead to the movement of tephra through bioturbation (Figure 4.1). The direct action of developing roots on peat can act to force tephra further from the surface (Walther *et al.* 2009). To a lesser extent, bioturbation may also be caused through disturbance caused by insects or animals.

Finally, peat disturbance such as erosion, bog burst, fire, and tree fall can be important to reworking of tephra abundances after deposition. However, these stochastic disturbances are not considered as likely as other causes of tephra movement (Payne & Gehrels 2010). There is evidence that early human settlement on peatlands is associated with disturbance of tephra layers, as an increase in burning leads to erosion and therefore tephra remobilisation (Swindles *et al.* 2013, 2019). More recently, peatlands are influenced by human activities such as management for hunting, or extraction of peat for fuel (Littlewood *et al.* 2010). With peat extraction, as well as reworking, there might be areas where entire removal of tephra layers from peat profiles occur.

There is a high degree of variability in tephra layers based on pre- and post- depositional factors, as well as the human land use history, and a high degree of uncertainty on the extent of all these effects and variation between sites. Generally, studies have considered single cores at individual sampling sites (Dugmore *et al.* 1995, Rea *et al.* 2012, Watson *et al.* 2017a), and not variation within sites. Local variation in deposits of Holocene cryptotephra has not been widely examined, with limited replication mainly employed as a tool to develop methodological techniques (Dugmore & Newton 1992). Some studies have explored variation in other analogous deposits such as large macro-deposits of tephra, which vary in thickness and composition around eruption sites. Such variation is not directly applicable as the processes governing deposition (winds etc.) are not the same, and the post depositional factors on thick deposits blanketing a landscape are inherently different to those of smaller ultra-distal cryptotephra deposits. A more applicable study is that of Watson *et al.* (2015) who look at variation of cryptotephra deposits across fifteen near surface replicates on a single site. The study finds that the presence of shards is largely consistent within a single site that was free of disturbance at the time of deposition, and that shard counts vary significantly across a site, possibly due to post depositional factors. The study also demonstrates that estimates of carbon accumulation vary across a site based on these differences, although it should be noted that any estimate of carbon accumulation using upper sections of acrotelm peat should be heavily caveated (Young *et al.* 2019). Importantly only the top 50 cm (or less) of peat were examined and may still be expected to be subject to the processes of reworking described above, and how these deposits may differ after several thousand years of continuing peat formation remains unstudied. These is therefore a need for highly replicated data sets examining differences in deep peat deposits of Holocene cryptotephra, where limited replication means that the extent and drivers of variability cannot be properly assessed. Lack of such studies has been in part due to the time required to process cores, as there has been no means to rapidly identify cryptotephra (Gehrels *et al.* 2008). For many years, X-radiography has been proposed as a tool for identifying tephra isochrones (Dugmore & Newton 1992) but was not readily applicable. However, recent developments in rapid core scanning techniques using ITRAX are beginning to allow likely areas of tephra deposits to be identified through combining XRF identification of chemical elements and X-radiograph imagery (Davies *et al.* 2015, Ratcliffe *et al.* 2017).

This study aims to use the rapid identification of tephra by ITRAX core scanning to describe the variation in the depth, shard abundance and shard morphology of tephra deposits in a single intensively studied site. The site-scale variation is then contextualised within the wider region through surveys of the same tephra at other sites. As well as providing a general description of tephra variability and the efficacy of the ITRAX method for rapid identification, the study will aim to address several hypotheses. We hypothesise that a replicated study on a single site will show that the presence or absence of a large deposit such as Hekla 4 will be consistent across a whole site, but that the abundance and depth of these deposits will vary. This assumes that the tephra deposition event was large enough to blanket the landscape, and there was limited influence of confounding factors such as snow cover at the time of deposition.

We further hypothesise that variation in tephra shard abundance and the depth of deposition on undrained and non-afforested portions of the site will be influenced by microtopography-dependent redistribution. Chiefly, tephra is likely to have been redistributed down an elevation gradient from hummock to lawn to hollow following hydrological pathways, and therefore shard counts should be highest at sampling points in hollows. This assumes that where the site is undrained and undisturbed, modern microtopography is representative of past microtopography. Data gathered will not be able to comment on deposition processes beyond those tied to microtopography, as data collection for such factors was not possible within the scope of this study. For example, high humification of the type found in such deep peat deposits makes the prevalence of biotic factors (root and invertebrate action) hard and time consuming to quantify.

Variation in distribution and abundance is also expected in afforested peat but will be complicated by the obscuring of the original microtopography arising from ploughing and drainage. Ploughing in particular leads to the redistribution of up to half a metre of peat from plough furrows to ridges (Sloan *et al.* 2018) and, therefore, can be expected to influence the depth at which tephra are found. We hypothesise that in afforested peat the depth of tephra deposits will be dependent on the new microtopography, but shard abundance will not. If shard abundance is driven by microtopography at the time of deposition as hypothesised above, it is expected to be random in afforested sampling points as the original microtopography is no longer known. This also assumes that ploughing and drainage has not been deep enough to directly remobilise tephra layers, which will probably be the case for Holocene deposits like Hekla 4 which should be deposited lower than one metre depth.

Methods

Study region

The study was conducted in the Flow Country of Scotland, the northernmost mainland region of the UK and an area of extensive peatland archives and abundant cryptotephras (Figure 4.2). In the UK in the second half of the twentieth century, there was widespread afforestation of peatland areas (Warren 2000), with approximately 9 % (190,000 ha) of the UK's deep peats (> 50 cm) drained for forestry (Hargreaves *et al.* 2003). Drainage and ploughing of peat lead to compression and compaction, as well as oxidative loss of peat carbon. In addition the physical process of ploughing furrows and drainage ditches leads to disruption and redistribution of parts of the peat surface (Sloan *et al.* 2018). Such changes in peat may disrupt the paleoecological record, including tephra layers.

Many tephra layers are now well reported, dated and geochemically described (Swindles *et al.* 2011). Iceland is the primary source of many Northern European tephra (Larsen & Eiriksson 2008, Watson *et al.* 2017b), including in Scotland, which is a relatively well studied region containing numerous peat tephra deposits (Lawson et al. 2012). These include prominent layers of Hekla 4 and Glen Garry (Charman *et al.* 1995, Dugmore *et al.* 1995), which are also reported throughout the British Isles (Pilcher & Hall 1992, 1996).

Study sites

The sampling design focussed on quantifying both fine-scale (metres) spatial variability within a single site and larger scale (10s of kilometres) variability across a landscape. The main study site, Bad a'Cheo (58.430356 ° N, 3.428056 ° W) is a large area of blanket peat in central Caithness. The site covers an area of approximately 50 ha of ombrotrophic blanket bog, at an elevation of approximately 90 m above sea level with peat depths ranging between 2 and 6 m, von Post humification H 3 - 8. In 1968, almost 30 ha of the site was drained and randomised blocks of Sitka spruce (*Picea sitchensis*), Lodgepole pine (*Pinus contorta*), or a mix of the two were planted. This site is known to contain several layers of tephra, including Hekla 4 (Ratcliffe *et al.* 2018), deposited 2395-2279 BCE (Pilcher *et al.* 1995, Swindles *et al.* 2013).

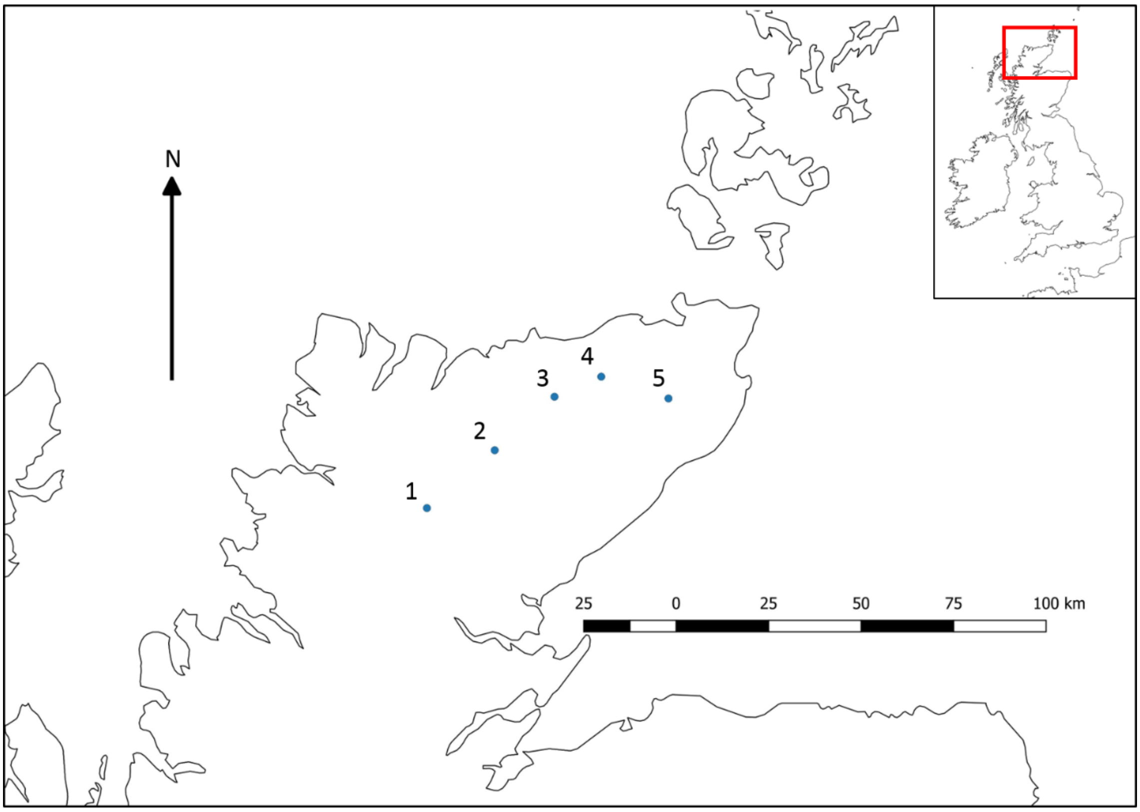

Figure 4.2. Map of Caithness and Sutherland, Scotland, showing peat sampling sites, with insert showing position in the British Isles. The sites sampled are 1. Dalchork 2. Rosal 3. Forsinard 4. Broubster and 5. Bad a'Cheo, the intensively sampled site.

Four other blanket bog sites with varying peat depth were cored (Table 4.1). These were Broubster forest (58.478883 ° N, 3.725633 ° W, 210 m above sea level, peat depth ≤ 2.8 m, von Post humification H 1 - 8), Dalchork forest (58.169150 ° N, 4.512133 ° W, 185 m above sea level, peat depth ≤ 2.6 m, von Post humification H 2 - 8), Forsinard National Nature Reserve (58.433170 ° N, 3.934040 ° W, 119 m above sea level, peat depth ≤ 2.7 m, von Post humification H 2 - 7), and Rosal plantation/Naver estate (58.303900 ° N, 4.211983 ° W, 103 m above sea level, peat depth ≤ 4.1 m, von Post humification H 2 - 8). These sites cover a variety of elevations and peat depths and all have had some form of drainage and partial afforestation.

Table 4.1. Information on Flow Country field sites used in this study.

Site name	Transects included	Location	Elevation above sea level (m)	Average peat depth (cm)		Range of von Post humification values
				Undrained	Afforested	
Bad a'Cheo	Pyatt 2, New 1, New 2	58.430356 ° N, 3.428056 ° W	90	492.4	361.4	H 3 - 8
Broubster	NA	58.478883 ° N, 3.725633 ° W	210	276	71	H 1 - 8
Dalchork	NA	58.169150 ° N, 4.512133 ° W	185	260	151	H 2 - 8
Forsinard	NA	58.433170 ° N, 3.934040 ° W	119	267	271	H 2 - 7
Rosal	NA	58.303900 ° N, 4.211983 ° W	103	333	474	H 2 - 8

Sampling methods

Cores were sampled using a 50 cm Russian corer (Aaby & Digerfeldt 1986). Each core was taken as a set of pairs which were sampled within 20cm of each other and overlapping by 5 cm.

Previous work at Bad a'Cheo had established several transects across the site, providing a supporting dataset on the extent of subsidence and thinning of peat (Sloan *et al.* 2019). These transects linked areas of drained and afforested peat with areas of undrained bog outside of the hydrological footprint of the forestry. The transects were organised to provide wide coverage across a site where random sampling was not possible due to the impassibility of some forest areas because of windfall and poor maintenance. Three of these 200 – 350 m long transects (designated Pyatt 2, New 1, New 2; Table 4.1) were used as the framework for the sampling design, with areas of undrained bog at the outer end of the established transect paired with areas of drained and planted forest further along the transect. On each transect a sampling area was designated for undrained bog, and two sampling areas for drained afforested bog to separately sample two types of forest stand; Lodgepole pine monocultures or Lodgepole pine and Sitka spruce mixed stands. Within each of these sampling areas three cores were taken to sample each of the three main classes of microtopography; hummock, lawn and hollow in the bog sections; and plough ridge, furrow and original surface in the afforested areas.

In the other four sites, paired cores were extracted within 200m of each other (one core from a lawn microform in unplanted blanket bog and one from mechanically-undisturbed original surface in afforested bog) in order to quantify between-site variability. For these paired sites, the main selection criterion was that there was an ownership boundary running between the drained afforested section and the undrained bog. This was to ensure that the characteristics of the peat were as similar as possible on each side, and that the reason for afforesting one side and not the other was a legal one rather than a practical or ecological one.

Cryptotephra identification by ITRAX

Given the number of cores under consideration, we adopted a protocol for the rapid identification of tephra deposits combining ITRAX core scanning with conventional methods. ITRAX analysis provides a novel tool for rapid core scanning (Croudace *et al.* 2006) and has shown some success for the identification of tephra layers in sediment (Peti *et al.* 2019, 2020). The use of the technique on peat to identify cryptotephra deposits has so far been limited, but where it has been attempted results are promising. Recent ITRAX studies in three separate sites in the Flow Country have allowed the identification of tephra from Hekla 4, Glen Garry and possibly Lairg A, plus several other layers that were not found in sufficient quantities to be identifiable (Ratcliffe *et al.* 2017). The study used ITRAX scanning to identify potential zones of interest and then conventional approaches were used to confirm the presence of tephra and quantify shard size and abundance.

Following extraction, cores were stored in a 4 °C cold storage room. Cores were XRF scanned using the ITRAX core scanner (Cox Analytical Systems) at the University of Manchester. The XRF was run with a 200-micron step size, 300 millisecond exposure time, 30 kV voltage, and 45 mA current. The top seven sections from each coring site (0 – 317.5 cm) were scanned, which was predicted to be sufficient to be certain to capture deposits as old as Hekla 4. XRF profiles for Al, Ca, Fe, K, Si, and Ti, which have been established as important elements in tephra identification, were examined (Kylander *et al.* 2012). The XRF profiles were examined by eye to identify distinct peaks in the data, and to identify points where two or more of these elemental peaks occured on the core. Fe was found to be particularly important in identification as the data would typically show a high peak relative to the 'noise' of the rest of the data. Where these overlapping peaks in elemental concentration were observed, the corresponding x-radiographs were studied for banding which might indicate a tephra deposit (Figure 4.3).

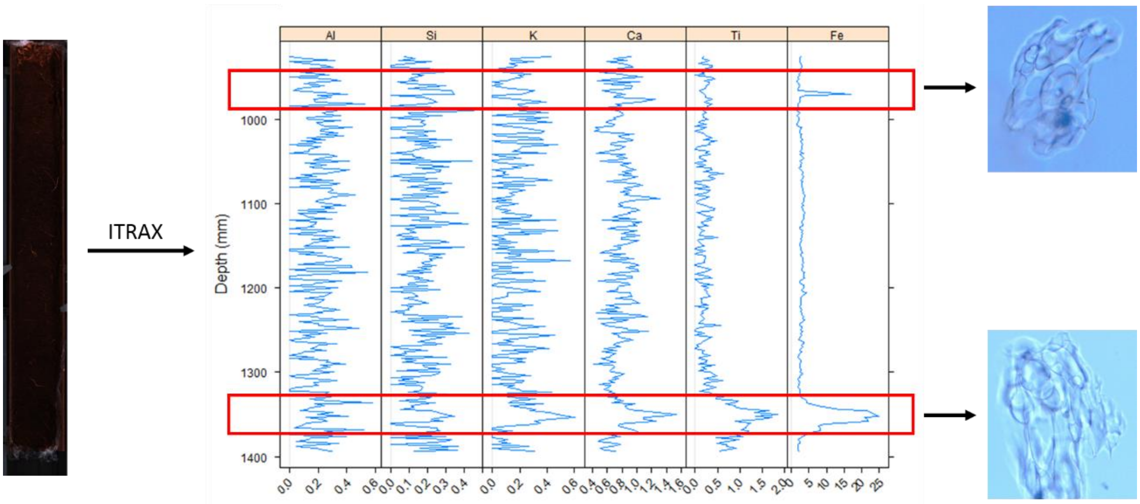

Figure 4.3. Sample core ITRAX output showing elemental peaks. Spikes in ITRAX output for key indicator elements (Al, Ca, Fe, K, Si, Ti), along with an increase in inorganic matter content indicate likely tephra locations.

Confirmation of tephra identification

Cores were divided and subsampled in 5 cm increments, discarding the top and bottom 2.5cm of each adjoining core, where the depth would overlap and there was the highest risk of contamination. Samples were dried at 105 °C in an oven, subsampled and then ashed at 550 °C in a muffle furnace for 4 hours to determine the loss on ignition (LOI). Where the core scan data and/or LOI-derived higher levels of inorganic material suggested the presence of tephra, further analysis to locate shards was carried out, as described by Gehrels *et al*. (2008) and Swindles *et* *al.* (2010). The previously ashed material was used in the initial stages of tephra analysis. In each instance the 5 cm increment corresponding to the peak was investigated, plus the adjoining 5 cm increments above and below in the profile. Samples were soaked in 10 % HCl heated to 65 °C for one hour to remove any remaining inorganic material. Where samples were contaminated with large quantities of charcoal or non-tephra inorganic material, they were disaggregated in an ultrasonic bath, then had larger fragments of material filtered out using a 180 µm mesh. Samples were mounted on a slide using Hystomount, then examined under a polarising microscope (Olympus, model BX43) to determine the presence of tephra.

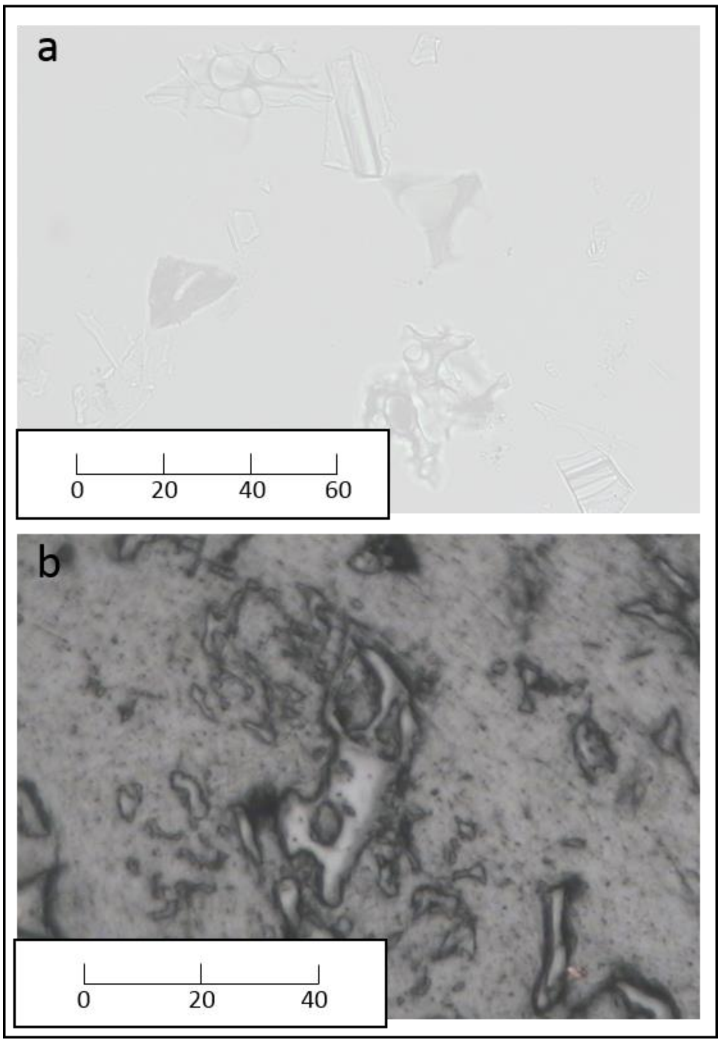

Figure 4.4. Images of Hekla 4 tephra shards sampled at Bad a'Cheo, Caithness, captured using (a) conventional microscopy and (b) a polished sample from an electron microprobe. Scale bars are in µm.

A clear deposit of tephra was found in the Bad a'Cheo cores, at around 210 cm in undrained bog and 130 cm in afforested stands. This depth was consistent with previous reports of Hekla 4 on the site (Ratcliffe *et al.* 2017). This deposit, as well as several others, were selected for identification. The cores were resampled in 5 cm increments at the same depth where the deposits had been found, and digested in heated sulphuric and nitric acid, using the 'Tephrabase' standardised method (Tephrabase 2019). They were then mounted in resin on slides, sanded, polished and carbon coated (Figure 4.4).

Geochemical analysis was performed using an electron microprobe (EMP) (Cameca SX100 wave-length dispersive spectrometer electron microprobe, University of Edinburgh). See "Supplementary material" for calibration details. The identity of tephra was determined through the composition and relative abundance of the elemental oxides in the shards. For Hekla 4, the ratio of FeO to TiO_2 is a commonly used identifier (Figure 4.5) (Lowe 2011), as is MgO to CaO for Glen Garry.

Age depth modelling

Additional confirmation of the tephra identification was provided through age depth modelling, with data points provided by radiocarbon dates. Two basal dates (for the undrained and afforested areas respectively) were produced for each of the sites using a subsample taken from the bottom of the peat column immediately before the mineral layer. Further subsamples were taken at 100 cm depth in all cores. A standardised depth of 100 cm was used as it was deep enough to not include material post-dating the industrial revolution or atomic testing. In the afforested sites it was also likely to be free of direct mechanical disruption by ploughing and to be below the likely extent of any drainage. Selecting 100 cm depth also had the advantage of giving a date close to the estimated location of the shallowest Hekla 4 samples in the afforested samples. In the 'New 1' undrained bog samples there were multiple candidate layers for Hekla 4, so two additional radiocarbon dates were taken from the peat profile.

Samples were pre-treated using the acid-base-acid method, and vertical rootlets were picked out prior to analysis. Radiocarbon analysis used the bulk carbon method (AMSDirect). Age depth modelling was completed with the BACON package in R (Blaauw & Christen 2011). The model used IntCal3 to calibrate the radiocarbon dates (Reimer *et al.* 2013). The likely eruption dates of identified tephra deposits (Pilcher *et al.* 1995) were added to this age depth model to confirm that the eruptions fitted into the likely stratigraphy of the sites.

Tephra abundance and shard size quantification

Finer scale measurements of tephra shard size and abundance were undertaken across the identified bands of tephra. A set of five 1 cm subsamples was taken around the 5 cm point of interest where tephra had been located and prepared using the method described above. Further sets of five 1 cm subsamples were taken in the adjoining section if the initial subsample was judged to have missed the deposit. Before being mounted on slides the samples were decanted into a 15 ml centrifuge tube and topped up with deionised water. *Lycopodium* tablets (batch number 3862, University of Lund) were dissolved in the tubes, and the samples were centrifuged at 3000 rpm for five minutes and rinsed with deionised water three times to remove any HCl. The remaining sample was topped up to 2 ml with deionised water, well mixed using a lab vortex, and 500 µl of the solution was then mounted on slides using Hystomount.

Slides were examined using a polarising microscope. Shards along transects of the slides were counted alongside *Lycopodium* spores. The abundance of tephra shards relative to the abundance of *Lycopodium* spores (of which a known quantity was added to the sample) was used to derive the number of tephra shards per dry weight of peat, as described by Gehrels *et al.* (2006). The ratio of tephra shards to *Lycopodium* spores was validated through a random selection of transects elsewhere on selected slides. The length of the longest axis of 100 tephra (where available) were measured and recorded along a transect down the middle of the slide (Watson *et al.* 2016b).

Results

Geochemical identification of tephra layers

Hekla 4 was widely identified across Bad a'Cheo, forming in a layer below 200 cm in the open undrained bog, and between approximately 100 and 150 cm within the drained afforested stands. Hekla 4 was identified via ratios of EMP derived elemental-oxide concentrations (Figure 4.5) (19 layers), with identification validated through age depth modelling (see "Supplementary material" for EMP data and age depth models).

Radiocarbon dates were determined for 46 peat samples (Table 4.2). The radiocarbon dates were used to produce age depth models (see "Supplementary material" for full age depth modelling). Due to a lack of suitable shards in some of the prepared slides it was not possible to obtain enough EMP analyses from some of the cryptotephra to allow their identification. In the eight tephra layers where this was the case, Hekla 4 was confirmed through a combination of age depth models and presence of an identified band of Hekla 4 in an adjacent microtopography replicate (Table 4.3). While this study primarily focuses on variation in Hekla 4, additional tephra layers were found across Bad a'Cheo (Table 4.3). This includes those which could not be identified through EMP, although age depth modelling may provide an indication of possible eruptions.

Table 4.2. Results of bulk radiocarbon analysis of peat material from specified depths. *At Bad a'Cheo a basal depth for the site was established using samples from core 3 in undrained bog and core 5 in drained afforested section.

Site name	Core ID	depth (cm)	radiocarbon age uncalibrated yrs BP	1σ error
Bad a'Cheo	Pyatt 2, Core 1	100	1472	25
	Pyatt 2, Core 2	100	1491	25
	Pyatt 2, Core 3	100	1710	29
		520*	9118	52
	Pyatt 2, Core 4	100	3964	33
	Pyatt 2, Core 5	100	3016	25
		410*	8899	49
	Pyatt 2, Core 6	100	2318	27
	Pyatt 2, Core 7	100	3498	35
	Pyatt 2, Core 8	100	3269	28
	Pyatt 2, Core 9	100	2433	26
	New 1, Core 1	100	2325	27
	New 1, Core 2	100	584	22
	New 1, Core 3	100	753	22
	New 1, Core 4	100	625	25
		236	2234	30
		315	2930	25
	New 1, Core 5	100	2347	28
	New 1, Core 6	100	1835	29
	New 1, Core 7	100	2281	30
	New 1, Core 8	100	2691	28
	New 1, Core 9	100	2258	24
	New 2, Core 1	100	1712	24
	New 2, Core 2	100	1695	27
	New 2, Core 3	100	1679	37
	New 2, Core 4	100	2772	27
	New 2, Core 5	100	2720	33
	New 2, Core 6	100	2908	29
	New 2, Core 7	100	3023	28
	New 2, Core 8	100	1705	25
	New 2, Core 9	100	3112	28
Broubster	Undrained bog	100	2750	29
		276	7263	34
	Forest	71	5149	29
Dalchork	Undrained bog	100	2725	27
		260	8180	38
	Forest	100	4157	31
		151	7875	43
Forsinard	Undrained bog	100	2340	27
		267	5293	33
	Forest	100	1054	27
		271	7632	35
Rosal	Undrained bog	100	2287	29
		333	9203	38
	Forest	100	2420	28
		474	9009	35

Distinct layers of tephra were found throughout each of the other study sites. Hekla 4 layers, confirmed via EMP as above, were found in the open bog and forest cores at Broubster and Rosal, and in the open bog core at Forsinard. In the Dalchork open bog core, insufficient EMP data points were collected to be able to positively identify a tephra layer, although the four points collected do fit the characteristics of Hekla 4 age depth modelling suggests that the layer cannot be Hekla 4. Age depth models in the Dalchork and Forsinard afforested sites, as well as the abundance and appearance of the deposits, strongly suggest that layers found here are also Hekla 4 (Table 4.3).

The EMP data for Hekla 4 deposits in the bog sites at Broubster and Forsinard included some data points which fell outside the expected values for Hekla 4 (Figure 4.5). Other supporting data including MgO to CaO ratios, age depth models and the abundance/morphology of the tephra suggest that these layers are Hekla 4 (see "Supplementary material" and Table 4.3). These readings may therefore represent instrumental error, contamination from tephra redistributed from another deposit, or samples which may represent outliers in values for Hekla 4.

At Rosal, additional layers of Glen Garry tephra were identified from the ratio of MgO to CaO (Figure 4.6), plus age depth modelling. An additional deposit of tephra in the Forsinard forest site could not be identified.

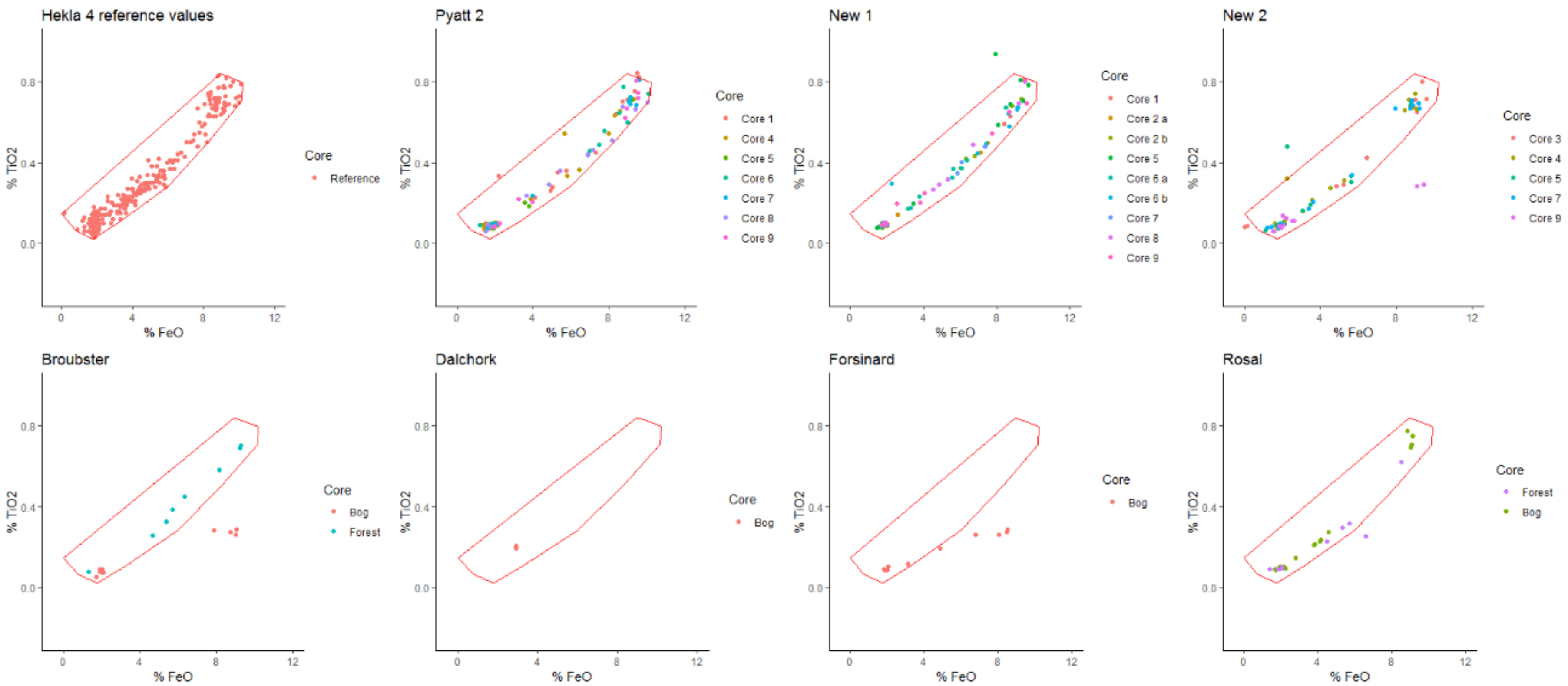

Figure 4.5. Values for percentage composition of FeO and TiO_2 in each sampled shard, a key identifier of Hekla 4. 'Hekla 4 reference values' show results from other studies in the Flow Country region (Newton *et al.* 2007, Tephrabase 2019), used to create a polygon to constrain possible Hekla 4 results in other plots. See Table 4.3 for full description of location and sampling depths of cores. Where samples are denoted 'a' or 'b' they are distinct tephra layers from the same core, with the 'a' value appearing nearer to the surface in the profile (Table 4.3). Hekla 4 was widely identified across the thee Bad a'Cheo transects (Pyatt 2, New 1, New 2). A distinct and strong layer is also found at Rosal in the 'b' layers of both the bog and forest samples. In Broubster bog and Forsinard bog the tephra was found corresponding to the age depth of Hekla 4, but the FeO to TiO_2 ratio fell outside the reference values. Subsequent analysis of CaO to MgO ratios was used to confirm the identity of Hekla 4. While some of the results for the Dalchork bog tephra are in line with the polygon, there are not enough to determine the identity of the eruption, while the forest results could not be measured.

Table 4.3. Tephra deposits found across study sites. *Where insufficient EMP measurements partial or missing, age depth modelling and the presence of an identified eruption in an adjacent microtopography replicate was used to confirm the likely eruption identity. ** In some instances EMP data provides good evidence for Hekla 4, although initial age depth estimates disagree with this by up to approx. 1000 years. *** At Dalchork open bog some data points fall within the TiO_2 to FeO ratio for Hekla 4, but there were not enough successful EMP geochemical analysis to make a firm positive identification, plus the data appears within 10 cm of the base which was dated to 9153 years calBP making it extremely unlikely to be Hekla 4.

Site name	Core ID	Type	Peak tephra depth (cm)	Number of EMP measurements	Likely eruption
Bad a'Cheo	Pyatt 2, Core 1	Bog hummock	259	-	Hekla 4 *
	Pyatt 2, Core 2	Bog hollow	255	-	Hekla 4 *
	Pyatt 2, Core 3	Bog lawn	212	19	Hekla 4
	Pyatt 2, Core 4	Forest furrow	92	18	Hekla 4
	Pyatt 2, Core 5	Forest original surface	126	15	Hekla 4
	Pyatt 2, Core 6	Forest ridge	151	12	Hekla 4
	Pyatt 2, Core 7	Forest furrow	98	9	Hekla 4
	Pyatt 2, Core 8	Forest original surface	122	13	Hekla 4
	Pyatt 2, Core 9	Forest ridge	133	10	Hekla 4
	New 1, Core 1	Forest furrow	133	2	Hekla 4 *
	New 1, Core 2	Bog hollow	a. 145	9	Unknown
			b. 183	9	Hekla 4 **
			c. 396	-	Unknown
	New 1, Core 3	Bog hummock	a. 221	-	Hekla 4 **
			b. 354	-	Unknown
	New 1, Core 4	Bog lawn	a. 237	-	Unknown
			b. 318	-	Hekla 4 **
	New 1, Core 5	Forest original surface	144	15	Hekla 4
	New 1, Core 6	Forest ridge	a. 160	10	Hekla 4
			b. 228	9	Unknown
	New 1, Core 7	Forest original surface	143	4	Hekla 4
	New 1, Core 8	Forest furrow	139	10	Hekla 4
	New 1, Core 9	Forest ridge	150	11	Hekla 4
	New 2, Core 1	Bog lawn	304	-	Hekla 4 *
	New 2, Core 2	Bog hummock	234	-	Hekla 4 *
	New 2, Core 3	Bog hollow	210	13	Hekla 4
	New 2, Core 4	Forest original surface	146	25	Hekla 4
	New 2, Core 5	Forest ridge	148	14	Hekla 4
	New 2, Core 6	Forest furrow	134	-	Hekla 4 *
	New 2, Core 7	Forest original surface	131	15	Hekla 4
	New 2, Core 8	Forest ridge	149	-	Hekla 4 *
	New 2, Core 9	Forest furrow	121	11	Hekla 4
Broubster	Open bog	Bog lawn	136	13	Hekla 4
	Forest	Forest original surface	21	10	Hekla 4
Dalchork	Open bog	Bog lawn	250	4	Hekla 4 ***
	Forest	Forest original surface	98	-	Hekla 4 *
Forsinard	Open bog	Bog lawn	189	10	Hekla 4
	Forest	Forest original surface	a. 129	-	
			b. 171	-	Hekla 4 *
Rosal	Open bog	Bog lawn	a. 84	13	Glen Garry
			b. 187	23	Hekla 4
	Forest	Forest original surface	a. 74	10	Glen Garry
			b. 154	10	Hekla 4

Variation in depth

The depth at which Hekla 4 was deposited varied within Bad a'Cheo and between all the sites. At Bad a'Cheo, Hekla 4 layers were located deeper in the undrained bog than in drained afforested peat ($n = 27$, $p < 0.001$), with an average depth of peak shard abundance of 244.0 ± 44.6 cm in undrained bog and 134.4 ± 17.9 cm in the afforested bog. There was no significant difference between the two forestry types, which had an average depth of 134.8 ± 16.3 cm and

134.1 ± 20.3 cm for Lodgepole and mixed stands respectively ($n = 18$, $Z = - 0.088$, $p = 0.931$). ANOVA tests (IBM SPSS, Version 26) showed that there was also a significant difference between the treatments in the depth of the deposits relative to the total depth of peat, averaging 49.87 % in undrained bog and 38.24 % in afforested bog ($n = 27$, $F = 4.937$, $p = 0.016$).

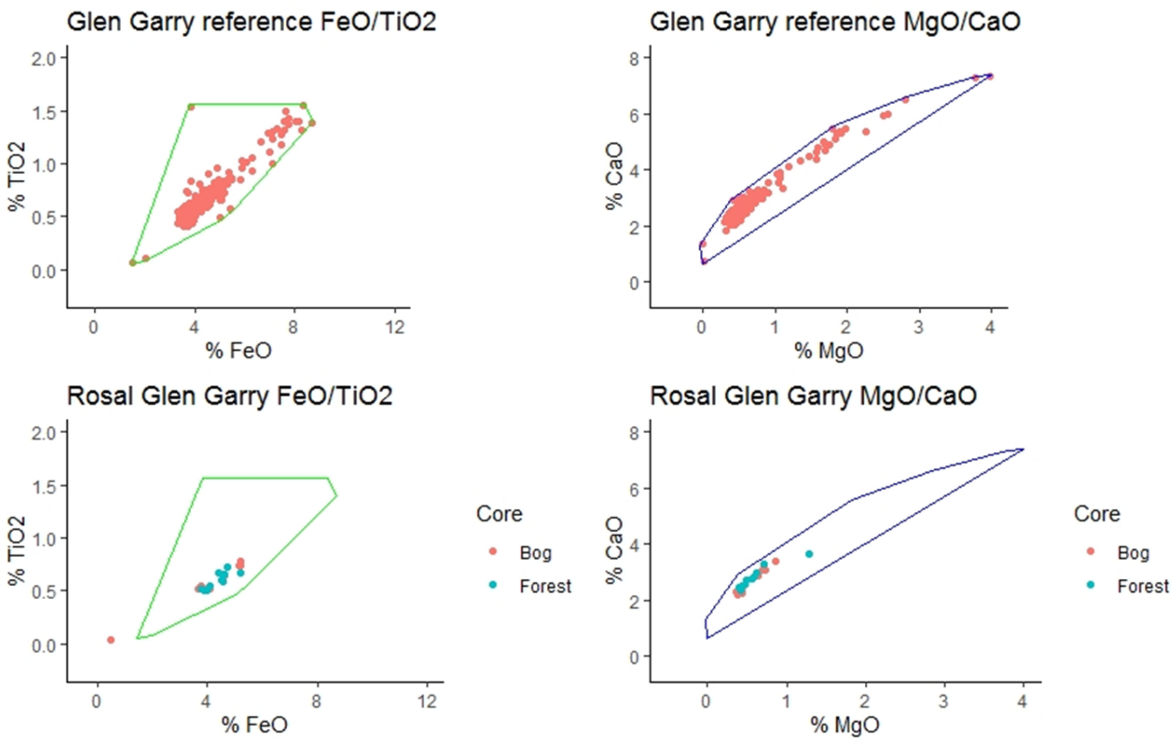

Figure 4.6 Values for percentage composition of FeO and TiO_2 and for MgO and CaO in tephra shards measured by EMP. 'Glen Garry reference' values show results from other studies in the Flow Country region (Newton *et al.* 2007, Tephrabase 2019), used to create a polygon to constrain possible Glen Garry results. These polygons are then used to constrain elemental results of a tephra deposit found at Rosal. See Table 4.3 for full description of location and sampling depths of cores from Rosal.

No significant difference was found on tephra depth between the three undrained sampling locations on Bad a'Cheo ($n = 9$, $F = 0.649$, $p = 0.556$) or based on microtopography in each of those sites ($n = 9$, $F = 0.533$, $p = 0.612$). The average range of values between the three microtopography replicates at a single sampling site was 39.89 cm. The depth of the Hekla 4 layer between the nine sampling points in the single bog ranged between 183.3 cm and 318 cm (Figure 4.7). Within the afforested areas, where the microtopography is more dependent on direct human influence, there was significant variation in Hekla 4 depth between the three transects ($n = 18$, $F = 4.113$, $p = 0.038$), and across the different microtopographies ($n = 18$, $F = 6.529$, $p = 0.009$).

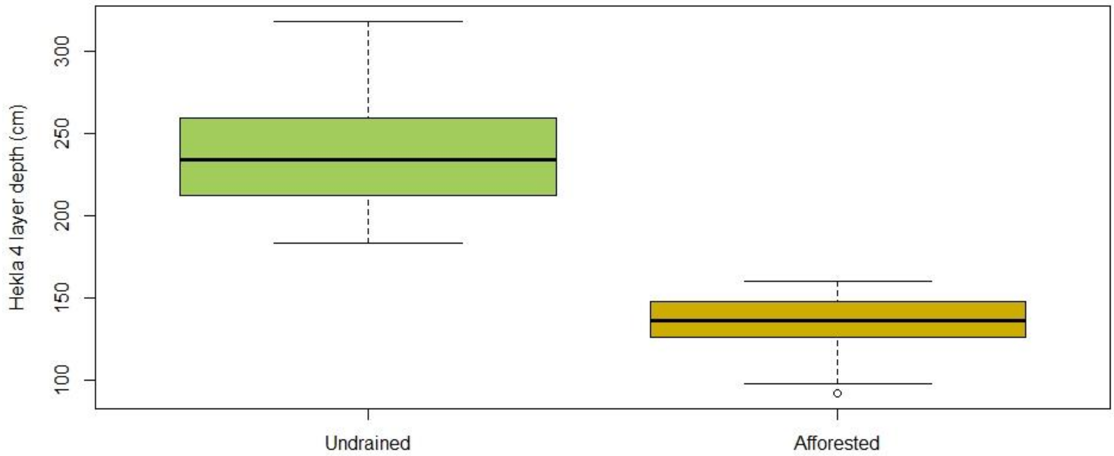

Figure 4.7. The spread of depths of Hekla 4 layers across the sampling points in undrained and afforested bog at Bad a'Cheo forest.

As the range of values for the depth at which Hekla 4 was found at Bad a'Cheo was large, and due to the significant differences between the microtopographies in the afforested samples, the three groups of Bad a'Cheo samples (Pyatt 2, New 1, and New 2) were compared individually to the other sites. In each case, the lawn undrained bog sample and an average of the two afforested original surface samples were used to make direct comparisons, as this replicates the sampling strategy used at the additional sites. Across all sites, the depth at which Hekla 4 was found was driven by the drainage and afforestation status of the peat, with the undrained open bog found significantly deeper in the profile than where the peat had been drained and afforested ($n = 6$, $Z = -2.201$, $p = 0.028$). Across all sampling points, the depth of Hekla 4 was strongly significantly correlated with the total depth of peat ($n = 34$, $r = 0.691$, $p < 0.001$) (Figure 4.8).

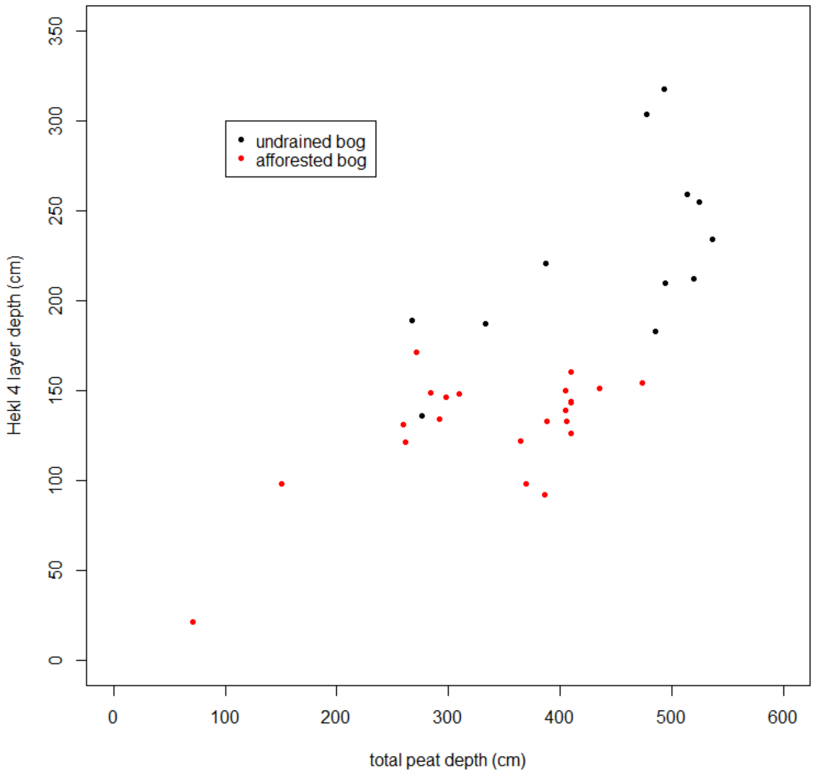

Figure 4.8. The depth of Hekla 4 deposits relative to the total depth of the peat at each sampling point.

Variation in shard abundance and size

Hekla 4 is one of the most abundant layers of cryptotephra in the peat bogs of the region, and this is reflected in several high shard counts, averaging 2220 shards per mg of dry peat. Shard abundance was highly variable within Bad a'Cheo and between sites (Figure 4.9), and this variation did not appear to be systematic. Shard abundance was not significantly different between undrained bogs and afforested bogs in Bad a'Cheo (n = 27, F = 0.304, p = 0.740). There was no significant variation in the shard counts based on microtopography in either the undrained bogs (n = 9, Chi squared = 0.622, p = 0.733) or in afforested stands (n = 18, F= 0.304, p = 0.740) (Mann-Whitney U tests and ANOVA tests respectively; Figure 4.10). Across the other sites, shard abundance was not significantly different between undrained and afforested bogs (n = 6, Z = - 0.314, p = 0.753).

The shard size along the longest axis of tephra from the Hekla 4 deposits did not vary significantly between the undrained bog and either of the forest treatments (n = 25, F = 0.097, p = 0.908). Average shard size at Bad a'Cheo was 52.69 µm and 53.35 µm in undrained bog and afforested bog respectively. Shard size was not significantly different between undrained and afforested bogs across the study sites (n = 6, Z = - 0.135, p = 0.893).

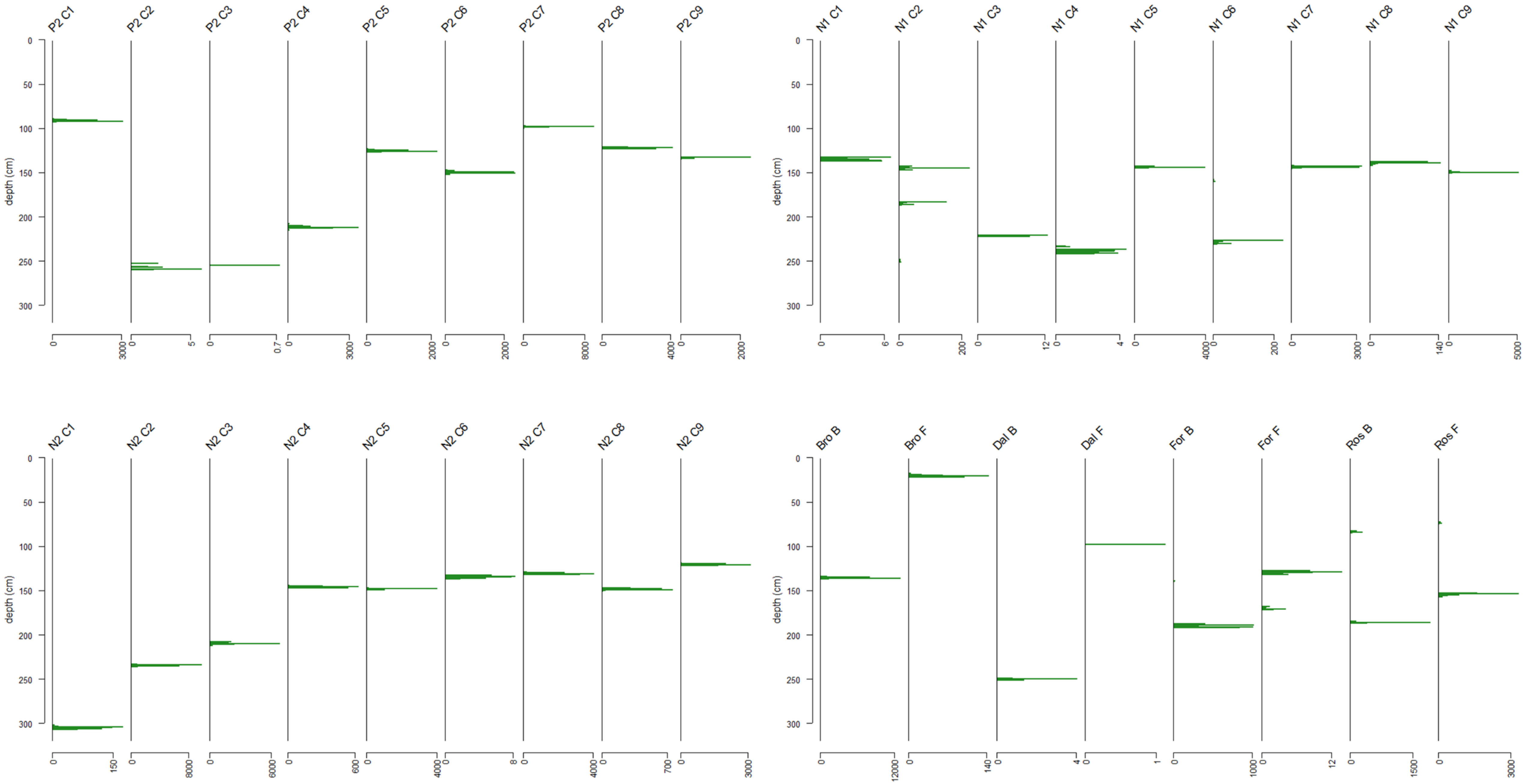

Figure 4.9. Tephra shard abundances in surveyed cores. Shards are shown as per mg of dry weight of peat, derived using Lycopodium.

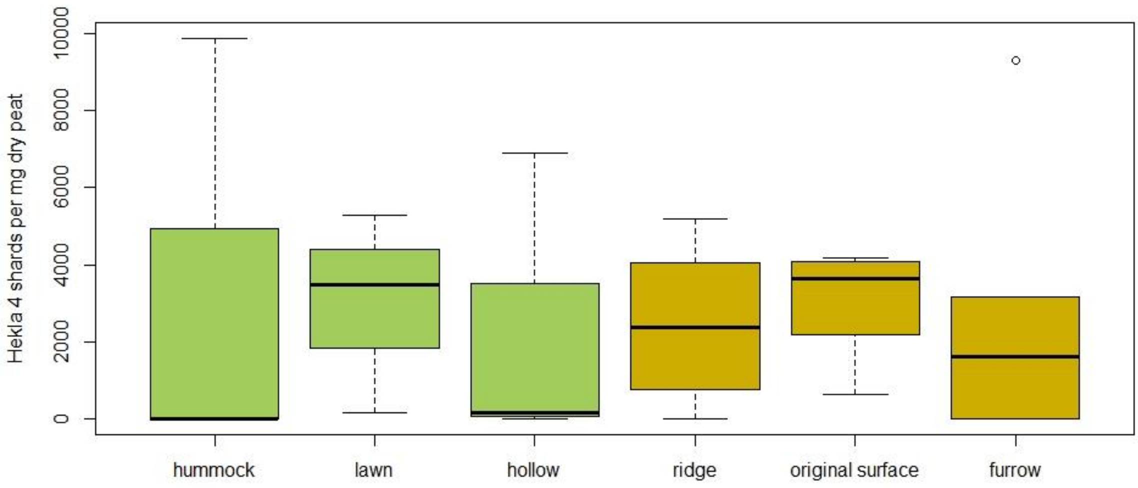

Figure 4.10. The spread of counts of shards (shards per mg dry peat) for each of the microtopography replicates at Bad a'Cheo. The green bar charts on the left are microtopography features of the undrained bog, while the yellow graphs on the right are those of the drained and afforested sections.

Discussion

Variation in tephra distribution and abundance in the North of Scotland

The Bad a'Cheo bog contains a substantial deposit of Hekla 4 throughout, as well as numerous other layers of cryptotephra. The recorded deposits of Hekla 4 from two previous samples taken in the north of Bad a'Cheo broadly conform to the results from Pyatt 2 (Ratcliffe *et al.* 2017), but they diverge strongly from other transects to the south of the site.

Several hypotheses were put forward. Firstly, it was predicted that a large cryptotephra deposit such as Hekla 4 would be found consistently across a single site (i.e. would be uniformly present or absent in all sampled cores). The data presented here supports this hypothesis, and therefore suggests that larger cryptotephra deposits are reliable stratigraphic markers in peat. It was also hypothesised that the depth of the deposits and the abundance of the shards would be related to microtopography in the undrained bog, which was not the case. No correlation between microtopography and either shard depth or abundance was found on the undrained bogs. Finally, it was hypothesised that the depth of tephra deposits in drained and afforested sections would be strongly driven by microtopography, although abundances would be random. This was found to be the case.

Microtopography in undrained sections did not influence the position of the Hekla 4 layer as predicted, while the much more pronounced human-influenced microtopography of the afforested sections was a significant factor. Indeed, the main driver of the depth of the deposit was the drainage status of the peat. The depth at which Hekla 4 was found varied markedly across the site, with deposits in drained, afforested bogs found significantly closer to the surface than those in undrained sections of the bog. Across the wider sites a similar pattern was observed, with afforested sites having significantly shallower deposits of Hekla 4 than undrained sites.

The differences in the depth at which Hekla 4 deposits were found are probably attributable to the disturbance in the upper layers of peat from ploughing techniques used in afforestation and the shrinkage of the peat caused by drainage. Previous study has shown that peat drained for forestry on Bad a'Cheo is on average 56.8 cm or 13 % shallower than comparable open bog peat due to compression and compaction (Sloan *et al.* 2019). The differences in the depth of Hekla 4 is consistent with this loss, and the redistribution of the peat surface into ridges involved in afforestation. During planting, material up to half a metre deep is removed from plough furrows and deposited as adjacent ridges. Consequently, furrow cores are likely to be incomplete, and ridge cores will contain up to 50 cm of redeposited material which is out of sequence. The accuracy of any age depth curve on such disturbed ground is therefore limited, at least above the first verified stratigraphic marker. In this case the age depth modelling was accurate enough to confirm the approximate ages of the tephra deposits but should be treated with caution in the in the upper metre of the ridge and furrow samples of the afforested sites (see "Supplementary materials").

The ploughing and drainage required for successful deep peat afforestation in the UK produces the characteristic microtopography captured by this study. Because this is a well-established feature of any afforested site, and was understood to be a possible source of variation, this study was designed to use replicates to capture and describe it. This underlines the importance of awareness of the contemporary use and land-use history of a site when carrying out paleoecological or archaeological research (Lowe & Walker 2014). This is especially important in the context of restoration work now underway on UK peatlands, including afforested bogs, which is likely to erase evidence of current microtopography and the human influence that has shaped them. The disturbance in such bogs prior to restoration will make them relatively poor sources of paleoecological evidence in the future, especially within the disrupted top metre of peat.

The depth of the Hekla 4 deposit was strongly correlated with total peat depth. The high heterogeneity of peatlands such as Bad a'Cheo means that within a single group of replicates (taken within a few metres), the depth of cryptotephra deposits vary markedly. There may be a difference of up to 94 cm in the depth of peak shard concentration between deposits less than 2 m apart. Current day microtopography in undrained bogs was not a strong predictor of Hekla 4 depth, or of total peat depth. The effect of microtopography is however observed where ground level has been manipulated by drainage or by the addition or removal of peat by ploughing.

Shard abundance was not influenced by microtopography, and the very high levels of variation seen across all the sites seems to have been random rather than systematic. This is in line with results reported for more recent, near surface cryptotephra deposits (Watson *et al.* 2015). Abundance of shards varied throughout all samples, while shard size remained relatively constant throughout. There seemed to be no pattern to the variation in shard abundance. This may represent a true variation in tephra fall from an eruption, or the reworking of layers through post-depositional factors (Dugmore *et al.* 2020). It may also suggest that modern microtopography does not accurately represent the topography at the time of deposition, and that environmental factors working in the approximately 4000 years since the deposition of Hekla 4 layer may have altered the surface topography. There may also be a slight bias in the *Lycopodium* method used to quantify tephra. It is a well-established problem that under very high abundances, the *Lycopodium* spores may be underrepresented, inflating counts to be higher than is truly representative (Maher 1981). Overall it is indicative of an erratic spatial distribution of tephra deposits. A similar erratic distribution was found by Watson *et al*. (2015) in their survey of near-surface deposits at Fallahogy in Northern Ireland. The within site variation of both the depth of the deposits and the shard counts at Bad a'Cheo was greater than that observed at Fallahogy, possibly as a result of the nature of the deposits. At Fallahogy, only the top 50 cm (or less) of peat was analysed, and the deposits are of the order of hundreds of shards per dry mg of peat at its most abundant. At Bad a'Cheo, a much deeper deposit was studied, which would be subject to centuries more post-depositional effects, with abundances that were up to thousands of shards per mg dry peat.

The shallowest deposit of tephra at Bad a'Cheo was found at 92 cm depth. This is deep enough to not be affected by the direct mechanical disruption associated with afforestation. Even in the best drained sites, a depth of around 1m is generally still below the water table, and in Bad a'Cheo and many other afforested peatlands drainage ditches were not maintained over the life of the plantation. Hekla 4 layers are therefore less prone to new reworking centuries or millennia after the original deposition, that would arise from human driven changes to the water table. Only the severely diminished peat of the Broubster deposit, at 21 cm, would be likely to be subject to such direct hydrological influence as the tephra was close enough to the surface following afforestation to be within the zone of flux of the drained water table. The average Hekla 4 shard abundance at the Broubster plantation was 145 per mg dried peat, below the 2220 per mg dried peat average (although not the lowest reported). Although such thinning of the peat was not typical for deep peat plantations, more tephra abundance data is needed on such severely denuded peats where the impact on the paleoecological record may be most severe. It seems likely that only the most recent deposits (within the last ca. 1000 years) are likely to be highly disrupted by these changes.

Tephra shard size showed little variation throughout the samples. This is aligned with the dimensions of cryptotephra Hekla 4 layers reported elsewhere (Watson *et al.* 2016b). It may also reflect some loss of the smallest fragments of shards (those which could pass through the 25µm mesh) during the preparation of slides. It is however unlikely that such small fragments could be positively identified as tephra and would typically not contribute to shard abundance or size measurements.

Advantages of ITRAX as a tool for tephrochronology

Traditionally, analysis of cryptotephra has been limited by the time and expense of identifying deposits, which either involved processing an entire core, or by relying solely on proxies such as LOI. The development of rapid core scanning techniques using ITRAX have massively increased the potential to identify tephra layers more rapidly than has previously been possible. In the 35 cores scanned for this study, ITRAX failed to identify a likely location for Hekla 4 in only one core, the Dalchork Bog. The use of ITRAX reduced the time required to identify Hekla 4, and made the level of replication possible. LOI data was included to support the identification of tephra deposits, but the reliability of the ITRAX scanning would have probably supported the identification of Hekla 4 even without this source of supporting data. While the addition of LOI increased the overall analysis time, the method adapted remained significantly quicker than a LOI only approach, which typically requires the time consuming chemical processing and microscope analysis (as described in the "Confirmation of tephra identification" section) of large numbers of ashed samples. The use of XRF data derived from ITRAX core scanning allows for a more highly targeted approach on zones of potential deposition, with attendant gains in efficiency.

This study demonstrates the viability of bulk scanning large numbers of cores, but the methodology could be streamlined further. For this study, peaks in the data were identified by eye. The process could be enhanced by the development of signal to noise ratio techniques to automate the identification of likely peaks. The significant 'noise' in the data (Figure 4.3) means that such an approach would be feasible for large and distinctive deposits such as Hekla 4, but may miss other, smaller deposits. A mixed approach combining all available data, including LOI values and x-ray chromatograph images, therefore continues to be necessary.

Due to the time required to identify tephra in a single core, multiple replicates from single sites are rarely taken. The high spatial variation of peat depth over short distances in peatlands surveyed in this study shows that single tephra samples may not be representative of deposits throughout a peatland. A single core taken in an undrained bog would be likely to give a misleading impression of both tephra depth and shard abundance. The standard deviation of the depth to Hekla 4 of cores sampled in the undrained bog at Bad a'Cheo was 44.6 cm, against a mean depth for the site of 244 cm. This means that any single core taken at Bad a'Cheo could differ by almost half a metre in the depth of Hekla 4, and the data also shows that the shard count could differ by an order of magnitude. The consequences of such unpredictable variation in the depth of Hekla 4 when used as a stratigraphic marker for the quantification of carbon stocks may be significant, as will be explored in the next chapter. As the variation did not correspond to the microtopography in the undrained bog, it is difficult to devise a sampling strategy that would identify a representative single core without some form of additional surveying. Overall peat depth was however found to be strongly correlated with the Hekla 4 deposit depth, suggesting that a peat depth survey should form an important part of any experimental design which does not feature site level replication. At a minimum, depth probing around sampling areas should be used prior to core extraction to find a representative location and to quantify the heterogeneity of the peat depth. Beyond this, replication in paleoenvironmental work should be used more routinely, and the increasing availability and proven efficacy of ITRAX scanning will make this possible.

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Using cryptotephra to quantify carbon stock

Chapter three has shown that subsidence and a thinning of peat had taken place at Bad a'Cheo over the course of a fifty-year forestry rotation. The summary of methods for establishing the change in carbon storage in peatlands in chapter two has established that while subsidence a is an indicator of possible carbon loss it cannot be used to produce an accurate estimate of that loss. In order to reliably quantify the change in carbon stored in peat over the whole forestry rotation, a stock-based approach is required.

Stratigraphic markers are required for an assessment of carbon stock to be successful. Chapter four has now reliably established the presence of a Hekla 4 cryptotephra deposit throughout the Bad a'Cheo study site, and at four other afforested deep peat sites across the Flow Country. This marker will be used in chapter five to quantify the changes in peat carbon after afforestation. This analysis will be complimented by an assessment of the carbon accumulated in tree biomass, which may go some way to offsetting any losses from peat. This will ultimately provide a total carbon budget for Bad a'Cheo, and an indication on how these changes reflect those of other afforested peatlands in the region, addressing the key project aims set out in chapter one.

Chapter 5 - A stock-based approach to quantifying carbon accumulation and loss in UK afforested peatlands

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Summary

The afforestation of large areas of blanket bog in the UK and Ireland between the 1950s and 1980s has potentially released large amounts of stored peat carbon. There is uncertainty about whether this has been offset by carbon accumulation in trees. The tax incentives that led to the afforestation of previously open blanket bogs have created plantations that are different to other well-studied sites globally. As the first forestry cycle is in many places coming to an end, decisions must urgently be made about whether to restore or restock these bogs – and if so, how. The existing evidence base is insufficient to predict the magnitude of the net carbon balance on these sites.

To fill this gap, we quantified above and below-ground carbon stocks along transects running from inside to outside an afforested peatland to determine net changes in ecosystem carbon stock. Further paired cores from other afforested bogs in the region were used to corroborate the peat loss data. Microscopic layers of volcanic ash (cryptotephra) were used as precise stratigraphic age-markers to compare peat carbon stock between open and afforested locations and to estimate potential carbon losses. Above and below-ground tree biomass and carbon content were estimated in felled trees and by morphometric surveys.. In a single 21.67 ha plantation, the stock of carbon accumulated following the deposit of Hekla 4 in the afforested peat was estimated to be approximately 1450 tonnes of carbon (t C) less than in comparable undrained peat. Tree biomass uptake was estimated to have been approximately 1024 t C, although the fate of this carbon is uncertain, producing a potential net loss over the fifty-year forestry rotation of approximately 426 t C. Although there is high variation in the data that

makes the range of these findings uncertain, afforestation of peatlands is potentially a significant source of carbon emissions.

Keywords: Peat, forestry, carbon stock, tephra

Introduction

The global soil carbon pool is significant but disproportionally distributed, with peatlands estimated to store between 400 and 600 Gt globally (Gorham 1991, Yu 2011), or 12-24% of the estimated global soil carbon stock (Tifafi *et al.* 2018), in approximately 3% of the land area. Peatlands are characterised by slow accumulations of biomass in anaerobic conditions. Land management across the world has led to the degradation of peat, and the rapid loss of stored carbon to the atmosphere (Joosten 2010, Houghton *et al*. 2012, Page & Baird 2016). Recent estimates suggest that approximately 60 years ago, the global peatland biome switched from a net sink into a net source of greenhouse gases (GHGs) (Leifeld *et al.* 2019). Of the increasing land use pressures on peatlands, drainage for forestry is widespread and in many instances, the consequences to carbon stocks are not yet fully understood (Sloan *et al.* 2018).

In the second half of the twentieth century, technological developments led to increasing afforestation of peatland areas in western Europe and Fennoscandia (Wood 1974). In the United Kingdom (UK), tax incentives contributed to the afforestation of bog areas that were not commonly used for forestry elsewhere in Europe (Warren 2000). Over 800,000 ha (circa 20%) and 200,000 ha (circa 16%) of peatland had been drained and planted with non-native conifers in the UK (Hargreaves *et al.* 2003, Artz *et al.* 2014, Payne & Jessop 2018) and in Ireland (Farrell 1990, Renou & Farrell 2005), respectively. The policy of afforestation became increasingly controversial, with initial opposition focused on the ecological disruption, particularly to ground nesting birds and other endemic species (Thompson 1987). In subsequent years, an increasing awareness of climate change has highlighted the importance of peatlands to carbon storage (Dise & Phoenix 2011).

Human modification of bogs, in particular drainage, leads to carbon losses through a variety of processes (Swindles *et al.* 2019). The drainage and ploughing of previously open bogs required for successful afforestation in the British Isles leads to compression and compaction of the peat, causing ground level subsidence (Anderson 2010) and ultimately, if the water table gets drawn down sufficiently, peat cracking (Pyatt & John 1989). As the water table lowers, more peat is exposed to oxidative loss of peat carbon through respiration (Eggelsmann 1975). This respiration will increase production of CO_2, although less CH_4 will be emitted (Drosler *et al.* 2008, Hermans *et al.* 2019). While the picture is complicated by the higher global warming potential (GWP) of CH_4 than CO_2, the overall increase in total carbon efflux from bogs is likely to have a net radiative forcing effect (Martikainen *et al.* 1995). In addition, the physical process of ploughing furrows and drainage ditches leads to the removal and redistribution of parts of the peat surface (Anderson *et al.* 2000), the creation of artificial microtopographic gradients, and the disruption of existing vegetation assemblages. Drainage is associated with the export of dissolved and particulate aquatic carbon which then have several pathways into the atmosphere (Freeman *et al.* 2001, Billett *et al*. 2007, Dinsmore *et al*. 2010). The closing of the forest canopy progressively increases interception of rainwater and lowers the water table further (Cummins & Farrell 2003), which in turn leads to the loss of peatland specialists plant communities and a species-poor understorey dominated by needle litter (Hancock *et al.* 2018). A changed albedo primarily associated with differences in snow cover may also have effects that have not been fully quantified, but are likely to be an additional positive driver of climate warming (Betts 2000, Lohila *et al.* 2010).

Carbon accumulation in developing tree biomass and litter may be expected to counter the loss of peat carbon to the atmosphere. The balance between the loss from peat and the uptake into wood biomass will determine the net change to stored carbon (Friggens *et al*. 2020). In commercial plantations on mineral soils, the production of high yields of good quality hard wood for construction or other such uses may 'lock up' carbon in tree biomass for centuries, producing net carbon uptake even after a plantation is felled (Lal 2005, Clemmensen *et al*. 2013). However, the driving force for much of the afforestation of UK blanket bogs was tax incentives rather than the expectation of a particularly productive crop (Oosthoek 2013). The relatively high water table and low nutrient content of drained peat bog plantations, if not adequately tempered by maintenance of the drainage system and fertilisation, could give rise to low yields of wood.

The complex mix of counterbalancing effects, which vary in intensity over time, means that there is uncertainty about the net balance of carbon flux of afforested bogs. The evidence base for whether peatland forestry produces a net uptake or release of carbon within the British Isles remains small. There are only a few studies that measure carbon directly (Hargreaves *et al.* 2003, Byrne & Farrell 2005, Yamulki *et al.* 2013), and concerns about the methodology of some of these studies have been reported (Artz *et al.* 2013, Sloan *et al.* 2018). A synthesis of results reported from the UK and climatically similar parts of north-west Europe (Evans *et al.* 2019), including both flux-based and stock-based approaches and incorporating the CO_2-equivalent emissions of N_2O and CH_4, estimated the average emission from afforested peat to be 9.91 tCO_2e ha^{-1} yr^{-1} (against an estimated net emission of 0.01 tCO_2e ha^{-1} yr^{-1} and sequestration of 0.61 tCO_2e ha^{-1} yr^{-1} in near natural bogs and fens respectively). This figure was for the peat itself and was not net of uptake into the trees. Due to this paucity of evidence, synthesis papers in the British Isles often use data from Fennoscandian and European studies, with their inherent inapplicability to British Isles peatlands, often suggesting a net increase in carbon storage under afforestation as a result (Cannell 1999). Indeed, studies from Fennoscandia may be of limited relevance to the British Isles where environmental conditions are different. For example, these studies include the afforestation of already treed fens which require less drainage than naturally treeless British bogs (Laiho & Laine 1997, Minkkinen *et al.* 2002, Laine *et al.* 2009).

An additional concern for determining changes in British Isles peatland carbon stocks is the reliance on carbon flux measurements to quantify net uptake and loss (Sloan *et al.* 2018). Such methods are vital to understanding the pathways through which carbon leaves the peat but will only provide a snapshot of carbon flux over relatively short periods. As full forestry rotations are measured in decades, flux studies may not give a true indication of net losses and gains of carbon over the life of a plantation. These approaches may also not consider other paths of carbon to the atmosphere, such as loss of particulate or dissolved organic carbon in water. Importantly, flux-based evaluations are not available for the early stages of afforestation, where losses of carbon to the atmosphere and the watercourses may have been the highest. A stock-based assessment of carbon differs in that it assesses the total carbon in a column of peat through means of bulk density measurements and elemental analysis. Such an approach, which integrates past losses, is therefore more appropriate to describe the net change in carbon storage (Pitkanen *et al.* 2013). While a stock-based approach cannot give the detailed information on the pathways through which carbon enters and leaves a bog, particularly the relative amounts of CO_2 and CH_4, it does give a complete picture of the net change in carbon storage over the life of a plantation.

Significant peatland afforestation in the UK began in the 1960s and was at its peak in the 1980s, until changes in the British tax system removed the incentive to plant in such marginal areas, which led to a sharp decline in the practice (Stroud *et al.* 2015). Restoration initiatives driven by biodiversity and changes in policy (Forestry Commission Scotland 2015) have already led to large scale removal of conifers on deep peat in areas adjacent to designated blanket bog habitats for conservation. However, with many of the initial plantations now reaching the end of the first rotation, decisions must be taken as to whether areas should be restored as open peatland or re-stocked, and whether these restocked forests should be new commercial conifer plantations or a mixed woodland of native species. With the current emphasis on afforestation as a strategy for climate change mitigation (Bastin *et al.* 2019) and associated political pledges towards large-scale tree planting, and with policies supporting both increased forest cover and peatland restoration in the UK, this is a critical issue for UK carbon management, for which robust data on the overall carbon impact of afforestation and restoration are urgently needed.

This study aims to use a stock-based approach to present a total carbon budget of a single, intensively studied afforested site, and to put this site into a regional context using measurements at other plantations. Specifically, this study aims to produce a highly replicated peat carbon budget for an intensively studied site. It does this by using a well reported tephra deposit as an easily replicable stratigraphic marker. It pairs this below ground carbon budget with series of tree biomass surveys to give an overall carbon budget for the site. Finally, a series of paired coring surveys are used to contextualise the finding of the intensive survey.

Methods

The total carbon budget was produced through direct measurement of carbon content within afforested bogs and undrained adjacent bogs, using a uniquely large sample size and level of replication, and through using the novel ITRAX technique for rapid identification of tephra isochrones (Kylander *et al.* 2012). This is coupled with tree carbon measurements via direct sampling.

Study sites

Our study focusses on the Flow Country of northern Scotland, Europe's largest blanket bog. The Flow Country covers an estimated 400,000 ha dominated by blanket bog habitat, of which around 67,000 ha was drained and afforested between the 1960s and 1980s (Stroud *et al.* 1987, Lindsay *et al.* 1988). Five afforested peat plantations were surveyed for this study along with an adjacent open blanket bog area (Figure 5.1, Table 5.1). The five sites were chosen because the boundary between undrained and afforested areas was an ownership boundary (i.e. one landowner gave over land to planting and the adjacent did not) rather than a catchment boundary. This was important to support the assumption that the initial pre-drainage physical properties (including areal carbon density) and carbon accumulation rate of the peat would have been similar and that therefore the comparison between the afforested and open area was meaningful.

For the intensive study, we use the Bad a'Cheo site, a 50 ha area of blanket bog at approximately 90 m elevation above sea level. The site was drained and planted in 1968 with randomised blocks of Lodgepole pine (*Pinus contorta*), Sitka spruce (*Picea sitchensis*) or mixed stands of Lodgepole pine and Sitka spruce. By 2016, peat had subsided by up to half a meter from interpolated 1966 values under forest stands, or 13 % of total depth (Sloan *et al.* 2019), suggesting some combination of peat compaction and oxidation.

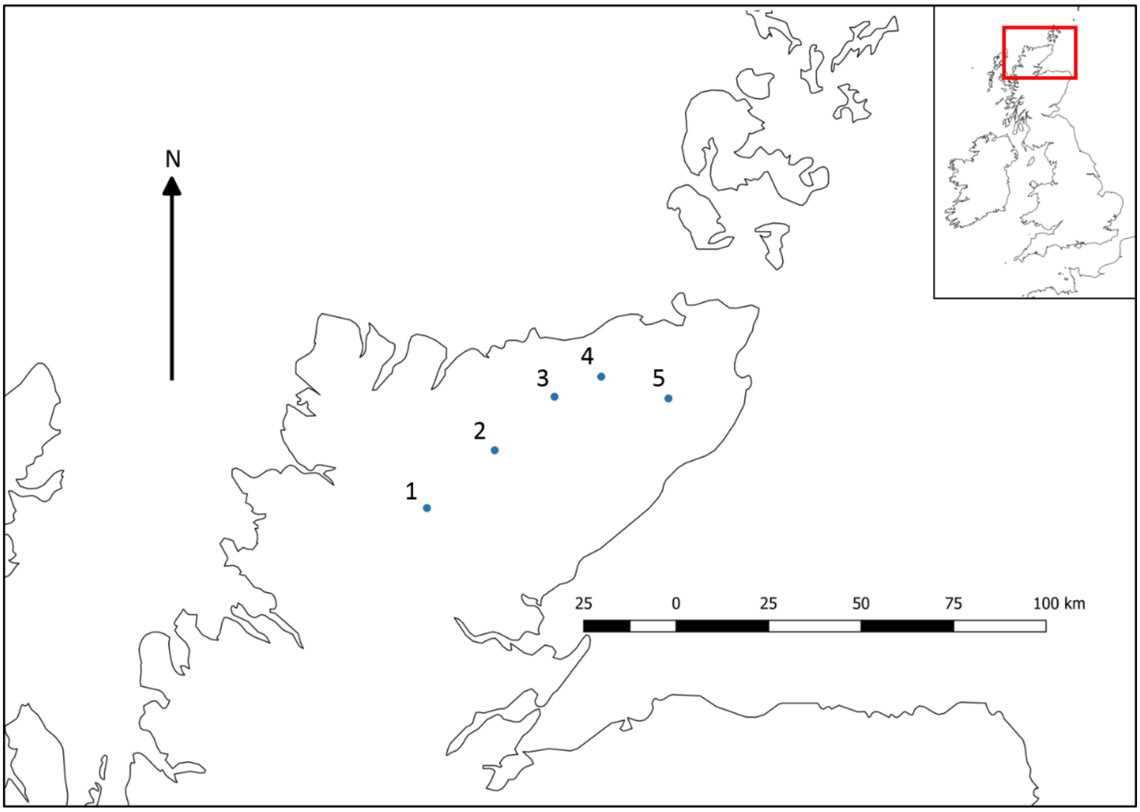

Figure 5.1. Sampling locations around the Flow County of Caithness and Sutherland, Scotland. The sites sampled are 1. Dalchork 2. Rosal 3. Forsinard 4. Broubster and 5. Bad a'Cheo, the intensively sampled site.

Table 5.1. Information on Flow Country field sites used in this study. Tree species encountered on site are Lodgepole Pine (LP) and Sitka Spruce (SS). * Peat depths given for Bad a'Cheo are derived from average values across 27 cores taken for this study. A broader assessment of peat depth across the whole site can be found in Sloan *et al.* (2019).

Site name	Location	Elevation above sea level (m)	Peat depth (cm)		Range of von Post humification values in column	Tree species
			Undrained	Afforested		
Bad a'Cheo	58.430356 ° N 3.428056 ° W	90	492.4 *	361.4 *	H 3 - 8	LP, SS
Broubster	58.478883 ° N 3.725633 ° W	210	276	71	H 1 - 8	LP, SS
Dalchork	58.169150 ° N 4.512133 ° W	185	260	151	H 2 - 8	LP
Forsinard	58.433170 ° N 3.934040 ° W	119	267	271	H 2 - 7	LP, SS
Rosal	58.303900 ° N 4.211983 ° W	103	333	474	H 2 - 8	LP

In 2016/17, three transects were established across Bad a'Cheo, running from the undrained, unplanted bog into the drained and planted blocks of forestry (Figure 5.2). Transect Pyatt 2 had originally been laid out in 1987 (Pyatt *et al.* 1992). Transects 'New 1' and 'New 2' were first established as part of this research in 2016 (Sloan *et al.* 2019). As previous data had been gathered along these transects, and it had been established that they were suitable for extracting peat samples, coring was undertaken at three locations along each transect (Figure 5.2). These locations were: (1) an undrained, unplanted area of open blanket bog located at least 100 m from the nearest drainage, and so assumed to be free of hydrological impact; (2) an area of Lodgepole pine monoculture; and (3) a mixed stand of Lodgepole pine and Sitka spruce. Within each of these three locations, three cores were taken to account for the microtopography of the ground surface. In the open bog, the cores were from hummock, hollow and lawn. In the two forest stand sampling locations, cores were taken from ridge, plough furrow and original surface. The plantation was felled from spring 2017, to be replaced by a wind farm.

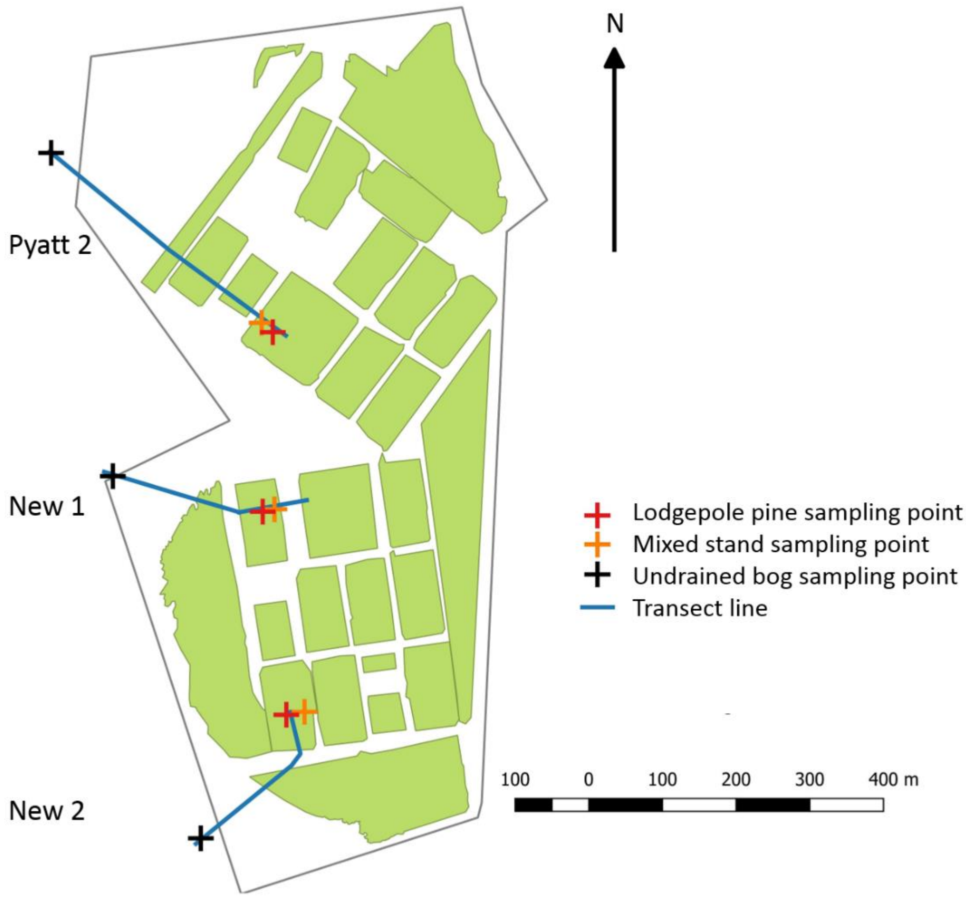

Figure 5.2. Bad a'Cheo site map showing location of transects. Blocks of forestry are highlighted in green.

Four additional study sites (Broubster, Dalchork, Forsinard and Rosal) from around the Flow Country were sampled. These sites covered a range of peat depths and elevations, plus a mixture of plantation ages and tree mixes (Table 5.1). Here, a simplified paired sampling design was used in which a pair of cores were sampled, one within and one outside of an afforested bog. Cores were taken from either lawn peat or original surface of the planted bog (i.e. the strips of ground not occupied by plough furrows or plough ridges). In each instance the paired locations of the coring site were approximately 200 m apart. Within each study site, a forest boundary that matched the ownership criteria (specified above) was identified. To avoid edge effects a point approximately midway across the forestry block was selected along the forest margin. A tape was run out at right angles to the forest boundary, into the plantation and into the undrained bog. At 100 m from the forest boundary the nearest point with the target microtopography (unploughed original surface and lawn respectively) was sampled.

In all cases, a 5 cm diameter Russian corer (Aaby & Digerfeldt 1986) was used to sample peat to the underlying bedrock. At each sampling point two holes were cored within approximately 20 cm of each other, and seven cores taken from alternating holes, overlapping by 5 cm to avoid core compaction during coring. While in the field, cores were described and humification assessed using the von Post scale (von Post & Granlund 1926).

Tephrochronology

The drainage and mechanical preparation of peat for afforestation, in addition to the typical high variability in peat depth in blanket bogs, leads to large differences in peat depth across an afforested site. As peat initiation time may vary and oxidative effects of drainage are likely to be greatest in the top section of the peat. Therefore rather than measuring carbon content to the base of the peat the use of a suitable stratigraphic marker ensures that any comparison is between the same isochrones within the peat cores. One such stratigraphic marker is tephra, shards of volcanic glass which are deposited in clouds of ash up to several thousand kilometres away from major eruptions (Lowe 2011). Cryptotephra, the most distal deposits which form fine layers not visible to the naked eye, are abundant throughout Scottish peat (Dugmore *et al.* 1995, Lawson *et al.* 2012). Cryptotephra deposits are therefore viable stratigraphic markers, although identifying such deposits has traditionally been time consuming (Gehrels *et al.* 2008).

The recent development of ITRAX scanning techniques, which combine radio chromatograph and XRF elemental analysis, now allows for the rapid identification of tephra layers (Croudace *et al.* 2006). The top 320 cm of each core (or the total length of the core where peat depth was less than 320 cm) were XRF scanned, as based on previous work in the region this was assumed to be deep enough to include Hekla 4 deposits. The ITRAX results compared to LOI data to indicate where deposits of inorganic material of interest were located. A strong band of tephra was identified throughout the cores, and through geochemical identification using an electron microprobe and age depth modelling was identified as Hekla 4 (Sloan *et al.* 2020). This layer of Hekla 4 tephra, dated as 2310 ± 20 BCE (Swindles *et al.* 2011), serves as the stratigraphic marker that forms the basis for like-for-like carbon comparisons.

The use of Hekla 4 ahead of other cryptotephra deposits has two advantages: that it is one of the most reliably reported deposits found in deep peat in the north of Scotland, meaning that it is likely to be present in all cores, and that it is typically deposited deep enough in the peat to appear below the extent of the effects of afforestation. It should therefore have not been directly disrupted by ploughing and should remain below the water table following drainage. An inventory of the peat above the point of deposition should therefore capture the most significant effects of drainage and afforestation. The most abundant and well described alternative tephra deposits in the region are Glen Garry (226 ± 244 BCE) and Lairg A (4950 ± 49

BCE), both also from Icelandic eruptions (Swindles *et al*. 2011, Watson *et al*. 2016). Glen Garry, while frequently reported, is a more recent deposit and therefore is likely to appear higher in the peat column in the zone where direct disruption is expected, possibly failing to capture effects lower in the stratigraphy. Lairg A is less frequently reported (it may pre-date peat initiation in some cases) and has additional difficulties in identification, occurring as it does very close to the Lairg B eruption in the column. Lairg A is an old enough deposit that using it on some sites would be functionally indistinguishable from surveying the entire column, a use of resources that would greatly reduce the number of replicates possible, and sampling deep peat less likely to be directly affected by afforestation. The focus on identifying an eruption to fit the criteria of the study is a key disadvantage of the tephra approach, presenting the possibility of a sampling design capturing more or less of the peat column than is required. Using an appropriate tephra deposit does, however, allow a shared stratigraphic marker to be identified with a very high degree of certainty. Due to the innate variability of peat depth and deposition rates even over short distances (Chaudhary *et al*. 2018), using such a marker is more cost and time effective than building age depth models with the number of radiocarbon dates required to reliably sample peat accumulated over the same time period across multiple replicates.

Age-depth models

Age-depth models were used to validate the geochemical tephra identification based on its position in the peat profile, and to provide a point of depth of appropriate age in the single instance where a layer of Hekla 4 could not be identified (the Dalchork bog core). Data for the age-depth modelling was derived from radiocarbon analysis of the peat at 1 m depth in all cores, at the base of the peat in open bogs and drained forest, and from points where additional validation was needed due to insufficient tephra deposits for microprobe analysis (Table 4.2). Samples were pre-treated using the acid-base-acid method, and vertical rootlets were picked out prior to radiocarbon analysis using bulk samples (AMSDirect). Age depth modelling was completed with the BACON package in R (Blaauw & Christen 2011), with IntCal3 used to calibrate the radiocarbon dates (Reimer *et al.* 2013).

Peat carbon stocks

During consolidation, drying, compaction and oxidative carbon loss associated with peatland afforestation, the physical properties and carbon content of the peat are likely to change. To assess this, all cores were subsampled in 5 cm intervals down to 317.5 cm. Sample volume was measured by water displacement, the samples were dried at 105 °C to a constant weight (typically for 48 hours) and weighed to calculate bulk density (Chambers *et al.* 2011). Loss on ignition (LOI) was determined by combusting the subsamples at 550 °C for four hours in a muffle furnace.

The carbon content of the peat was determined at 10 cm intervals using a C/N analyser (Elementar, Vario Macro), with each 10 cm increment providing carbon data for the adjoining two 5 cm increments. Where there were gaps in the data (86 samples), carbon content was calculated based on the regression between LOI and carbon content for all other measurements taken in this survey (888 samples).

Core carbon stock was calculated using the Apparent Carbon Accumulation since Hekla 4 (ACA_{Hek4}). This uses a formula adapted from Mäkilä & Goslar (2008):

$$ACA_{Hek4} = \sum \text{inc} \times BD \times C$$

Where 'ACA_{Hek4}' is the accumulation since Hekla 4 (g cm^{-2}), 'inc' is the depth in cm of the increment (usually 5 cm), 'BD' is the dry peat bulk density (g cm^{-3}), and 'C' is the carbon content in proportion to % bulk density. The sum of all the carbon in each 5 cm increment to the depth of Hekla 4 gives the final carbon accumulation figure. Carbon content of undrained bog and afforested bog were directly compared to estimate the total net change in carbon on the planted site.

It is important to note that ACA_{Hek4} represents an assessment of the current stock of carbon above Hekla 4, and is not presented as a yearly rate of carbon accumulation. There are additional considerations in deriving such a rate using these data, primarily due to the mechanical removal of peat by ploughing. Other studies do not include the top 50 cm of peat when calculating ACA in order to avoid acrotelm peat which may be less decomposed than in the deeper catotelm (Ratcliffe *et al.* 2018). The inclusion of this upper portion of peat is an important factor in calculations of carbon accumulation rates. Young *et al.* (2019) have demonstrated that carbon in the upper acrotelm cannot be considered as part of the long-term peat store as it may still potentially be oxidised and lost to the atmosphere. Any carbon accumulation rate derived from only near surface peat is therefore likely to be incorrect, and studies that use the near surface peat to calculate the rate of peat formation may present misleading rates of carbon accumulation. This study does not calculate a figure for the current rate of accumulation (although trends may be broadly inferred), but rather assesses net differences in total carbon stock between paired samples over a discrete period (a single forestry rotation). That all the carbon held in the upper peat layers may not form part of the long-term carbon store should be noted when assessing the data, and would be important in projecting rates of accumulation, but otherwise does not alter the overall assessment of current stock.

Further consideration of the disruption inherent in afforestation has dictated the sampling design used here. Ploughing for afforestation directly disturbs the peat surface and vegetation, meaning that 50 cm may already have been lost or added (as furrows and ridges). Removing the top 50 cm in the undrained bog eliminates a portion of un-humified material with a low bulk density and carbon content. Removal of a similar increment in the afforested bog is likely to eliminate former catotelm peat with a high bulk density and carbon content, which is now closer to the surface. That this material is now above the water table in afforested sites means that it is as likely to be lost to the atmosphere as any other near surface peat and should be removed under the same criteria. To do would therefore introduce a systematic bias underestimating the stock of carbon remaining in afforested sites.

Forest biomass measurements

While drainage and afforestation are likely to give rise to carbon loss from peatlands this will be compensated, to a varying degree depending on tree growth, by carbon accumulation in tree biomass. To assess this, tree plantations at the study sites were surveyed. In each case, standard forestry practice for peat afforestation had been used (furrows and ridges ploughed into the peat, drainage ditches surrounding forest blocks, 2 m tree spacing). These surveys took two forms: (1) felling of sample trees at Bad a'Cheo to directly measure carbon content, and (2) a non-destructive survey of tree morphometrics which were then used to assess variation across the site.

Within Bad a'Cheo, 20 plots of 16 trees were established along the transects in blocks of 4 x 4 trees, with ten plots each of Lodgepole pine monocultures and mixed stands. For all trees in these plots diameter at breast height (DBH) was measured, and the height of trees was derived from a trigonometric calculation based on measurements using a clinometer.

For the destructive sampling at Bad a'Cheo, ten Sitka spruce and 20 Lodgepole pine (ten each from the pure Lodgepole pine and the mixed stands) were chosen at random and felled from within the measured plots. The trees were subdivided into ten sections, five above and five below the crown. These sections were weighed in the field, with a disk taken from the base of each section. Branches from the different sections were weighed separately (Bert & Danjon 2006, Major *et al.* 2013, Bembenek *et al*. 2015).

Fresh weights of the subsampled tree disks were recorded, the disks were air dried and then oven dried at 60 °C to constant weight. The dry:fresh weight ratios of the disks were used to determine the dry weight of each trunk section. 100 subsamples were taken from a selection of ten trees representing the range of tree sizes found on the site and measured for carbon content (Elementar, Vario Macro). A subsample of 25 branches were also dried, milled and analysed for carbon content as above.

Belowground tree biomass is typically not part of the commercial yield of wood. Sampling large woody roots presents logistical challenges, meaning that we adopted differing strategies for the two species.

For Lodgepole pine, woody roots were sampled from wind-thrown trees where the root architecture is readily available for sampling. Lone wind-thrown trees might provide a sample biased towards the more weakly rooted trees in a stand, which could imply an unrepresentative biomass. However, wind-throw once initiated can often propagate widely across a plantation leading to the toppling of normal trees with typical root structures. To avoid bias we targeted trees in the centre of such large areas of wind-throw. Trees toppled by the wind typically expose a large portion of roots but remain rooted on the fallen side. These trees were visually assessed for large snapped roots (which would suggest biomass remained in the ground) then the exposed top half of the root plate and base of the trunk was sectioned and weighed, assuming that this section represented half of the total root mass. The DBH of the fallen tree was measured so that the root mass could be allometrically related to the other stems sampled.

Large areas of wind-thrown Sitka spruce were not present on the site and it was therefore necessary to excavate these roots. This was undertaken by digging around trees felled in the above ground biomass survey. The excavated root balls and stumps were then winched out of the ground and weighed. For both tree species, a subsample from the central root ball and from a younger outlying root was taken to determine dry weight and carbon content.

Fine roots were sampled using a box corer, but it proved unfeasible to practically separate tree roots from other root and plant fibre material. Fine roots have been excluded from the analysis in order to avoid the overestimate of biomass that would arise from this. While this will lead to a small systematic underestimate of biomass that would otherwise be expected to remain in situ after harvesting, such an underestimate is likely to be extremely small. Indeed, fine root biomass data is often excluded from assessments of tree biomass due to the difficulty of reliably measuring it (Bert & Danjon 2006).

Biomass and carbon content were determined for all destructively surveyed trees as described above. The Bad a'Cheo plantation was felled from spring 2017 when the site was taken over by a renewable energy company for a wind farm development. Information on timber yield and destination and use of the wood was obtained from the logging company. Directly measured carbon content was allometrically scaled up a total tree carbon content for Bad a'Cheo using the biomass removed from the site and assuming that loss of moisture between felling and loading onto lorries, where the wood was weighed, was negligible.

Mann-Whitney U tests were used to compare means bulk density values, and Kruskal-Wallis or ANOVA tests in the surveyed DBH of the Lodgepole pine and Sitka spruce respectively (IBM SPSS, Version 26). All uncertainties are reported as one sigma standard deviation.

Uncertainties in the data

While the methods used here represent the most reliable way to determine the net change in stock, limitations of the approach and possible sources of uncertainty should be acknowledged. An assessment of carbon stock allows the net change in carbon to be measured, but it does not describe the relative importance of the components of the carbon budget (gas flux, DOC etc.; Sloan *et al.* 2018). When water tables in peatlands change, the balance of greenhouse gasses released will also alter. Drainage increases the oxidative production of CO_2 while undrained bogs typically release more CH_4, a gas with a greater global warming potential over 100 years (Drosler *et al.* 2008, IPCC 2013). While this study describes the net change in stored carbon, the relative components of the atmospheric and aqueous losses are not described. It is assumed that most carbon is released as CO_2, an assumption supported by the balance of literature measuring gas flux in peatlands (Von Arnold *et al.* 2005, Yamulki *et al.* 2013, Ojanen *et al.* 2014).

The inclusion of the top 50 cm of the peat profile in the assessment of carbon stock has already been explored, and has been justified based on the nature of the data (a calculation of the current stock rather than of a rate of accumulation) and on the mechanical disruption and peat removal already evident on the afforested peat. It should however be acknowledged that this data set may not be appropriate for calculating long term rates of peat accumulation without omitting this near surface peat.

The number of replicates in this study is made possible through the use of cryptotephra as a rapidly identifiable stratigraphic marker. Such an approach is applicable where an abundant, easily identifiable deposit is available at a point in the stratigraphy below the expected extent of the disruption caused (Swindles *et al.* 2010). While advances in rapid identification of cryptotephra through ITRAX scanning has aided the locating of cryptotephra layers, finding a deposit to constrain the observed affects is more problematic. Hekla 4 is the best choice in that it is widely distributed throughout the north of Scotland, abundant, and deep enough in the profile to capture the main impacts of drainage and afforestation. Some secondary impacts may however be missed by this approach. Although peat below Hekla 4 has not been directly aerated or mechanically influenced, it may have been subjected to compression and compassion due to increased pressure from the weight of drained peat above the water table. In addition, in the afforested Broubster samples the peat was so diminished that Hekla 4 was near enough to the surface that the effects of drainage probably extend below the deposit. In such a case the use of the whole column may have been more accurate. Such an approach would, however, have been impractical from a cost and time perspective.

Carbon content was calculated using 10 cm increments. Some of the granularity in the data (arising from charcoal, inorganic material, large wood fragments, ash deposits etc.) may not be fully captured this sampling strategy. Although inorganic sections tend to form a relatively small part of the peat this is a source of uncertainty in the data, albeit one that is unavoidable for time and budgetary reasons.

A paired design based on previously surveyed transects was adopted at Bad a'Cheo rather than a fully randomised design. This experimental design paired sampling points in the undrained bog with those in the afforested plantation (further subdivided into Lodgepole monocultures and mixed stands) along already established transects. This was in part an attempt to build on previous work on the site which used this broad framework, including the data presented in chapter 2 on the extent of subsidence within the transects (Pyatt *et al.* 1992, Sloan *et al.* 2019). The survey design was also based on considerations around site accessibility. Bad a'Cheo had high levels of wind-throw in some areas, and in others access was restricted by a lack of proper brashing of branches. Had a random design be adopted, there is no guarantee that sampling points could be accessed safely. Land use around the site also dictated the distribution of the undrained bog sampling points, which are all sited along the eastern edge of the plantation (Figure 5.2). To the west the site was bordered by a road, to the north a windfarm, and to the south an area of commercial peat cutting. The eastern margin was the only portion that fulfilled the criteria set out for site selection, particularly the stipulation that similar peat divided by a land ownership boundary should be sampled. In addition, a system of pools between Pyatt 1 and New 1 prevented sampling between these two points. The range of usable sampling points was therefore limited, and the design adopted here offers the best balance between robustness and accommodation to the conditions on site.

The root component of the wood biomass assessment is a possible source of uncertainty and error. Due to the inability to use standard root excavation techniques on a deep peat plantation (see methods), the total sample size for woody roots samples was smaller than the number of above ground destructive samples. There is also a risk of underestimating the woody root biomass inherent in using wind-toppled trees, in which some of the root material may remain in the peat. While on site assessment of damage to roots was undertaken during the survey, this should still be acknowledged as a possible source of error in the data. As previously discussed, fine roots were also excluded from the investigation, although the contribution of fine root biomass to total biomass is negligible (Levy *et al*. 2004).

More generally, there is also uncertainty about fate of the carbon stored in the wood biomass that remains on site, which is tied to the future use of the plantations. Carbon in tree roots and brash left after harvesting may be released to the atmosphere, along with a large portion of the carbon in near surface peat, if water tables remain lowered in preparation for a second forestry rotation. The release in such a second rotation is likely to be greater than in the first rotation due to brash mats supporting better tree growth and a possible priming effect from the roots (although the evidence for the priming effect is mixed; Linkosalmi *et al*. 2015). Conversely, if sites are restored reprofiling will integrate that wood into the peat largely below the water table and carbon is consequently less likely to be lost (Zeng 2008). As this study represents results of a first rotation is not fully representative of losses through the second rotation, where oxidative loss of carbon is likely to be more pronounced.

Results

Peat carbon

Differences in peat carbon accumulation (‘ACA_{Hek4}‘) was determined using deposits of Hekla 4, bulk density measurements and carbon contents. Previous work on site has established that the depth of Hekla 4 from the peat surface showed a significantly different depth distribution on undrained peat (244.0 cm) and in drained afforested peat (134.4 cm), for an average difference in 120 cm (Sloan *et al.* 2020). This is in line with an average total peat depth of 492.4 ± 44.11 cm and 361.4 ± 59.25 cm respectively, suggesting a consolidation and possible loss of peat due to afforestation.

Mean bulk density of the peat down to Hekla 4 was 0.0679 ± 0.02 g cm^{-3} in the undrained samples, and 0.1092 ± 0.04 g cm^{-3} in the drained, afforested samples (Table 5.2). Bulk density was significantly higher in drained afforested samples than in undrained samples ($n = 958$, $Z = -20.631$, $p \leq 0.001$). There was no significant difference in bulk density between the peat under the mixed stands and Lodgepole pine monocultures ($n = 514$, $Z = -1.189$, $p = 0.234$). Across the afforested samples, the bulk density at New 2 (0.1381 ± 0.05 g cm^{-3}) was higher than that of Pyatt 2 (0.0624 ± 0.01 g cm^{-3}) and New 1 (0.0809 ± 0.02 g cm^{-3}), and this difference was statistically significant ($n = 339$, $Z = -8.953$, $p < 0.001$) ($n = 341$, $Z = -13.631$, $p < 0.001$).

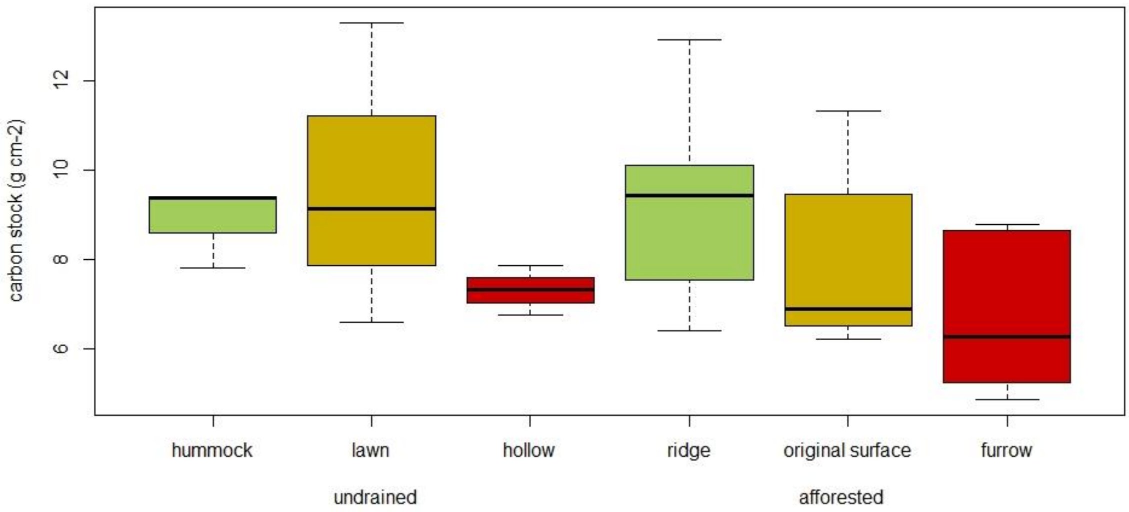

Figure 5.3. Apparent carbon accumulation since Hekla 4 (ACA_{Hek4}) (g cm^{-2}) across the various microtopography replicates at Bad a'Cheo.

Three microtopography replicates were used in each site to account for possible differences in depth over small spatial scales. ANOVA tests were used to determine the differences between the means of ACA_{Hek4} for the three microtopography categories in both the undrained and afforested samples (Figure 5.3). In the undrained bog the hummock, hollow and lawn section had an average ACA_{Hek4} of 8.87 ± 0.76 g C cm^{-2}, 9.68 ± 2.77 g C cm^{-2}, 7.31 ± 0.46 g C cm^{-2} respectively. There was no significant difference between these means ($P = 0.395$). The differences in microtopography were more evident in the afforested sections where ploughing and drainage make the microtopography differences more pronounced. For ridges, original surface and hummock mean ACA_{Hek4} was 9.31 ± 2.09 g C cm^{-2}, 7.88 ± 1.88 g C cm^{-2}, 6.67 ± 1.54 g C cm^{-2} respectively. This difference was also not significant ($P = 0.094$). The standard deviation of all the results are relatively high, which is explicable as they represent cores from a range of sampling points across a site with highly variable peat depth. The difference in ACA_{Hek4} between the undrained and afforested samples varied across the three Bad a'Cheo sample sets (Figure 5.4; Table 5.3). Across the three transect groups, comparing the average carbon accumulation to Hekla 4 in the undrained bog to the afforested plots showed an average difference of ACA_{Hek4} of 0.93 ± 1.35 g cm^{-2} at Pyatt 2, loss of 3.39 ± 1.19 g cm^{-2} at New 1, and an accumulation of carbon of 2.32 ± 1.09 g cm^{-2} at New 2. The average difference across these three transects was a reduction of 0.67 ± 2.86 g cm^{-2} in afforested peat relative to undrained peat. This figure will form the basis of the comparison with uptake in wood biomass, but there are important caveats about the variability of the data. The data is highly variable, both between the transect groupings and within sets of microtopography replicates. This is especially evident in the broad range of results observed in the afforested samples (Figure 5.5). Because of this variation the carbon stock calculations will include likely ranges, based on standard deviation, into which the results may fall.

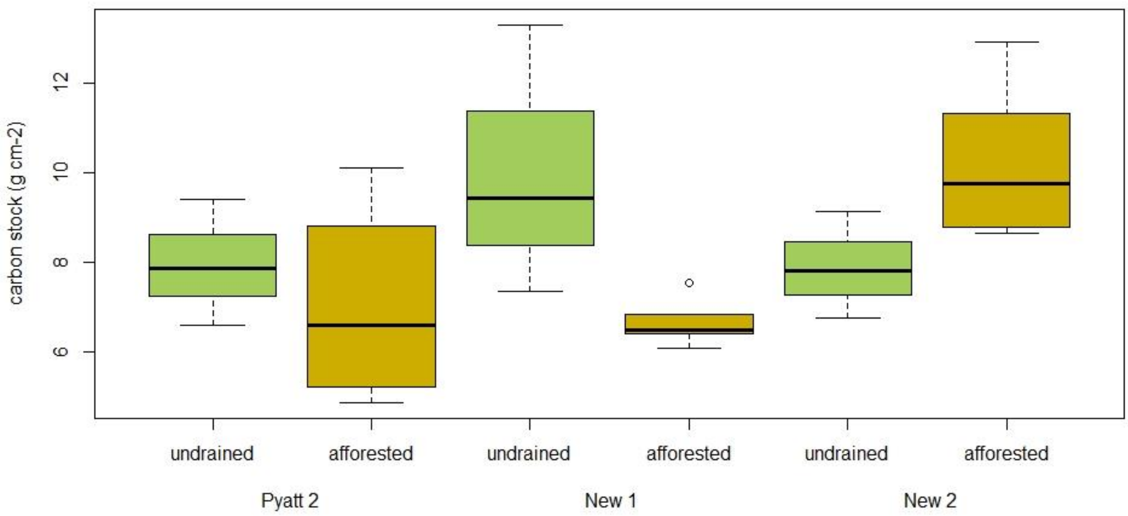

Figure 5.4. Apparent carbon accumulation since Hekla 4 (ACA_{Hek4}) (g cm^{-2}) across each sample grouping, divided by drainage and afforestation status, at Bad a'Cheo.

A principal component analysis (PCA) was used to assess the data further. Figures for ACA_{Hek4}, bulk density, depth to Hekla 4, carbon content, and total peat depth (Table 5.2) were included. For the purpose of the analysis, and to meet the assumptions of the PCA, the microtopography was categorised as high (hummocks in undrained bog and ridges in afforested plantations), medium (lawn and original surface) and low (hollows and furrows). Figure 5.6 shows that the drainage and afforestation status of the sites are clearly distinguishable along the first axis of the PCA, with afforestation sampling locations associated with denser peat and shallower peat depth, (both in total and to Hekla 4). The second PCA axis relates more strongly to ACA_{Hek4} and illustrates the high variation between sites and microtopography, as no clear clusters emerge.

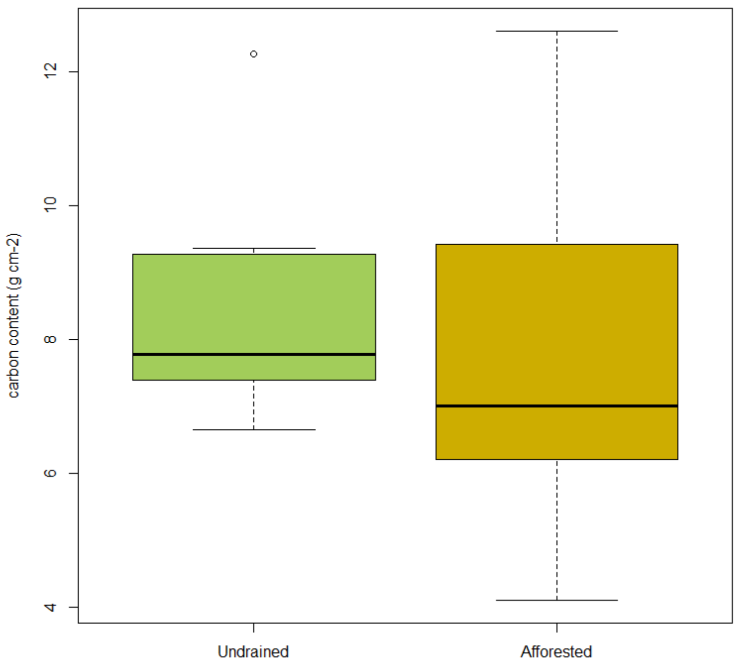

Figure 5.5. Apparent carbon accumulation since Hekla 4 (ACA_{Hek4}) (g cm^{-2}) in undrained and drained, afforested samples at Bad a'Cheo, not grouped by transect.

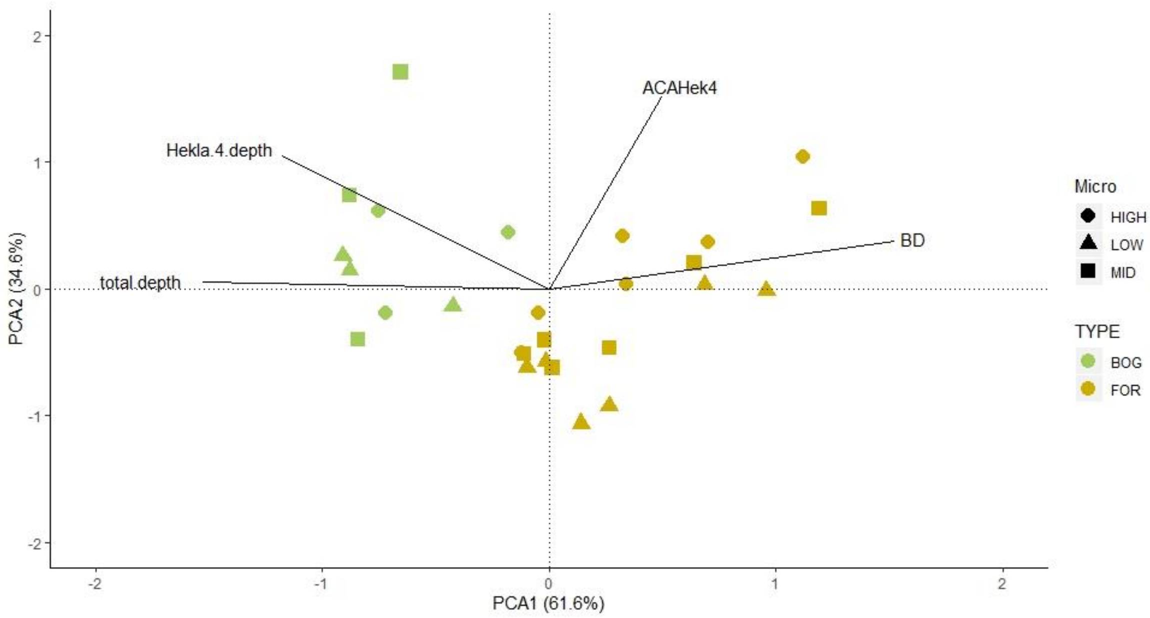

Figure 5.6. PCA analysis of variation at the Bad a'Cheo site. Microtopography types are shown as 'micro' and drainage status as 'type', with 'bog' referring to undrained blanket bog, and 'for' referring to drained, afforested sampling points.

Table 5.2. Peat characteristics from all sampled cores. LP denotes a stand of Lodgepole pine, and SS denotes a stand of Sitka spruce.

Site	Core I.D.	Sample type	Microtopography type	Tree species	total peat depth (cm)	Hekla 4 deposit depth (cm)	Mean bulk density (g cm^{-3})	mean LOI (%)	mean C content (%)	Carbon accumulation since Hekla 4 (g cm^{-2})
Bad a'Cheo	P2H2	Bog	hummock	NA	514	259	0.07	97.41	52.53	9.39
Bad a'Cheo	P2H3	Bog	hollow	NA	525	255	0.06	97.26	51.94	7.87
Bad a'Cheo	P2H4	Bog	lawn	NA	520	212	0.06	97.58	53.96	6.59
Bad a'Cheo	P2H1	Forest	furrow	LP/SS mix	386	92	0.09	97.66	50.48	4.85
Bad a'Cheo	P2H5	Forest	original surface	LP/SS mix	410	126	0.09	97.44	54.14	6.20
Bad a'Cheo	P2H6	Forest	ridge	LP/SS mix	435	151	0.13	95.11	51.62	10.11
Bad a'Cheo	P2H7	Forest	furrow	LP	370	98	0.10	96.95	51.97	5.22
Bad a'Cheo	P2H8	Forest	original surface	LP	365	122	0.10	96.43	51.93	6.95
Bad a'Cheo	P2H9	Forest	ridge	LP	406	133	0.12	92.85	51.59	8.82
Bad a'Cheo	N1H2	Bog	hollow	NA	485	183	0.08	95.81	49.57	7.33
Bad a'Cheo	N1H3	Bog	hummock	NA	387	221	0.08	96.73	51.81	9.42
Bad a'Cheo	N1H4	Bog	lawn	NA	493	318	0.08	95.84	48.04	13.32
Bad a'Cheo	N1H1	Forest	furrow	LP/SS mix	388	133	0.08	95.81	49.57	6.47
Bad a'Cheo	N1H5	Forest	original surface	LP/SS mix	410	144	0.09	97.76	52.65	6.83
Bad a'Cheo	N1H6	Forest	ridge	LP/SS mix	410	160	0.09	96.94	52.04	7.53
Bad a'Cheo	N1H7	Forest	original surface	LP	410	143	0.08	97.43	52.35	6.50
Bad a'Cheo	N1H8	Forest	furrow	LP	405	139	0.08	97.50	53.07	6.06
Bad a'Cheo	N1H9	Forest	ridge	LP	405	150	0.08	97.31	51.58	6.40
Bad a'Cheo	N2H1	Bog	lawn	NA	478	304	0.06	97.79	51.01	9.13
Bad a'Cheo	N2H2	Bog	hummock	NA	536	234	0.06	95.39	50.97	7.80
Bad a'Cheo	N2H3	Bog	hollow	NA	494	210	0.06	97.30	51.35	6.74
Bad a'Cheo	N2H4	Forest	original surface	LP	298	146	0.12	96.79	53.70	9.46
Bad a'Cheo	N2H5	Forest	ridge	LP	310	148	0.13	97.67	51.34	10.07
Bad a'Cheo	N2H6	Forest	furrow	LP	292	134	0.12	97.01	52.00	8.79
Bad a'Cheo	N2H7	Forest	original surface	LP/SS mix	260	131	0.16	96.77	52.27	11.34
Bad a'Cheo	N2H8	Forest	ridge	LP/SS mix	284	149	0.16	96.23	52.22	12.92
Bad a'Cheo	N2H9	Forest	furrow	LP/SS mix	262	121	0.14	96.87	52.16	8.65
Broubster	BroB	Bog	lawn	NA	276	136	0.09	97.53	51.76	6.23
Broubster	BroF	Forest	original surface	LP/SS mix	71	21	0.18	95.16	48.88	1.99
Dalchork	DalB	Bog	lawn	NA	260	136	0.07	98.26	53.89	5.30
Dalchork	DalF	Forest	original surface	LP	151	98	0.10	92.63	51.13	5.18
Forsinard	ForB	Bog	lawn	NA	267	189	0.10	96.86	53.53	10.91
Forsinard	ForF	Forest	original surface	LP/SS mix	271	171	0.12	97.26	53.94	11.10
Rosal	RosB	Bog	lawn	NA	333	187	0.08	97.49	52.97	7.85
Rosal	RosF	Forest	original surface	LP	474	154	0.09	97.16	53.13	7.53

Generally, the results from the other sites showed a comparatively lower ACA_{Hek4} in afforested areas than in undrained areas, although this was highly variable (Figure 5.7). Peat in undrained sites was generally deeper than in drained afforested sites, at 325.68 ± 87.23 cm and 265.68 ± 143.98 cm respectively, although there was insufficient data to determine statistical significance (n = 5). The bulk density was significantly lower on the undrained sites than on the drained afforested sites, 0.0709 ± 0.02 g cm^{-2} and 0.1087 ± 0.04 g cm^{-2} respectively (n = 1176, Z = - 21.283, p ≤ 0.001), as it had been in the Bad a'Cheo samples. In three of the four additional sites, carbon accumulation to Hekla 4 was lower where sites were drained and afforested (Table 5.3). The exception was at Forsinard, where the ACA_{Hek4} accumulation in the afforested sample was 0.19 g cm^{-2} greater than that of the undrained sample (11.10 g cm^{-2} and 10.91 g cm^{-2} respectively).. The average difference between peat carbon content above Hekla 4 across all five study sites was 1.03 g cm^{-2} ± 1.82 (an average of 7.78 ± 2.18 g cm^{-2} and 6.75 ± 3.39 g cm^{-2} for undrained bog and afforested bog respectively)although this value is in part attributable to the large negative result for the Broubster samples, as evidenced by the higher standard deviation value for the afforested sites.

Table 5.3. Change in peat carbon accumulation since Hekla 4 in afforested and undrained peatlands.

Site name		Afforested LP/SS mix		Afforested LP		Afforested Average		Undrained bog		Net carbon change
		g C cm^{-2}	SD	g C cm^{-2}	SD	g C cm^{-2}	SD	g C cm^{-2}	SD	g C cm^{-2}
Bad a'Cheo transects	Pyatt 2	7.05	2.23	6.99	1.47	7.02	1.89	7.95	1.14	-0.93
	New 1	6.94	0.44	6.32	0.19	6.63	0.46	10.02	2.48	-3.39
	New 2	10.97	1.76	9.44	0.52	10.20	1.51	7.89	0.98	2.32
	Bad a'Cheo average	8.32	1.87	7.58	1.34	7.95	1.67	8.62	0.99	-0.67
	Broubster	1.99	-	-	-	-	-	6.23	-	-4.24
	Dalchork	-	-	5.18	-	-	-	5.30	-	-0.12
	Forsinard	11.10	-	-	-	-	-	10.91	-	0.19
	Rosal	-	-	7.53	-	-	-	7.85	-	-0.33

Figure 5.7. Difference in Apparent Carbon Accumulation in peat since Hekla 4 (ACA_{Hek4}) between undrained bog and adjoining drained, afforested bog (g per cm^{-2}). Negative values show a net loss of carbon under afforestation. Pyatt 2, New 1 and New 2 are three separate transects at the Bad a'Cheo plantation, the average value of which is presented with the other sites.

Forest carbon

The destructive sampling of tree biomass showed that on average dry tree biomass is 56 % and 50 % of fresh biomass in Lodgepole pine and Sitka spruce respectively (Table 5.4). In the lowest section of stem, which comprises the oldest wood, dry tree biomass is 62 % and 55 % in Lodgepole pine and Sitka spruce respectively. In the top section it is 44 % and 44 % in Lodgepole pine and Sitka spruce respectively. The average carbon content in dry wood in the stem is 48 % in both species.

In Lodgepole pine carbon derived from stem biomass represented 68 % of total carbon, branch biomass 16 % and woody roots 16 %. In Sitka spruce stem biomass represented 61% of total carbon, branch 18 % and woody roots 21 %.

There was no significant variation in average DBH of Lodgepole pine trees across the three transects ($n = 241$, Chi square = 1.390, $p = 0.499$), which were 19.55 cm on Pyatt 2, 19.32 cm on New 1 and 19.58 cm on New 2. Sitka spruce size did vary significantly across the site ($n = 80$, $F = 8.564$, $p = \leq 0.001$) with the average of Pyatt 2 (16.38) and New 1 (12.60 cm) significantly smaller than the average DBH of New 2 (24.58 cm).

Table 5.4. Fresh to dry weight ratios and average carbon contents in sections of Lodgepole Pine (n = 20) and Sitka Spruce stems (n = 10), divided into ten sections, where A is at the base and J is at the top.

Tree increment	Lodgepole pine				Sitka spruce			
	fresh:dry ratio	SD	C content (%)	SD	fresh:dry ratio	SD	C content (%)	SD
A	0.62	0.04	48.98	0.51	0.55	0.06	48.57	0.60
B	0.61	0.07	48.48	0.92	0.54	0.04	48.38	0.97
C	0.60	0.07	48.95	0.78	0.52	0.04	48.12	0.56
D	0.59	0.08	48.33	0.72	0.51	0.03	48.42	0.84
E	0.58	0.09	48.65	0.79	0.48	0.03	48.21	0.83
F	0.59	0.09	48.17	1.27	0.52	0.05	48.58	0.40
G	0.52	0.07	47.25	2.88	0.47	0.04	48.37	0.69
H	0.53	0.11	47.46	2.16	0.47	0.05	48.32	0.49
I	0.48	0.11	48.08	0.81	0.44	0.02	48.29	0.36
J	0.44	0.12	47.90	0.82	0.44	0.02	48.37	0.63
Roots	0.42	0.05	47.06	1.64	0.48	0.08	46.90	1.42

When the Bad a'Cheo site was felled in 2017, 21.43 ha were mechanically felled with brash and stumps mulched, and in the remaining areas 0.24 ha trees were hand felled. From these 21.67 ha timber weighing 2946 tonnes was extracted. Of this, 2363.16 tonnes were used for wood fuel, power generation and wood pellets. The remaining 583.28 tonnes were split between fuel use and the manufacture of boards. In a further 5.86 ha whole trees were mulched and not removed from Bad a'Cheo.

Using the allometrically derived ratios of carbon to fresh biomass and assuming that moisture loss between felling and weighing is negligible, this extracted fresh weight is the equivalent of 680.53 tonnes of carbon (t C). Including the components that were not removed from site, this is the equivalent to 1023.58 t C, when including the other (non-harvestable) components of the tree. Of this, woody roots left in the peat accounts for 172.68 t C. The remaining 170.37 t C is in the form of brash from branches left on site, or portions of stem not removed. The proportion of trees not removed because they were damaged by wind-throw, misshapen, dead or used to reinforce extraction routes is not known but will have contributed to the low yield.

Total carbon budget

A mean difference in carbon storage of 0.67 g cm^{-2} is equivalent to 66.72 tonnes ha^{-1} less storage in afforested bogs relative to the accumulation in the same area of undrained bogs. A total area of 27.53 ha was afforested on Bad a'Cheo. Upscaling the average differences of carbon accumulated since Hekla 4 across the afforested portion of the site gives a total loss of 1838.94

tonnes of carbon. The 27.53 ha of afforested area of Bad a'Cheo was subdivided into several components for harvesting. In the 21.67 ha of Bad a'Cheo which produced an extracted wood product, 1449.51 t C was lost from the peat through drainage and afforestation, while 1023.58 t C was sequestered in tree biomass. This represents a net loss of 425.93 t C across the section of the plantation which produced harvestable wood products, or the equivalent of 19.49 t C ha^{-1} (Table 5.5).

While the mean value for the carbon stock data suggests a net loss of carbon from afforested peat, the data were highly variable and therefore a range of possible scenarios should be acknowledged. Within one standard deviation of the data, the Bad a'Cheo difference in peat carbon stock ranges from a net loss of 305.69 t C ha^{-1} to a net gain of 266.72 t C ha^{-1}. The range is less pronounced when the data from across all sites are considered using the bad a'Cheo forest accumulation values, varying from a net loss of 238.00 t C ha^{-1} gain of 126.72 t C ha^{-1} (Table 5.5).

Table 5.5. Summary of the range of estimates of change in carbon storage at Bad a'Cheo and across all study sites. Estimates of net change with forestry are based on the Bad a'Cheo tree biomass values, which yield a carbon sequestration of 47.23 t C ha^{-1}.

site	estimate type	change in carbon stock (g C cm^2)	change in carbon stock (t C ha^{-1})	Net change with forestry (t C ha^{-1})
Bad a'Cheo	upper (+ 1 SD)	2.19	219.48	266.72
	mean	-0.67	-66.72	-19.49
	lower (- 1 SD)	-3.53	-352.92	-305.69
all sites	upper (+ 1 SD)	0.79	78.83	126.07
	mean	-1.03	-103.20	-55.97
	lower (- 1 SD)	-2.85	-285.23	-238.00

On the remaining 5.86 ha of afforested bog no trees were removed, but were mulched on site, again most likely due to wind-throw or misshapen and dead wood, or problems of accessing portions of the site with machinery. The amount of wood biomass unharvested in this way is unknown, and the fate of wood carbon will depend on the future drainage status of the peat. The total carbon stored in peat in this 5.86 ha area is 391.43 t C lower than would be stored in undrained peat.

Discussion

Impact of afforestation on peatland carbon

The average depth of peat as the sampling point was significantly less in drained, afforested blocks than in undrained areas. This is in line with previous ground elevation surveys and peat depth probing on the wider site (Sloan *et al.* 2019). The average depth of Hekla 4 was also significantly greater in undrained peat than in drained, afforested sites (Sloan *et al.* 2020). Bulk density was significantly higher in the afforested site than in the undrained bog. These two characteristics, decreased peat depth and increased bulk density, show consolidation and compaction which is a result of drainage and the subsequent growth of trees.

This study presents a carbon budget for drained afforested peatlands during the first forestry rotation, which shows a degree of compensation in tree biomass for carbon loss from peat, but an overall net reduction of carbon storage relative to what would be expected if the peat had remained undrained. Over the course of the fifty-year forestry cycle at Bad a'Cheo, there was a net loss of 19.63 t C ha^{-1}, and a likely even greater loss if the 5.86 ha of forestry which did not produce a wood crop were factored in. The reduction in the carbon stored in afforested peat at the other study sites in the region were of the same order of magnitude. As UK peatlands are estimated to store up to 2302 Mt of carbon (Billett *et al.* 2010), these findings suggest that afforestation of this peat may be a significant source of carbon emissions.

While a loss of carbon in the afforested sites is suggested by the data, the values presented for ACA_{hek4} are highly variable. This must caveat the reliability of the results, suggesting there is a wide range of values into which the overall carbon budgets could fall. The variation is such that within one standard deviation of the mean a large net gain of carbon could be possible, even before the addition of forest biomass (Table 5.5). This broad variation is in part due to the spatial variability inherent in peatlands, and indeed data sets in the much better studied Finnish afforested peat context have also reported wide variation (Simola *et al.* 2012). The spread of the results suggests that more data is needed to minimise, and possibly to exclude, outlying data points (potentially including New 2 and Broubster, at the extremes of the range, discussed below). The carbon budgets produced in this study rely on upscaling the peat and tree biomass components. It is Important to understand where variability and uncertainty may arise from in these two estimates.

In the peat component of the carbon stock, the use of a cryptotephra marker as a stratigraphic point of comparison allows a level of replication that would not have otherwise been possible.

The heterogeneity of peat bogs in which several factors (most notably depth and bulk density) can vary across short distances means that much of this replication was focused on accounting for variation in microtopography, and there was a high degree of variability within many of the sampling sites. Figure 5.5 shows a much higher range of ACA_{Hek4} results in the afforested than in the undrained sections, especially in the upper quartiles. This seems to be driven by the large spread of ACA_{Hek4} results in Pyatt 2 and New 2 (Figure 5.4). This is explicable by the nature of the disruption caused by ploughing. The mechanism of ploughing, in which furrow peat is redistributed to ridges, creates high artificial differences between the microtopographies of approx. 50 cm between levels (Sloan *et al.* 2018). The range of variation within replicates at an afforested sampling point should therefore be expected to be higher than in an undrained landscape, where differences in microtopographies are less pronounced. The variation in the data is also high when the microtopographies are grouped and analysed together (Figure 5.3), which is likely to be a result of the broader differences in the depth of the peat across the site. Therefore, two cores from the same class of microtopography sampled several hundred meters apart are likely to vary based on overall depth. Total depth and depth to Hekla 4 were determined by the PCA to be components in the factor which explained 55.29 % of the variance. Microtopographic variation may give a misleading impression of the total carbon stock as much of the upper sections of the peat profile are not a secure part of the long-term carbon store. The relocation of peat creates a longer peat column in the ridges, where the depth to Hekla 4 is greater than in the furrows from which peat is removed, with an associated larger stock of carbon (Table 5.2). While at the end of the first rotation this represents a large stock of dense peat carbon relative to the other afforested microtopographies, it would be expected to be lost to the atmosphere more rapidly over subsequent rotations than those other microtopographies. This is as the original surface and furrow peat has relatively less material above the water table and therefore has anoxic conditions over a greater proportion of the remaining peat column. It is a weakness of a stock-based approach that the current rates of oxidative loss are not measured, and therefore a key element of future work should be to incorporate flux data into carbon stock data in mature peat plantations.

The second major component of the carbon stock is the tree biomass, although uncertainty in this dataset was not as pronounced as in the peat results. A possible source of uncertainty in the tree carbon estimate arises from the relatively small destructive sample sizes. The low standard deviation and variation in the data on wood carbon content and fresh:dry ratios, even over the wide range of tree sizes captured in the destructive sample (Table 5.4), suggests that these values are broadly representative. A more likely source of uncertainty is in the below ground component of the biomass. Due to the restrictions discussed in the method section, woody root biomass was measured using relatively few samples, and in measuring the Lodgepole roots there may be an inherent underestimation due to damage to the root system as trees topple. This bias may therefore systematically underestimate the amount of carbon sequestered in roots as derived allometrically from the quantity of stem wood extracted from the site. How much such an underestimate impacts the overall budget is debatable as in a further rotation where the drainage status of the peat was maintained this biomass would be prone to rapid decay and a rapid return of carbon to the atmosphere.

The sampling strategy adopted was based on a series of transects around which data had already been collected that were then used as a framework for a design pairing undrained and afforested areas. This strategy does not represent a truly randomised design, which arguably could have produced a stronger data set that would have been more appropriate for statistical analysis. Such a randomised design would not have been possible at Bad a'Cheo due to the condition of the site (in particular issues with accessibility and wind-throw). The strategy chosen provided the best comparisons between sampling sites which were more likely to have had similar conditions prior to drainage, while also allowing for variations in microtopography to be accounted for. An additional uncertainty which serves to underline the poor conditions and wind-throw at Bad a'Cheo is in the 5.86 ha of the plantation where no commercial wood product was extracted, and all biomass was mulched and remained on site. Integrating this section of forestry into the overall budget, as a component which lost peat carbon but produced no wood products, would have shifted the budget into a larger net loss than was otherwise reported. It is reasonable to assume that the wood was of a poorer quality and lower yield if it was not selected for harvest, so the surveys of the standing wood in the plantation could not be directly applied to produce an estimate of biomass. No practical survey method within the scope of this study could properly account for biomass in fallen wood, nor were such areas directly surveyed as sites were chosen along transects that were selected in part to ensure that a practical sampling point with no wind-throw could be accessed. It is therefore better to exclude this section of the plantation for the analysis rather than introduce another source of uncertainty into the estimate. The data from the Bad a'Cheo New 2 transect and from the afforested sections of Forsinard and Broubster account for the extreme ends of the range of results. The New 2 transect and the Forsinard site both reported larger carbon accumulation on afforested than undrained bog, while Broubster showed an above average loss. These three components drive the wide variation observed across the study. New 2 was characterised by dry, well drained peat (which was shallower than at the other afforested sites and had a significantly higher bulk density) and significantly larger Sitka spruce trees than elsewhere. The dryness of the site suggests that although the carbon content was high, it may be more readily oxidised and released to the atmosphere were there to have been a second cycle of forestry. The high bulk density and shallowness of the afforested peat (Table 5.4) suggests that the drainage of the peat may have extended to below Hekla 4, suggesting that deep drainage may have resulted in additional losses of carbon below Hekla 4. At Forsinard, the paired coring sampling strategy may not have targeted peat with comparable Holocene carbon accumulation rates. There was only a 4 cm, difference in the total peat depths of the two Forsinard (Table 5.1). As forestry causes compression and compaction, and the bulk density at Forsinard shows that this has been the case here, it may be that Forsinard was initially much deeper in the afforested site. This suggests that the paired undrained and afforested sites may not have been comparable, which due to the heterogeneous nature of peatlands would be difficult to determine before coring. Conversely, the largest negative difference in carbon stocks between undrained and afforested peat was recorded in the Broubster cores. Here the preparation of the afforested site seems to have caused an unusually large loss of peat. The relative thinness of the peat below the Hekla 4 deposit also suggests that the accumulation rate has been relatively slow compared to other peat in the region. The younger basal date of the afforest section relative to the undrained (5149 ± 21 and 7263 ±34 uncalibrated years BP respectively) suggests a later initiation, or an unknown disruption of the peat post-initiation. The variable results seen for these samples emphasises the importance of replication and increasing sample size should be a key objective of any further work undertaken.

There are reasons to expect that net carbon loss from the Bad a'Cheo will have been larger than measured here. This study does not consider areas which were outside the forest blocks but affected by the forestry drainage system. Areas peripheral to planting are known to have a significantly reduced peat depth (Sloan *et al.* 2019) and effects of drainage may extend between 40 to 100 m from the forest blocks (Shotbolt *et al.* 1998, Lindsay 2010). In these areas a lowered water table may also expose more peat to oxidative loss of carbon.

The average loss of carbon out of the system is greater than that reported in peat in other afforested regions, such as those elsewhere in Europe. Fennoscandian peatland afforestation has certain qualities that distinguish it from the afforestation of the blanket bogs of the northern British Isles. In Fennoscandia, forestry is a more well-established practice on fen peats, which may already be naturally treed/forested (Laiho & Laine 1997). While there are some plantations in open sites, in the already afforested areas relatively little work is required to bring these areas into commercial production (Minkkinen *et al.* 2002, Maljanen *et al.* 2010), and generally the wood produced is of high quality. Within the UK, the tax incentives for afforestation meant that the land selected was extremely marginal, often deep peat blanket bogs. The level of preparation required, and the resulting low quality of the wood produces are likely to give widely different carbon loss and capture outcomes than elsewhere.

Fennoscandian peatland afforestation is has been generally shown to provide a net boost in overall carbon sequestration, with a wide range of possible changes to peat carbon (from gains to losses) compensated by a larger uptake of carbon in tree biomass. Analyses by Minkkinen *et al.* (2002) and Turunen (2008) both consider the net change in Finnish peat carbon due to afforestation over different time scales and with different methods. While both studies ultimately conclude that the growth in wood biomass leads to a net uptake of carbon in these systems, their respective findings on peat carbon change encompasses the range of estimates found in the literature. For the 5.7 million ha of afforested peat in Finland, Minkkinen *et al.* calculate that 50 Mt C has been gained over a century (a figure in line with much of the twentieth century literature) while Turunen calculates a 73 Mt C loss over 50 years. Converting these values to changes in a single hectare over the 50-year period of growth of Bad a'Cheo, Minkkinen *et al.* would predict gain of 4.39 tonnes ha^{-1}, while Turunen would predict a loss of 12.81 tonnes ha^{-1}. The mean estimate of carbon lost in the peat component of Bad a'Cheo was 66.8 tonnes ha^{-1} over this period. This is of the same order of magnitude as the Simola *et al.* (2012) resurvey of previous coring sites, a study that provides the upper end of the range of estimates of Finnish peat loss (approximately 75 tonnes ha^{-1} over fifty years if the data is converted), a figure large enough to arguably be an outlier relative to the rest of the Fennoscandian literature.

Fennoscandian studies generally show a net carbon accumulation regardless of loss of carbon from peat due to production of high quality hard wood products (Minkkinen *et al.* 2002, Drosler *et al.* 2008, Ojanen *et al.* 2013). Such wood products ensure that carbon remains in tree biomass for long periods, as opposed to lower quality wood products intended for fuel use. This raises the question of the ultimate fate of the carbon stored in tree biomass in these deep peat forestry plantations, although it is important to note that for carbon accounting purposes this loss would not be considered to be from the plantation itself, but rather is "embedded" in the emissions from the industries that use the timber. The majority of the Bad a'Cheo biomass was earmarked for uses which will return carbon to the atmosphere rapidly, in this case fuel wood and power generation. That said, these emissions are likely to replace power generation through non-renewable fuel that would otherwise have taken place.

A significant volume of woody material was mulched and left to decay or be preserved (through integration onto the acrotelm) by new peat formation, as the site was transitioned to a wind farm. The fate of the carbon in this remaining biomass is dependent on the future drainage status of the peat. Should a subsequent forestry cycle have been undertaken this could have produced greater net emissions than in the first cycle though the continued decay of the dry material left on the ground at the cycle's start. As the site was re-wetted the brash was reprofiled (buried in the furrows as the plough throws are flattened). As with the roots, the rate of decay and carbon release of the remaining biomass will be related to future use of the site. The upper 30 cm layer of peat from afforested sites has been shown to have higher lignin and recalcitrant material concentrations and lower soluble component concentrations than peat from adjacent undrained sites, suggesting that forest-derived material is inherently less decomposable than material derived from bog vegetation (Hermans *et al.* 2019). Successful rewetting would be expected to increase the amount preserved in new peat and decrease carbon release. There have been no studies documenting emissions from peatlands transitioning from forestry to wind farm, which includes forest-to-bog restoration interventions. However, recent studies have shown that forest-to-bog restoration can return net GHG sinks within 10-15 years, but that sites under restoration can be net GHG sources following tree removal, with brash contributing to emissions (Hambley *et al.* 2018, Hermans *et al.* 2019). In other words, there may be an additional "legacy" loss associated with the decomposition of the remaining woody debris after the forestry has been removed.

In the context of the carbon loss demonstrated in this study, as well as the likely increase in GWP through other factors such as albedo, the planting of conifer forests on peatlands has broadly increased the GWP of the Flow Country. In some areas, the process of drainage may have led to the development of peat cracks which necessitate the more energy-intensive backfill trenching rewetting technique (Pyatt & John 1989). In such situations it may not be practical or desirable to fully restore peat to an undrained state. On those marginal peat areas that are considered for re-stocking but have not yielded high quality timber the first time around, other forestry practices could be considered. The development of native woodlands, or mixed native and non-native species in 'peatland edge woodland' (Forestry Commission Scotland 2015, 2016) might produce a better net carbon balance, and advantages for biodiversity. Policy decisions on the future of peatland afforestation must take into account the role that peat has had in absorbing carbon over millennia, and implement management in such a way that will not result in unintended consequences, such as increased carbon emissions.

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Chapter 6 - Discussion

Response to project objectives

The widespread afforestation of peat bogs in the UK in the second half of the twentieth century has been thought to have affected carbon storage in peat, but relatively little data existed on the extent of this impact. This thesis aimed to use paleoecological techniques to explore this current conservation problem. The introduction set out several general objectives:

1. This study will assess the current evidence base for the effects of peatland afforestation on peatlands, drawing comparisons between existing data in the UK and in other regions, in particular Fennoscandia.

2. It will quantify the changes at Bad a'Cheo, a peat bog which was drained and afforested in 1968. This will include the physical alteration of the landscape (the widespread subsidence in the ground level) and the changes to carbon stored on within the bog.

3. It will further quantify the carbon storage provided by tree biomass at Bad a'Cheo through direct measurement of tree morphometrics.

4. Using the peat and tree carbon data for the previous two objectives it will produce a carbon budget for Bad a'Cheo, using a stock-based approach.

5. It is important to ascertain whether the results taken from our intensively studied site are applicable throughout the Flow Country. The results from Bad a'Cheo will therefore be placed into a regional context. This will be achieved by carrying out a simplified version of the Bad a'Cheo sampling protocol on several other sites that form a rough transect across the Flow Country.

6. This study uses tephrochronology to identify isochrones in peat (Lowe 2011). It therefore has the additional aim of further developing the methodology for rapid identification of cryptotephra deposits using ITRAX scanning (Dugmore & Newton 1992, Kylander *et al.* 2012). It aims to greatly enhance the evidence base for the geochemistry in tephra deposits in northern Scotland, using a level of replication uncommon in other studies.

Literature review and identified gaps in the evidence base

The first objective was to assess the current evidence base for the effects of afforestation on peatlands, drawing comparisons between existing data in the UK and in other regions. Chapter 2 addressed this, discussing the history of deep peat afforestation in the British Isles, and placing it in the context of comparable forestry in Europe and Fennoscandia. A review of the available literature showed that there were few studies directly measuring carbon losses from UK afforested peat, and that many of the existing assessments assumed a similar carbon balance to that reported in Fennoscandia.

Key areas of concern identified in chapter 2 include the need for better data on the yields, quality and ultimate use of peat bog forestry plantations. It also concluded that whole-column inventories of carbon stocks should be made to provide peat carbon budgets over the life of a plantation. These conclusions informed the planning of the data collection in subsequent chapters.

Changes in peat depth and ground level under afforestation

Bad a'Cheo plantation in Rumster Forest, Caithness, was used as the primary, intensively studied site. Chapter 3 of the thesis addressed the physical changes to the peat within the plantation, using data from the Forestry Commission archives, previously published data, and new primary data collection. Together, these addressed the first component of objective 2, to quantify the physical changes at Bad a'Cheo.

Using GIS to re-interpolate previous data sets against a new series of ground level transects and peat depth probing, we demonstrated that the ground level had subsided by an average of 44.9 cm, and the peat had thinned by an average of 56.8 cm (13 %). This is to be expected by the compression and compaction caused by drainage and tree growth, but also indicated a possible oxidative loss of peat carbon. However, in order to fully disentangle the compaction effects from the carbon losses, confirmation via further direct measurement was required.

Carbon budgets for afforested and undrained peat

Chapter 5 addressed objectives related to changes in carbon stored within the bog at Bad a'Cheo (objective 2), quantifying the carbon storage provided by tree biomass (objective 3), and producing a carbon budget for the plantation (objective 4). It found an average of 0.67 g C cm^{-2} loss of carbon from Bad a'Cheo peat, which is equivalent of 66.8 t C ha^{-1}, with a total predicted loss of on site from peat of 1838.94 t C over 27.53 ha of afforested peat. Carbon uptake in tree biomass compensated for some of this loss of peat carbon, producing a net loss of carbon from areas with harvestable wood of 19.63 t C ha^{-1}. It should however be noted that these results were highly variable, with a large range of possible outcomes (including a net gain of carbon at the upper end of the estimate range).

Objective 5 was to determine whether the results taken from Bad a'Cheo are applicable elsewhere in the Flow Country. A simplified version of the Bad a'Cheo sampling protocol, using a single set of paired cores on four other sites was used. On average afforested peat contained 1.03 g cm^{-2} less carbon than in adjoining undrained bog (an average of 8.02 g cm^{-2} and 7.09 g cm^{-2} respectively), although the data was also highly varied across these samples. These results suggest that the findings from Bad a'Cheo are broadly applicable to other deep peat forestry plantations in the region. While the additional sites surveyed cover a range of peat depths and elevations, plus a mixture of plantation ages and tree mixes, a larger sample size is required to draw firm conclusions (see "Recommendations for future research"). The addition of tree biomass data would also allow a full assessment of an overall net change in carbon for a plantation.

The peat and wood data indicate net loss of peat carbon from Bad a'Cheo and most of the other sites, but there are additional factors which suggest that this may be an underestimation. The ground level survey found that that peat was significantly thinned in areas of open internal ground; those sections of the peat that were not directly afforested but may be impacted by the adjacent drainage. These areas extend between 40 and 80 m from edge of the forestry (Shotbolt *et al.* 1998, Lindsay 2010). Open internal ground would have been difficult to fully account for through additional coring as it represents a heterogeneous set of areas with a variety of characteristics. They range from well drained forest rides which exist in gaps of 50 m or less between forest stands to wider zones of more marginal hydrological impact. To survey them comprehensively was beyond the scope of this study, but if such areas were accounted for the net loss of carbon is likely to have been greater.

Peat carbon stock was calculated using a whole column inventory approach down to a stratigraphic marker, the Hekla 4 tephra. However rates of carbon accumulation in near-surface, recently formed peat may be unreliable as acrotelm peat will continue to decompose until it enters the catotelm. (Young *et al.* 2019). Other studies using similar methodologies (and incorporating some data from this site) have chosen not to include the top 50 cm of cores (Ratcliffe *et al.* 2018). But in the case of afforested peat, the redistribution of peat by ploughing means that top 50 cm of the forestry sites is dense former catotelm peat brought back to surface. Such disruption to the peat surface would make the removal of an arbitrary 50 cm unjustified.

Chapter 5 examines whole carbon stocks over the course of the first forestry cycle. While this is an effective approach to estimated carbon losses or gains in situ, some sources of greenhouse gasses that contribute to the overall “footprint” cannot be measured using a stock-based approach. Digging and maintaining drainage ditches, ploughing furrows for planting trees, putting up fencing, felling mature trees and transporting wood off site all require use of vehicles, from which carbon emissions will add to the total global warming potential (GWP) of forestry (Morison *et al.* 2012). As for most land uses of UK peatlands, data are lacking on operational and management emissions. While the contribution of such emissions will be relatively minor, comparisons of greenhouse gas balance among land uses would be improved by including such estimates, including emissions associated with the mechanical restoration of bogs (e.g. diggers for drain blocking and stump flipping) and where applicable to the management of protected sites. The use of heavy machinery also raises other factors such as additional compression which will further change characteristics of the peat (Heinemeyer *et al.* 2019).

While greenhouse gas emissions may be the most important factor in determining the likely impact on radiative forcing, there are also non-carbon related impacts. The carbon stock assessment in this paper does not consider sources of radiative forcing not related to carbon flux, such as albedo. Forest cover has been associated with decreased albedo, especially in high latitudes prone to snow cover (Betts 2000). Indeed when these other factors are considered the record of forestry in mitigating climate change is mixed (Naudts *et al.* 2016). The Scottish government aim to afforest 15,000 ha per year by 2024/25 (Scottish Government 2019) will undoubtably absorb carbon. Even on marginal sites such as Bad a’Cheo, although there was a net loss of carbon biomass in the harvested trees did account for around 681 tonnes of carbon sequestered. However, when other factors including changes in albedo are considered even forestry with a net uptake of carbon may still show an overall warming effect (Luyssaert *et al.* 2018).

While not strictly associated with the plantation in carbon accounting terms but rather the industry associate with the end use, it is relevant to consider the ultimate destination of the carbon stored in wood biomass. This will be based on the quality of the wood, and the section of the tree the carbon is stored in. Broadly speaking, there are three relevant sections; trunk, branch and root. Branches will typically be cut off and their fate depends on a range of factors including market demand, access and end-goal (e.g. full restoration or simply tree removal); at some sites, brash may be chipped at the road side and used for biofuel, in others it may be left in situ and will decompose over time (e.g. windfarm sites), or it may be incorporated in the peat with restoration techniques such as re-profiling (where micro-topographic features inherited from forestry practices are flattened out and furrows blocked with peat dams sometimes incorporating trunk base, brash and root plates). Stumps and roots will remain in the ground, the rate of decay depending again on wetness. The trunk is the commercial section of the crop, and carbon residency time in the wood product is strongly tied to the use the wood is put to (Thompson & Matthews 1989).

Rapid identification of tephra as a stratigraphic marker

Chapter 4 addressed objective 6, further developing the methodology for rapid identification of cryptotephra deposits using ITRAX scanning. The use of ITRAX in peat is still relatively novel, with potential problems of data reliability arising from the high water and organic matter content of peat. However recent work has shown that XRF geochemical data from ITRAX reliably produces representative data for many of the key elements in identifying tephra deposits, such as iron and titanium (Longman *et al.* 2019). This study has further demonstrated that ITRAX is a reliable tool for identifying the deposits of cryptotephra, especially those widespread deposits with the highest shard counts.

Cryptotephra is an important tool in paleoecology, but until recently has been limited in its usefulness by the time required to identify deposits. Core scanning with ITRAX significantly reduces this time requirement (Kylander *et al.* 2012). Core scanning is rapid process but subsequently identifying points of likely tephra deposition in the resulting data remains labour intensive. There is a balance to be struck between speed and detail when working with tephra ITRAX data. Automation of this process using signal to noise ratio analysis may improve the speed of tephra discovery, although the highly "noisy" nature of the data would restrict the usefulness of ITRAX to only the most abundant deposits of cryptotephra. The most effective, if not the fastest, approach is probably a mixed one combining data from several sources including visual assessment, X-radiography, XRF derived geochemistry, loss on ignition, and magnetic susceptibility.

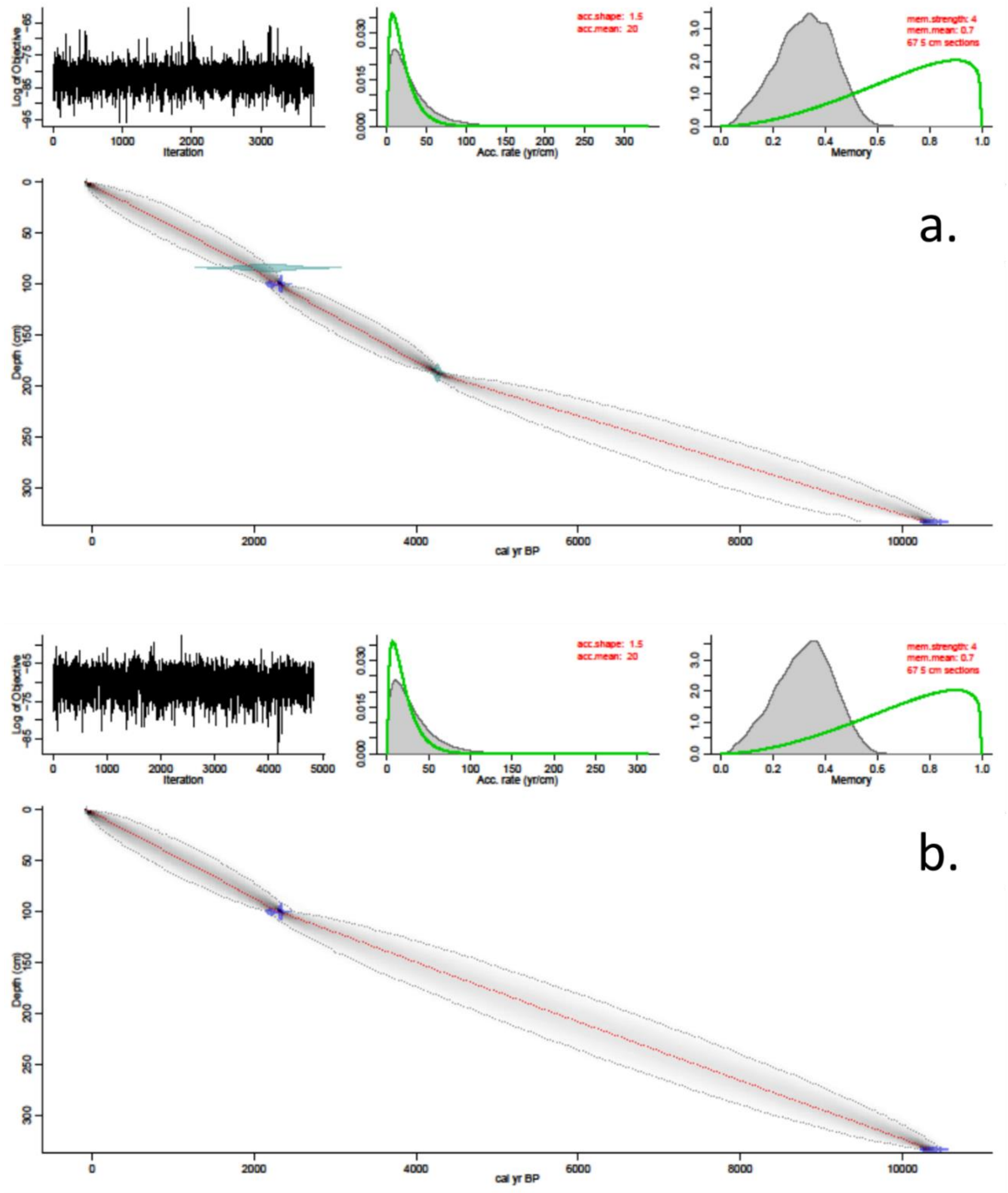

Figure 6.1. BACON derived age depth models (Blaauw & Christen 2011) for the Rosal undrained bog core. Model a. Shows the age depth with radiocarbon dating from 1 m depth and at the base. Model b. adds age points for deposits of Glen Garry (226 ± 244 BCE) and Hekla 4 (2310 ± 20 BCE).

The addition of identified tephra deposits allows additional reliably dated points to be added to age depth models (Figure 6.1). Tephra is also a good stratigraphic marker in itself (Lowe 2011), providing replicable points from which the peat carbon calculations were made. The usefulness of such a marker beyond very abundant deposits such as Hekla 4 (identified in 34 of 35 cores) is more limited, as it must be reliably located to allow for comparisons between samples. The variability in abundance (and therefore ease of identification), as well as the possibility that post depositional factors may change the position of tephra, mean that the usefulness is more situational than radiocarbon dating. Tephra deposits do however have the additional advantage of providing paleoecological evidence beyond dating. Tephra deposition has direct impact on plant communities (Blackford *et al.* 1992, Payne & Blackford 2008, Hughes *et al.* 2013) and therefore provides important context to a full understanding of the paleoecology of a site.

Finally, Objective 6 further intended to greatly enhance the evidence base for the geochemistry in tephra deposits in northern Scotland. Chapter 4 identified 34 deposits of Hekla 4 across five sites and included geochemical analysis of 25 deposits (producing 350 sets of EMP results). An additional two deposits of Glen Garry tephra were also found and geochemically analysed (23 EMP results). The EMP data gathered in this study (see ”supplementary material”) will be archived on Tephrabase (Newton *et al.* 2007).

Net change in carbon storage on afforested peat

The overarching aim was to provide evidence towards answering the question: *Has peatland afforestation in the UK led to a net uptake or loss of carbon?* These results suggest that there was a loss of carbon from the afforested peat at Bad a'Cheo, in line with the average losses from other surveyed sites. Sequestration of carbon in wood biomass only partially compensated for this loss, resulting in an overall net loss of 19.63 t C ha^{-1}.

The results presented in this thesis show a large net loss of carbon on the Bad a'Cheo plantation, and suggest that there may be losses of the same magnitude from other afforested deep peat sites around the Flow Country. While the afforestation of UK deep peats ended following the ending of tax incentives and a change in Forestry Commission guidance (Patterson & Anderson 2000, Oosthoek 2013), many UK afforested peatlands are nearing the end of the initial forestry rotation and will soon be felled. There is therefore a pressing need to decide on whether such afforested peatlands should be restocked for a new forestry rotation or restored through rewetting. The evidence presented in this thesis suggests that many deep peat forests have been a net source of greenhouse gas emissions. Future management of these sites should focus on minimising further loss of carbon. If sites were earmarked for second rotation, we argue that it would be essential to set up long-term programmes of monitoring of GHG to determine whether these sites continue to lose carbon or eventually stabilise or even switch back to being carbon sinks.

Within the context of the declared climate emergency, there is a strong international drive to focus on tree planting (Bastin *et al.* 2019) and there is potentially a risk if that strategy is adopted without consideration of soil carbon upon which the trees are planted. While the evidence presented here focussed on deep peat, planting on shallower peaty soils may also have long-term unintended consequences for carbon storage which must be considered and assessed properly (Friggens *et al.* 2020), and ideally before large-scale programmes are rolled out.

Advances in forestry technology allowed what had been very marginal land for silviculture to be widely afforested. The motivation of the foresters and policy makers was broadly to improve the domestic wood supply (Wood 1974) and to provide industry and jobs to a region which lacked economic development. However, these aspirations were tied to a tax system which did not take account of the suitability of the areas to be afforested, and tax efficiency became the motivation for many wealthy individuals who later became involved in peatland forestry (Mather 1986, Mather & Murray 1988). This led to planting on some deep peat areas which were not likely to yield a commercial crop, a problem which was particularly acute in the Flow Country (Warren 2000).

The initial debate focused on the disruption to the biodiversity, and the fragmentation of a large habitat (Stroud *et al.* 1987, Thompson 1987). The growing awareness of the importance of peatlands to carbon sequestration and storage has added a new consideration to how these landscapes should be managed (Joosten *et al.* 2012). Scotland is committed to increasing forestry coverage to 21 % of its total land area by 2032 (Scottish Government 2019), and to restoring 40 % of the estimated 600,000 hectares of damaged peatlands by 2030 (Scottish Government 2017). In some areas of afforested deep peat these are mutually exclusive goals, but site-specific assessments and mixed approaches to restoration may provide a way forward. There are three main options for the future of deep peat plantations in the UK: re-stocking, restoration or a mixed approach using “edge woodlands”.

In areas where timber quality is good and of a high yield class (≥ 8), and the site needs minimal extra inputs, guidance recommends restocking with similar trees (Forestry Commission Scotland 2015). This is more likely to be the case on sites with shallower peat, where drainage has been the most successful, and the peat is driest (Payne *et al.* 2018, Vanguelova *et al.* 2018). Here it is possible that high quality wood products may be more effective in offsetting the carbon loss from peat over several forestry rotations. More survey data is needed to determine which afforested peatlands are likely to produce high yield class crops.

Restoration is already underway in some areas. Restoration is the default for designated conservation areas (Natura 2000 sites, SSSIs, etc.), where restocking would affect the hydrology of adjoining peat, where restoring would prevent greenhouse gas release (Forestry Commission Scotland 2015). Restoration is achieved through rewetting of peat and the removal of trees. In early efforts at restoration, in plantations where trees were still relatively small, they were felled and left on site. Now that plantations are more mature tree stems are felled and removed, but tree stumps are flipped and buried with brash in a reprofiled landscape, (Andersen *et al.* 2017), with some whole trees also be mulched. Such techniques may aid the restoration process in terms of water chemistry, which otherwise has been found to have not completely recovered in up to 17 years (Gaffney *et al.* 2018). Burial of wood on a rewetted site is an effective means of ensuring that the carbon in the wood is not oxidised to the atmosphere (Zeng 2008). Studies on restoration have shown that water tables can be restored, and subsidence is partly reversible (Anderson & Peace 2017). It has already been established that forest-to-bog restoration has the potential to back net climate benefits, with sites returning to being net CO_2 sinks within 6 (raised bogs) to 15 (blanket bogs) years after tree removal (Hermans *et al.* 2018, Creevy *et al.* 2019). However, restoration is a slow process which can take in excess of many decades to return peatlands to a fully functional state, supporting specialist species, sequestering carbon and forming new peat. Sites may also require continued investment and maintenance, as regeneration with new growth of trees is initially a problem.

In some areas a partial restoration mixed with the maintenance of some trees may be the most appropriate future management option. One such proposal is to develop "edge woodlands", which would maintain a low density tree cover with approximately 50 % planted, 50 % open ground (Forestry Commission Scotland 2016) on the margins of restoration areas. This would be achieved using a combination of the regenerated conifers which are common on felled plantations, supplemented with plantings of a mix of native species. Such an approach would theoretically allow a degree of rewetting and resumption of peat carbon sequestration, increase biodiversity, contribute to national forestry targets, and avoid some of the costs of full restoration. The viability of this approach, and whether it can meet all these targets, is a key question in the future of afforested peatland restoration and management (Payne & Jessop 2018).

The evidence presented in this thesis suggests that a net loss of carbon has occurred at Bad a'Cheo and the dense catotelm peat near to the surface in the plantation was likely to become an ever-greater source of carbon release in a second rotation. This suggests that the rewetting that was carried out at Bad a'Cheo following felling was the best strategy to minimise further carbon loss. More generally, different management approaches will be appropriate in different circumstances, but due to the high variability in the responses of peat to afforestation and in the yield of peatland forestry the approach taken should be determined by site specific evidence. Further uncertainties on the best approach are exacerbated by the predicted impacts of climate change. Modelling predicts that under some future climate scenarios some peatlands may switch too being net sources of carbon emissions (Ferretto *et al.* 2019). This is in addition to carbon lost through land use changes of the type described here.

Recommendations for future research

There are several sources of uncertainty in the dataset, and a lack of further data on the fate of carbon under future land use. Future work should address these issues and the other evidence gaps outlined below.

1. A weakness of this data set was the broad range of possible outcomes for carbon storage caused by the high variation in the data, both between sites and within the microtopographic replicates at Bad a'Cheo. More replication of the paired core design at other afforested sites would allow for a stronger analysis and the potential identification and exclusion of outliers.

2. A key uncertainty in this dataset is around the fate of the carbon stored in the tree biomass. At Bad a'Cheo the wood that was not mulched on site was known to have been used in industries (chiefly wood fuel pellet production) that would return carbon to the atmosphere quickly. A broad survey of the fate of wood products from deep peat plantations would increase the accuracy of carbon budgets and would allow for a clear assessment of differences in the carbon budgets of deep peat forestry and plantations on mineral soils.

3. The dataset in this thesis measures the net change in carbon stock under afforestation. It does not detail the pathways to the atmosphere of carbon (gaseous or aqueous) or the relative importance of these components. The analysis would benefit from integration of stock surveys with gas flux measurements and water samples to provide data for both net change and to indicate rates of change for mature plantations. This is particularly relevant for any second forestry rotation, where carbon flux from surface peat is expected to be higher.

4. Afforested deep peat may be difficult to restore to a pre-drainage state. Whether extensively drained and cracked peat will revert to a net sink of carbon with comparable

yearly accumulation rates undrained pristine bog is not certain. Long term systematic monitoring of the outcome on restoration strategies in terms of carbon sequestration is required (in addition to species and water quality monitoring). Such monitoring needs to consider the fate of buried wood on bogs which have been reprofiled to integrate the non-commercial forest biomass into the peat. A combined approach that emphasises flux studies in years immediately following restoration, and stock studies over decadal scales will produce the most comprehensive datasets.

5. The ITRAX rapid core scanning technique made the level of replication used in this study possible. However, the cryptotephra analysis still involved visually identifying overlaps of peaks of several indicator elements and matching those elemental profiles to LOI data. It would be useful to further automate the process of identification, possibly using the signal to noise ratio. Rapid identification of tephra in peat through ITRAX is a relatively novel technique, but as it is increasingly adopted datasets containing XRF elemental analysis, LOI, and tephra data (from microscopy and geochemical analysis) should become available to develop techniques for automation.

Summary of key messages to stakeholders

The findings of this thesis have implications for several groups involved in the study and conservation of peatlands, particularly within the Flow Country of northern Scotland.

1. Governmental and regulatory bodies

   The results of this study suggest that the afforestation of the Flow Country has resulted in a net loss of carbon from peat, although the data were highly variable. Many of the Scottish deep peat plantations are now approaching harvesting age, and decisions must soon be made about whether to replant for a second rotations or to restore these areas. Because of the wide variability of the data a broader survey of additional deep peat plantations in the Flow Country would improve the quality of the analysis. Such work should integrate stock and flux assessment and should wherever possible underpin decisions about the future management.

   More generally, the peatland afforestation described in this thesis took place largely because of tax incentives offered for forestry, rather than the predicted commercial viability of the plantations. In addition to the impact on peat carbon stock, there was a large disruption to species assemblages and other ecosystem services, the restoration of which is still an ongoing challenge (Hancock *et al.* 2020). While public policies promoting afforestation may improve carbon sequestration if suitable areas are selected (Matthews *et al.* 2020), forestry schemes must be designed to prevent exploitation and be monitored for efficacy.

2. Management interests (conservation and forestry organisations)

This study describes the net changes in carbon stock over an initial forestry rotation of a peatland plantation, suggesting that there has been a net loss of carbon over fifty years. The fate of the near-surface peat carbon in such plantations remains uncertain, based on future management. Should these plantations remain drained for a second forestry rotation, oxidative carbon losses are likely to be larger than in the first rotation. Restoration should therefore be the preferred outcome for deep peat plantations, but under any future management regime long term monitoring is vital.

3. Scientific interests

This assessment has shown that cryptotephra is a viable stratigraphic marker for constructing comparative assessments of carbon stock in peat. This underlines that paleoecological methodologies can not only provide information on the history of a site but that such data can be used to inform decisions on modern land use. The use of paleoecological techniques should therefore be considered in designing monitoring and conservation studies.

Much of the work of this study, in particular the description of ground level and peat depth change, was dependent on the archive of data available from previous work by the Forestry Commission at Bad a'Cheo. Summaries of this data had previously been published (Pyatt et al. 1992, Shotbolt et al. 1998) but would not have been sufficient to create the new GIS interpolations used in chapter 3. This underlines the importance of archiving previous research, and of having mechanisms available to access such data.

General Summary

Peatlands are important terrestrial stores of carbon, holding over 2300 Mt carbon in the UK (Billett *et al.* 2010). Afforestation on deep peat threatens this carbon store through the lowering of the water table and subsequent oxidative decomposition of peat. At Bad a'Cheo plantation, a fifty-year forestry rotation on deep peat caused ground level subsidence, thinning of peat, and a net loss of carbon. The sequestration of carbon in wood biomass did not completely compensate for this loss, and on average 19.49 tonnes of carbon was lost per hectare.

Other afforested bogs showed a comparable average net loss of carbon from peat, although this was highly variable. It is therefore likely that the yield of wood from the first rotation in many deep peat plantations will not offset the carbon lost. This could have major implications for meeting carbon emission reduction targets in the UK and be a source of global climate forcing. This study suggests that in deep peat plantations restocking with commercial stands of non-native conifers may not be appropriate.

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Supplementary material

Electron microprobe specifications

All analyses were undertaken on a five-spectrometer Cameca SX100 electron microprobe at the School of GeoSciences at the University of Edinburgh and analysed using the wavelength-dispersive method. An accelerating voltage of 15kV and a beam current of 2 nA (Na, Mg, Al, Si, Ca, Fe and K) and 80 nA (Ti, Mn and P) were used with a beam diameter of 8 um. These conditions allow for good precision on the elements present in small abundances, but reduces Na mobility. All abundances are shown as oxides and FeO as total iron. A BCR2g glass standard and piece of Lipari obsidian were analysed to establish instruments stability and accuracy during the analyses.

Electron microprobe data

Supplementary data table 1. EMP data for Hekla 4 tephra deposits with site designations.

Site	Na_2O	MgO	Al_2O_3	SiO_2	K2O	CaO	P_2O_5	MnO	FeO	TiO_2	Total
Bad a'Cheo	4.7721	0.8937	14.909	62.817	1.7097	4.5666	0.3087	0.2987	9.5002	0.8464	100.62
Bad a'Cheo	4.7267	0.1756	14.02	69.057	2.3074	3.0158	0.0546	0.2126	5.7457	0.3586	99.674
Bad a'Cheo	4.8145	0.2546	14.039	67.12	2.1039	3.7147	0.0936	0.2517	7.3039	0.4521	100.15
Bad a'Cheo	5.0758	0.1548	12.649	73.829	3.98	0.5306	0.0266	0.1119	2.182	0.3329	98.872
Bad a'Cheo	4.2128	0.5215	14.376	64.442	1.7	4.3354	0.1736	0.2786	8.4024	0.6424	99.084
Bad a'Cheo	4.8573	0.0291	12.714	75.215	2.7463	1.3935	0.0101	0.0783	2.1459	0.0895	99.279
Bad a'Cheo	4.5243	0.6694	14.208	64.117	1.8025	4.3503	0.2161	0.2892	9.0151	0.7126	99.905
Bad a'Cheo	4.5948	0.6157	14.753	64.392	1.8755	4.3857	0.2159	0.3033	8.6987	0.7027	100.54
Bad a'Cheo	4.7843	0.0764	13.904	69.093	2.1468	2.8623	0.0414	0.1991	5.022	0.2807	98.411
Bad a'Cheo	4.6023	0.0333	12.915	74.461	2.8509	1.3842	0.0051	0.096	2.0811	0.1029	98.532
Bad a'Cheo	4.5732	0.0132	12.371	73.323	2.973	1.3264	0.0122	0.0723	1.8691	0.0945	96.629
Bad a'Cheo	4.3352	0.0369	12.225	73.86	2.861	1.3803	0.0003	0.0616	1.7581	0.0748	96.594
Bad a'Cheo	4.4669	0.9432	14.819	62.326	1.7319	4.751	0.2858	0.2822	9.5846	0.8246	100.02
Bad a'Cheo	4.7634	0.0287	12.639	73.416	2.9642	1.3246	0.005	0.0702	1.933	0.0959	97.24
Bad a'Cheo	4.7075	0.0172	12.833	75.538	2.704	1.2962	0.0103	0.087	1.6759	0.0822	98.951
Bad a'Cheo	5.249	0.0415	13.835	70.903	2.4972	2.5111	0.02	0.1606	4.1201	0.229	99.566
Bad a'Cheo	5.0756	0.0904	14.106	69.134	2.2939	2.8412	0.0314	0.1984	4.9191	0.2631	98.953
Bad a'Cheo	4.5903	0.8175	13.947	62.45	1.7036	4.5115	0.2334	0.3005	9.341	0.7548	98.65
Bad a'Cheo	4.8812	0.142	14.272	69.663	2.1748	3.1329	0.0555	0.1987	5.3078	0.3531	100.18
Bad a'Cheo	4.5985	0.0252	12.891	72.854	2.828	1.4022	0.0113	0.0802	2.0006	0.0927	96.783
Bad a'Cheo	5.0678	0.4048	13.829	64.849	1.9006	4.0478	0.1189	0.2858	7.976	0.5443	99.024
Bad a'Cheo	4.2543	0.0015	12.525	75.947	3.1695	1.1127	0.0005	0.07	1.9973	0.1029	99.177
Bad a'Cheo	8.2728	0.0103	23.611	61.704	0.4942	5.4218	0.002	0.0046	0.183	0.0077	99.711
Bad a'Cheo	4.7577	0.4753	14.628	65.109	1.8569	4.0915	0.1749	0.2961	8.3229	0.6331	100.35
Bad a'Cheo	4.7596	0.0578	13.415	72.937	2.7579	1.9258	0.0261	0.1559	3.8988	0.2171	100.15
Bad a'Cheo	4.69	0.0055	12.365	74.776	3.0025	1.1465	0.0098	0.0598	1.3554	0.0884	97.498
Bad a'Cheo	4.4729	0.0235	12.632	73.22	3.033	0.9639	0.0058	0.0633	1.4781	0.0861	95.978
Bad a'Cheo	5.657	0.3618	17.912	63.315	1.0752	5.4938	0.1633	0.1906	5.6842	0.5426	100.4

Bad a'Cheo	4.5984	0.6165	14.65	64.275	1.8037	4.3765	0.1953	0.2871	9.3273	0.7142	100.84
Bad a'Cheo	4.8094	0.145	14.597	68.343	2.1708	3.4583	0.0578	0.23	6.4361	0.3645	100.61
Bad a'Cheo	4.6753	0.0053	12.854	73.846	3.0102	1.2839	0.0037	0.0809	1.8569	0.0924	97.708
Bad a'Cheo	4.8779	0.1335	14.171	68.707	2.1547	3.1047	0.0364	0.2167	5.7958	0.3362	99.533
Bad a'Cheo	4.5795	0.0032	12.892	74.498	2.9732	1.3039	0.0064	0.0837	1.8743	0.0916	98.306
Bad a'Cheo	5.6471	0.0053	14.337	73.434	2.3513	1.6359	0.0079	0.0671	1.4283	0.0704	98.984
Bad a'Cheo	4.6844	0.0102	12.745	73.922	2.833	0.9671	0.0045	0.0501	1.54	0.0992	96.856
Bad a'Cheo	4.6109	0.0185	12.753	73.996	2.7833	1.3142	0.0026	0.088	1.8695	0.0957	97.532
Bad a'Cheo	4.555	0.5822	14.558	64.255	1.7316	4.2735	0.1431	0.2966	8.5838	0.6572	99.636
Bad a'Cheo	4.75	0.0174	13.128	75.833	3.0592	1.3325	0.0058	0.0842	2.0775	0.0964	100.35
Bad a'Cheo	4.1001	0.0139	12.184	71.832	2.9118	1.1737	0.0048	0.0888	1.74	0.0924	94.141
Bad a'Cheo	4.386	0.0158	12.166	74.218	2.7699	1.4627	0.0108	0.085	2.0867	0.0868	97.287
Bad a'Cheo	4.9976	0.0323	14.278	69.684	2.3989	2.2858	0.0114	0.1362	3.5593	0.2033	97.586
Bad a'Cheo	4.3661	0.0204	12.32	72.625	2.8659	1.2972	0.0087	0.0908	1.9631	0.0896	95.647
Bad a'Cheo	4.4123	0.0024	12.642	73.153	2.8705	1.3068	0.0042	0.095	2.0233	0.099	96.6
Bad a'Cheo	4.4854	0.0167	12.911	74.075	3.0352	1.3456	0.0124	0.0825	2.0612	0.0935	98.118
Bad a'Cheo	4.3969	0.02	11.742	72.488	3.1725	1.0602	0.0142	0.1117	1.7446	0.0939	94.844
Bad a'Cheo	5.0318	0.2279	13.27	69.861	1.9594	2.9069	0.0151	0.1193	3.795	0.1858	97.372
Bad a'Cheo	4.1895	0.7588	13.843	62.822	1.8392	4.4849	0.2184	0.3026	9.2403	0.7104	98.409
Bad a'Cheo	4.7019	0.0214	11.737	72.4	2.9284	1.2533	0.0038	0.0904	1.8911	0.1001	95.127
Bad a'Cheo	4.588	0.0171	11.977	73.966	2.9298	1.3136	0.0014	0.0888	1.9249	0.0975	96.904
Bad a'Cheo	4.6737	0.0255	11.851	71.28	2.9663	1.2633	0.0073	0.0779	2.0589	0.087	94.29
Bad a'Cheo	4.7934	0.0077	11.581	71.992	2.842	1.3905	0.0052	0.0692	2.0649	0.0856	94.832
Bad a'Cheo	5.3804	0.0045	13.104	74.431	2.6706	1.5563	0.0125	0.0515	1.9265	0.0748	99.212
Bad a'Cheo	4.6708	0.0279	11.629	72.713	2.8423	1.4541	0.0007	0.0633	1.9961	0.0976	95.494
Bad a'Cheo	4.1361	0.2775	13.731	66.343	2.0149	3.7292	0.0985	0.2727	6.9859	0.4587	98.047
Bad a'Cheo	4.4601	0.3275	13.77	66.089	1.9844	3.7955	0.1274	0.2714	7.4797	0.4891	98.794
Bad a'Cheo	3.8645	0.0019	10.505	76.177	2.9728	0.773	0.0064	0.0569	1.1909	0.0912	95.639
Bad a'Cheo	4.3643	0.0199	12.935	72.539	2.7577	1.3484	0.0027	0.0854	1.8737	0.0949	96.021
Bad a'Cheo	4.6998	0.3615	13.876	65.007	1.9719	3.7777	0.1316	0.2723	7.7785	0.5555	98.432
Bad a'Cheo	4.2662	0.4723	13.446	64.618	1.8482	4.3043	0.1414	0.2965	9.0055	0.6021	99
Bad a'Cheo	4.0072	0.7103	14.274	62.98	1.7171	4.3299	0.2122	0.3095	9.1291	0.7136	98.383
Bad a'Cheo	4.2523	0.794	13.559	63.228	1.676	4.6521	0.2084	0.3133	10.11	0.7406	99.533
Bad a'Cheo	4.7915	0.0034	11.607	72.007	2.8273	1.2573	0.0041	0.0879	1.9796	0.0929	94.649
Bad a'Cheo	4.5856	0.7789	13.68	62.861	1.8227	4.4665	0.2302	0.2751	8.7871	0.7752	98.262
Bad a'Cheo	4.3296	0.015	12.204	72.969	2.8873	1.3223	0.0094	0.068	1.9383	0.0893	95.832
Bad a'Cheo	4.3457	0.5613	13.654	63.399	1.8081	4.2314	0.171	0.3002	8.5565	0.6491	97.676
Bad a'Cheo	4.7077	0.0024	11.747	73.399	3.1455	0.9942	0.0086	0.099	1.8822	0.0991	96.067
Bad a'Cheo	5.1492	0.6409	15.03	62.815	1.7699	4.4449	0.2003	0.3112	9.0414	0.7037	100.11
Bad a'Cheo	4.7292	0.0146	13.509	75.174	2.8455	1.376	0.0114	0.0772	2.0105	0.0882	99.835
Bad a'Cheo	4.7856	0.9143	14.74	62.944	1.693	4.6531	0.2689	0.2931	9.596	0.8085	100.7
Bad a'Cheo	4.8203	0.6327	14.113	63.465	1.922	4.429	0.1844	0.2979	9.4546	0.6848	100
Bad a'Cheo	5.219	0.0515	14.218	70.782	2.3257	2.4117	0.0338	0.1599	3.9754	0.2346	99.412
Bad a'Cheo	4.5748	0.6332	15.315	63.375	1.713	4.4985	0.2088	0.2792	9.1316	0.6894	100.42
Bad a'Cheo	4.7493	0.7179	15.624	63.395	1.7904	4.3799	0.2028	0.2814	9.1263	0.7265	100.99
Bad a'Cheo	5.1877	0.0321	11.828	73.462	2.6387	1.2931	0.0041	0.0823	1.6166	0.0743	96.219
Bad a'Cheo	4.7829	0.0208	12.179	72.76	3.1303	1.0398	0.0107	0.1009	2.0467	0.0931	96.164
Bad a'Cheo	5.5056	0.5999	14.204	63.154	1.9202	4.2793	0.1507	0.3033	10.027	0.6977	100.84
Bad a'Cheo	4.2985	0.0055	12.368	74.29	3.132	1.0488	0.0139	0.0801	1.7583	0.0837	97.079
Bad a'Cheo	5.4121	0.2404	15.394	65.887	2.2102	3.5451	0.0798	0.2541	6.8792	0.4402	100.34
Bad a'Cheo	4.6342	0.6363	15.435	62.994	1.724	4.4404	0.1915	0.3021	8.7355	0.6759	99.769
Bad a'Cheo	5.2935	0.009	13.502	73.065	2.9774	1.0999	0.0036	0.0782	1.5931	0.0946	97.716
Bad a'Cheo	4.9066	0.4572	14.753	64.789	1.7411	4.456	0.0902	0.2967	8.1575	0.5107	100.16
Bad a'Cheo	5.0167	0.0366	14.556	71.079	2.422	2.0386	0.0282	0.1628	3.6575	0.2354	99.233
Bad a'Cheo	5.4612	0.1092	14.446	68.465	2.2553	2.8488	0.0235	0.1941	4.8598	0.2909	98.953
Bad a'Cheo	5.8723	0.0387	14.518	71.385	2.3322	1.9622	0.0052	0.0668	1.5082	0.0621	97.751
Bad a'Cheo	5.1168	0.1359	14.977	69.047	2.1648	3.1449	0.0427	0.2085	5.4447	0.3583	100.64

Bad a'Cheo	5.2408	0.2497	14.585	65.476	2.0529	3.442	0.0895	0.2616	6.9057	0.4385	98.742
Bad a'Cheo	5.0325	0.6363	14.909	62.648	1.8169	4.5047	0.1776	0.305	9.4061	0.6659	100.1
Bad a'Cheo	4.9254	0.2691	14.115	65.966	2.0934	3.5558	0.0898	0.2671	7.1839	0.4641	98.93
Bad a'Cheo	4.5813	0.6968	15.096	63.203	1.7277	4.5375	0.1933	0.2805	9.5426	0.7188	100.58
Bad a'Cheo	4.8711	0.5186	15.021	63.67	1.7783	4.3148	0.1393	0.2892	8.8379	0.6208	100.06
Bad a'Cheo	5.3471	0.0355	13.651	75.013	2.8173	1.399	0.0128	0.0877	2.2642	0.0975	100.73
Bad a'Cheo	5.3825	0.0788	15.409	71.097	2.2197	2.7987	0.0127	0.1808	3.2547	0.22	100.65
Bad a'Cheo	4.8408	0.8421	15.153	62.544	1.699	4.4647	0.2255	0.2665	9.4356	0.8066	100.28
Bad a'Cheo	4.8948	0.0277	13.394	74.012	2.9147	1.2953	0.0038	0.0762	1.9917	0.0875	98.698
Bad a'Cheo	4.9206	0.809	15.604	63.639	1.7232	4.5806	0.2111	0.2848	9.551	0.7447	102.07
Bad a'Cheo	5.1884	0.0282	14.625	70.705	2.4655	2.3579	0.0285	0.1577	4.0018	0.2068	99.765
Bad a'Cheo	4.9911	0.0048	12.512	74.481	2.7695	1.3417	0.0087	0.0807	1.951	0.0831	98.223
Bad a'Cheo	4.8313	0.5657	14.846	63.817	1.7535	4.4811	0.183	0.2966	8.9318	0.667	100.37
Bad a'Cheo	4.7944	0.4629	13.965	62.172	1.7016	4.0048	0.168	0.2864	8.3938	0.5925	96.542
Bad a'Cheo	4.3178	0.5688	13.85	62.598	1.7659	4.3918	0.1688	0.3066	8.734	0.6292	97.33
Bad a'Cheo	4.7364	0.0129	11.677	72.214	2.8208	1.2843	0.0252	0.081	1.8783	0.0853	94.815
Bad a'Cheo	4.3704	0.0077	11.9	71.915	2.7453	1.3401	0.0121	0.0717	1.9129	0.0879	94.364
Bad a'Cheo	5.3919	0.2029	14.598	66.661	2.0492	3.3978	0.0781	0.2449	6.345	0.4213	99.39
Bad a'Cheo	4.8712	0.575	14.912	64.102	1.8387	4.347	0.1819	0.3025	8.8011	0.6837	100.61
Bad a'Cheo	4.9408	0.2474	13.168	65.81	1.991	3.4652	0.0665	0.249	6.7957	0.4341	97.168
Bad a'Cheo	4.1508	0.669	13.121	62.604	1.8107	4.4073	0.2248	0.2747	9.333	0.7144	97.31
Bad a'Cheo	4.8303	0.0214	12.352	73.476	2.7545	1.3605	0.0177	0.0749	1.803	0.1027	96.792
Bad a'Cheo	4.9114	0.0132	12.503	71.13	2.7685	1.206	0.0633	0.0956	1.9576	0.085	94.734
Bad a'Cheo	4.875	0.0382	12.797	73.678	2.7561	1.2831	0.0125	0.0655	1.7351	0.0785	97.319
Bad a'Cheo	4.9509	0.0318	12.264	74.168	2.8688	1.2739	0.0041	0.0809	2.0275	0.0875	97.757
Bad a'Cheo	4.4278	0.3062	13.578	65.846	1.9533	3.6596	0.0764	0.2559	7.512	0.4976	98.113
Bad a'Cheo	9.6291	0.0057	20.372	64.402	0.1266	3.4375	0.0032	0.0013	0.013	0.0008	97.989
Bad a'Cheo	5.0104	0.0216	11.835	74.068	2.8871	1.3575	0.0087	0.0885	1.8501	0.0883	97.216
Bad a'Cheo	4.6356	0.0262	12.375	72.137	2.8186	1.277	0.0393	0.0775	1.8771	0.0852	95.348
Bad a'Cheo	4.4831	0.0008	12.26	72.996	2.8883	1.3101	0.0118	0.0779	1.9309	0.0964	96.055
Bad a'Cheo	4.672	0.001	12.86	73.444	3.1521	1.1746	0.0118	0.0811	1.762	0.0997	97.258
Bad a'Cheo	4.5051	0.0155	12.626	73.962	3.121	1.2318	0.0063	0.0776	2.0459	0.0941	97.685
Bad a'Cheo	5.0768	0.2923	13.927	64.685	2.0772	3.7347	0.0956	0.2369	7.1138	0.4484	97.687
Bad a'Cheo	4.6792	0.0209	12.12	70.076	2.6764	1.6963	0.0084	0.1163	2.5867	0.1437	94.123
Bad a'Cheo	4.5295	0.0127	11.724	73.225	3.0808	1.2163	0.0034	0.0732	1.7269	0.0991	95.691
Bad a'Cheo	5.1055	0.1839	13.444	67.422	2.2607	3.2287	0.0666	0.2135	6.0413	0.3745	98.34
Bad a'Cheo	4.078	0.0374	10.939	71.113	2.7612	1.2359	0.0134	0.0616	1.4574	0.0768	91.774
Bad a'Cheo	4.495	0.0054	12.263	72.677	2.7797	1.2081	0.0038	0.0846	1.8542	0.0901	95.461
Bad a'Cheo	4.5978	0.922	13.658	62.39	1.6921	4.5933	0.2681	0.2926	9.7144	0.7825	98.911
Bad a'Cheo	4.6614	0.013	12.149	74.305	2.8346	1.1338	0.0042	0.0839	1.5651	0.0829	96.833
Bad a'Cheo	4.4835	0.5474	12.784	65.994	2.5468	3.1637	0.3327	0.1801	7.9043	0.9395	98.875
Bad a'Cheo	4.3163	0.8816	13.981	62.731	1.6667	4.6453	0.2767	0.3157	9.3098	0.8099	98.933
Bad a'Cheo	5.0378	0.5447	12.684	62.892	1.712	4.408	0.164	0.2671	8.7162	0.6908	97.116
Bad a'Cheo	5.2884	0.0331	12.228	74.132	2.8189	1.4517	0.0038	0.0701	2.0277	0.0913	98.145
Bad a'Cheo	4.4312	0.0128	12.064	71.229	2.9534	1.3411	0.0038	0.0637	1.8805	0.0886	94.068
Bad a'Cheo	4.7961	0.4357	13.835	63.749	1.8282	4.0374	0.1361	0.2928	8.0907	0.5861	97.787
Bad a'Cheo	4.281	0.6645	14.011	62.526	2.3556	4.2952	0.2147	0.2911	9.4578	0.7075	98.805
Bad a'Cheo	4.6609	0.0386	13.416	70.043	2.4937	2.2012	0.0192	0.1317	3.4518	0.1958	96.651
Bad a'Cheo	5.136	0.15	13.941	68.651	2.158	2.899	0.0511	0.1889	5.3469	0.3166	98.839
Bad a'Cheo	5.3609	0.1909	13.82	66.764	2.1347	3.0289	0.0993	0.2188	5.588	0.3279	97.533
Bad a'Cheo	5.3213	0.0069	14.427	71.381	2.5097	2.0596	0.021	0.1273	3.3213	0.1751	99.351
Bad a'Cheo	4.4997	0.0235	12.574	72.864	2.8026	1.3591	0.0135	0.0641	1.7409	0.0837	96.025
Bad a'Cheo	5.0124	0.5703	14.639	63.322	1.7279	4.471	0.1833	0.3022	8.4801	0.6742	99.383
Bad a'Cheo	4.9273	0.1556	14.973	67.14	2.1026	3.2874	0.0613	0.2245	6.0675	0.3714	99.311
Bad a'Cheo	5.2823	0.2709	14.895	67.252	2.0059	3.437	0.0801	0.2179	6.3863	0.4126	100.24
Bad a'Cheo	5.3484	0.0901	15.369	70.791	2.5038	2.2945	0.0257	0.149	3.7997	0.2301	100.6
Bad a'Cheo	5.633	0.1889	14.646	68.065	2.116	3.1253	0.0739	0.2145	5.6392	0.3689	100.07

Bad a'Cheo	5.3246	0.0387	14.824	73.931	2.4911	1.9571	0.0079	0.1249	3.1837	0.1719	102.06
Bad a'Cheo	5.1369	0.0065	12.891	73.39	3.022	1.2778	0.0035	0.0816	2.0301	0.0887	97.928
Bad a'Cheo	4.8075	0.3077	13.599	66.243	1.9412	3.5293	0.098	0.2507	7.3828	0.4795	98.638
Bad a'Cheo	4.816	0.2293	14.061	66.72	2.2701	3.0377	0.0649	0.1946	5.8469	0.3468	97.588
Bad a'Cheo	4.7148	0.2266	13.097	66.227	2.0705	3.3529	0.0869	0.2208	6.1115	0.4038	96.512
Bad a'Cheo	5.0853	0.0232	13.641	73.691	2.8325	1.3008	0.0135	0.0856	1.8442	0.105	98.622
Bad a'Cheo	4.9774	0.0103	12.898	74.402	3.0963	1.1657	0.0063	0.0642	1.9093	0.093	98.622
Bad a'Cheo	5.3062	0.1072	14.568	67.921	2.3426	2.8758	0.0481	0.1968	5.3504	0.3152	99.031
Bad a'Cheo	4.9024	0.6722	15.111	63.233	1.8492	4.3464	0.2082	0.2711	9.1973	0.6947	100.49
Bad a'Cheo	4.4743	0.8045	15.247	62.492	1.8842	4.4467	0.2728	0.2823	9.5215	0.8012	100.23
Bad a'Cheo	5.0534	0.0283	13.592	73.733	2.9052	1.3453	0.0063	0.0692	1.9428	0.1033	98.778
Bad a'Cheo	5.1948	0.27	14.579	66.485	1.5631	3.8625	0.1129	0.241	6.6778	0.4891	99.475
Bad a'Cheo	5.5539	0.0917	13.904	67.908	2.2344	2.7509	0.0477	0.1983	4.853	0.2904	97.832
Bad a'Cheo	5.2255	0.0953	14.779	69.809	2.3285	2.7169	0.0234	0.1791	4.5242	0.2653	99.946
Bad a'Cheo	4.7918	0.0149	12.914	73.872	2.9084	1.1547	0.0128	0.0878	1.7146	0.0887	97.559
Bad a'Cheo	5.0533	0.6651	14.386	63.791	1.7823	4.3708	0.2091	0.2883	9.6048	0.6938	100.84
Bad a'Cheo	4.7566	0.863	15.366	62.503	1.6364	4.5499	0.2492	0.2513	9.534	0.8119	100.52
Bad a'Cheo	5.3673	0.0519	15.009	72.973	2.6899	2.175	0.0299	0.0886	2.5591	0.1992	101.14
Bad a'Cheo	4.6579	0.0103	12.767	72.269	2.7859	1.3084	0.0014	0.0728	1.8912	0.0899	95.853
Bad a'Cheo	5.1581	0.4995	14.737	64.576	1.7816	4.2292	0.1788	0.2661	8.7024	0.6528	100.78
Bad a'Cheo	4.3528	0.0136	12.856	72.929	3.2345	1.1736	0.0045	0.0896	1.9575	0.0948	96.705
Bad a'Cheo	5.2784	0.0228	14.312	71.824	2.4416	2.4197	0.0302	0.1488	3.8346	0.2005	100.51
Bad a'Cheo	5.0298	0.0104	12.506	71.968	2.8396	1.2458	0.0048	0.0787	1.798	0.0923	95.574
Bad a'Cheo	4.7509	0.4462	14.254	60.807	1.5999	3.9758	0.2271	0.2275	7.7565	0.5424	94.587
Bad a'Cheo	4.7771	0.0123	12.645	74.193	2.9978	1.297	0.0103	0.0744	1.9313	0.0926	98.032
Bad a'Cheo	5.1512	0.0769	13.303	69.03	2.2901	2.6178	0.0287	0.1753	4.0649	0.2476	96.985
Bad a'Cheo	5.1258	0.5666	14.812	64.541	1.8446	4.3186	0.171	0.282	8.7004	0.6701	101.03
Bad a'Cheo	5.1326	0.6669	14.692	62.578	1.8232	4.4419	0.1915	0.2988	8.8333	0.7052	99.363
Bad a'Cheo	5.4133	0.0291	13.967	70.794	2.482	2.2998	0.0287	0.1527	3.6089	0.2073	98.982
Bad a'Cheo	4.9171	0.6327	14.658	63.639	1.7802	4.1929	0.1631	0.2971	8.8693	0.6861	99.836
Bad a'Cheo	5.2081	0.0162	13.559	73.724	2.8825	1.2881	0.0045	0.0999	1.9048	0.0988	98.786
Bad a'Cheo	4.9006	0.0049	12.628	71.062	2.83	1.235	0.0049	0.0856	1.9756	0.0879	94.814
Bad a'Cheo	5.3408	0.1005	14.246	68.405	2.2825	2.8193	0.0515	0.2005	5.6474	0.3374	99.431
Bad a'Cheo	5.5373	0.0199	13.978	71.964	2.7425	2.0194	0.0065	0.1265	3.367	0.1736	99.935
Bad a'Cheo	4.5721	0.0252	12.735	71.53	3.032	1.2733	0.0007	0.0825	1.8844	0.085	95.22
Bad a'Cheo	5.1169	0.5945	15.365	63.62	1.543	4.5741	0.1769	0.2751	9.1764	0.6944	101.14
Bad a'Cheo	5.1809	0.5611	14.403	63.708	1.7581	4.0565	0.1606	0.2742	7.9311	0.6691	98.702
Bad a'Cheo	4.9249	0.041	12.802	74.427	2.7238	1.3203	0.0141	0.0694	1.7831	0.0762	98.182
Bad a'Cheo	4.9498	0.5769	14.557	61.899	1.7361	4.5102	0.1711	0.3106	9.2224	0.669	98.602
Bad a'Cheo	4.6588	0.0489	12.985	73.798	2.9545	0.7996	0.0063	0.0176	1.2044	0.0765	96.549
Bad a'Cheo	5.1758	0.0046	12.332	75.42	2.5589	1.3353	0.0003	0.0525	1.4048	0.0836	98.368
Bad a'Cheo	4.5924	0.0818	13.444	69.664	2.293	2.8636	0.0266	0.1882	4.8547	0.2826	98.291
Bad a'Cheo	4.7447	0.0157	12.382	74.116	2.8144	1.3248	0.0104	0.0887	1.9939	0.099	97.59
Bad a'Cheo	4.4596	0.5466	14.243	63.367	1.7723	4.2332	0.1535	0.2973	9.0706	0.6534	98.796
Bad a'Cheo	4.823	0.2076	13.367	66.829	2.029	3.4042	0.0709	0.2365	6.4409	0.4224	97.831
Bad a'Cheo	4.654	0.6661	13.795	62.948	1.7301	4.386	0.1875	0.277	8.9752	0.7133	98.333
Bad a'Cheo	3.9727	0.5885	13.561	63.04	1.5989	4.2034	0.2038	0.2572	8.9012	0.6749	97.002
Bad a'Cheo	4.6797	0.1475	13.571	68.321	2.3546	2.8056	0.0334	0.1928	5.2055	0.2905	97.602
Bad a'Cheo	4.1836	0.8726	13.227	60.792	1.9002	4.5155	0.2498	0.287	9.3584	0.8008	96.187
Bad a'Cheo	4.3495	0.7037	13.752	63.331	1.8014	4.3535	0.1895	0.2727	9.6025	0.7179	99.073
Bad a'Cheo	4.6595	0.636	13.683	63.421	1.8361	4.348	0.1909	0.2858	9.0376	0.6705	98.768
Bad a'Cheo	4.7818	0.0148	11.813	73.295	2.0906	1.0215	0.0195	0.0059	0.1335	0.0844	93.26
Bad a'Cheo	4.5935	0.0308	12.148	74.208	2.3922	0.862	0.0141	0.0002	0.0244	0.0811	94.354
Bad a'Cheo	4.2166	0.0227	12.431	75.99	2.8058	1.3645	0.0114	0.0647	1.6772	0.0879	98.672
Bad a'Cheo	4.7435	0.0197	12.956	76.019	2.9843	1.3316	0.0066	0.0807	1.8561	0.0852	100.08
Bad a'Cheo	4.7555	0.4888	14.574	64.926	1.7389	4.2011	0.1479	0.2774	8.462	0.6618	100.23
Bad a'Cheo	4.6142	0.1412	14.337	68.878	2.1607	2.9548	0.0382	0.2174	5.2555	0.3144	98.911

Bad a'Cheo	4.8714	0.0213	13.334	71.271	2.5837	1.7492	0.008	0.1342	3.087	0.1577	97.217
Bad a'Cheo	4.5732	0.0126	12.756	74.78	3.3501	1.214	0.0069	0.0907	1.8174	0.0963	98.697
Bad a'Cheo	4.8218	0.0609	14.214	70.102	2.2642	2.629	0.0267	0.1814	4.532	0.2741	99.105
Bad a'Cheo	4.6067	0.0292	12.566	75.17	2.8508	1.3854	0.0037	0.0822	1.8063	0.0732	98.573
Bad a'Cheo	5.0679	0.0145	13.781	76.46	2.8076	1.3446	0.0016	0.0824	1.9228	0.0944	101.58
Bad a'Cheo	4.9084	0.1658	12.092	71.069	3.8743	0.5417	0.0189	0.1323	2.2317	0.3234	95.357
Bad a'Cheo	4.6415	0.0248	12.533	73.388	2.9783	1.2441	0.0026	0.0958	1.7188	0.083	96.71
Bad a'Cheo	4.7782	0.0898	13.216	77.878	2.3644	1.7826	0.0112	0.065	2.1001	0.1095	102.39
Bad a'Cheo	4.4429	0.7048	14.652	63.274	1.7358	4.443	0.2027	0.3092	8.9854	0.7404	99.491
Bad a'Cheo	4.7272	0.0028	12.824	74.5	2.8774	1.2879	0.0084	0.0767	2.0395	0.0934	98.437
Bad a'Cheo	5.0654	0.0366	13.726	69.802	2.3336	2.2886	0.0197	0.1351	3.583	0.2135	97.203
Bad a'Cheo	4.4587	0.0086	12.717	72.603	2.8973	1.2735	0.0026	0.0755	1.9196	0.0923	96.047
Bad a'Cheo	4.617	0.0335	13.229	75.191	2.8938	1.4054	0.0085	0.0785	2.0076	0.0906	99.555
Bad a'Cheo	4.7948	0.5923	14.317	64.449	1.859	4.458	0.1471	0.2902	9.0838	0.666	100.66
Bad a'Cheo	4.5983	0.0054	12.718	73.612	2.9944	1.4047	0.0011	0.0776	1.9231	0.087	97.419
Bad a'Cheo	4.717	0.0077	13.078	73.64	2.9633	1.2226	0.0003	0.0729	1.621	0.1005	97.407
Bad a'Cheo	4.654	0.6096	14.706	64.24	1.8977	4.2345	0.1819	0.2705	8.6902	0.7122	100.2
Bad a'Cheo	4.7063	0.6144	14.467	63.221	1.8449	4.4043	0.1589	0.2805	8.7247	0.6815	99.103
Bad a'Cheo	4.3253	1.1909	13.365	66.955	1.6156	4.1159	0.2875	0.1343	6.6083	1.1348	99.732
Bad a'Cheo	4.5697	0.0189	12.777	74.843	3.105	1.1788	0.0095	0.0789	2.1254	0.0939	98.8
Bad a'Cheo	4.5929	0.641	14.501	64.564	1.7767	4.1986	0.1761	0.2971	8.8113	0.6903	100.25
Bad a'Cheo	4.1449	0.5805	14.699	63.079	1.913	4.2379	0.1968	0.29	8.8495	0.6726	98.663
Bad a'Cheo	5.1978	0.0342	13.329	71.604	2.6263	2.2479	0.0065	0.1479	3.4908	0.1943	98.879
Bad a'Cheo	4.6298	0.0384	12.052	73.641	2.9665	1.2607	0.0127	0.0694	1.7095	0.0891	96.469
Bad a'Cheo	5.1508	0.1581	13.686	67.843	2.1966	3.0572	0.045	0.1902	5.6022	0.333	98.262
Bad a'Cheo	4.4552	0.0062	12.065	72.972	2.8424	1.3605	0.0107	0.0864	1.9945	0.0904	95.871
Bad a'Cheo	4.8213	0.0326	12.227	73.483	3.0144	1.3223	0.0072	0.0788	1.9444	0.0972	97.028
Bad a'Cheo	4.8923	0.0291	12.886	74.839	3.07	1.1485	0.0091	0.0973	2.0145	0.0965	99.083
Bad a'Cheo	4.675	0.191	11.598	71.98	3.8802	0.5148	0.0371	0.1201	2.2352	0.481	95.712
Bad a'Cheo	4.7078	0.0038	12.725	69.28	2.7494	1.7871	0.0216	0.1155	3.0751	0.1617	94.619
Bad a'Cheo	6.286	0.0192	15.214	72.56	1.4762	2.1172	0.0004	0.0193	1.0982	0.0657	98.855
Bad a'Cheo	4.8372	0.1683	13.411	65.928	2.2106	3.088	0.0961	0.1746	5.6297	0.3056	95.849
Bad a'Cheo	4.6085	0.0131	11.978	73.108	2.786	1.2995	0.001	0.0894	1.7894	0.0863	95.759
Bad a'Cheo	4.5424	0.6164	14.059	62.784	1.7492	4.39	0.2022	0.288	8.7666	0.6882	98.086
Bad a'Cheo	4.5247	0.0275	12.242	74.685	2.9592	1.2462	0.0059	0.0806	1.8796	0.0945	97.745
Bad a'Cheo	4.6092	0.0356	12.111	72.064	2.6938	1.3271	0.0024	0.0809	2.131	0.0963	95.147
Bad a'Cheo	4.582	0.0427	12.317	74.765	2.3908	1.5857	0.0104	0.0873	1.5601	0.0598	97.401
Bad a'Cheo	4.6197	0.021	13.205	72.71	2.8745	1.783	0.0127	0.1276	2.6333	0.1107	98.097
Bad a'Cheo	4.9675	0.0192	11.634	73.672	2.7636	1.2626	0.0056	0.065	2.0035	0.1381	96.53
Bad a'Cheo	5.2505	0.0229	13.35	71.688	2.572	1.6685	0.0105	0.1388	2.5043	0.1123	97.318
Bad a'Cheo	4.6241	0.6685	13.622	63.325	1.7344	4.2326	0.1972	0.6873	9.0677	0.2847	98.442
Bad a'Cheo	5.1665	0.023	12.164	74.472	2.8499	1.2723	0.0118	0.0948	1.938	0.0772	98.07
Bad a'Cheo	4.4062	0.0288	11.927	74.774	2.8116	1.3944	0.0128	0.081	1.5388	0.0618	97.037
Bad a'Cheo	4.8004	0.0031	12.6	73.901	2.8584	1.2856	0.0076	0.0958	2.0794	0.0892	97.699
Bad a'Cheo	4.7343	0.6627	13.798	61.926	1.8105	4.4682	0.2246	0.698	9.472	0.2902	98.084
Bad a'Cheo	4.853	0.0054	12.131	75.057	2.8723	1.2988	0.0138	0.0949	1.8595	0.0861	98.272
Bad a'Cheo	4.206	0.1423	11.521	72.102	5.1128	0.4821	0.0425	0.5074	2.208	0.1261	96.45
Broubster	0.5094	0.0021	19.499	65.135	16.16	0.0205	0.0027	0.0197	0.0632	0.0624	101.31
Broubster	4.9074	0.0202	13.592	72.847	2.8743	1.2825	0.0128	0.0811	1.9041	0.0767	97.598
Broubster	5.0422	0.0184	13.118	74.571	2.9022	1.4346	0.0131	0.0956	1.9305	0.0914	99.217
Broubster	4.9115	0.0203	12.958	72.966	2.8185	1.2835	0.0045	0.0917	1.8872	0.0794	97.02
Broubster	5.0249	0.0244	12.669	73.607	2.7865	1.3811	0.0062	0.0932	2.0243	0.0915	97.708
Broubster	4.7161	0.016	12.119	72.248	2.5393	1.1994	0.0063	0.0783	1.747	0.0531	94.722
Broubster	5.0788	0.5677	14.702	64.348	1.8305	4.3057	0.1963	0.653	8.7278	0.2739	100.68
Broubster	5.0832	0.022	12.239	71.412	2.7115	1.29	0.0118	0.0978	1.9917	0.0915	94.95
Broubster	4.7453	0.0235	13.177	72.41	2.9147	1.3176	0.0035	0.0834	2.0363	0.0744	96.778
Broubster	4.4653	1.3287	14.492	62.518	1.6971	4.9986	0.192	0.95	9.0346	0.2609	99.937

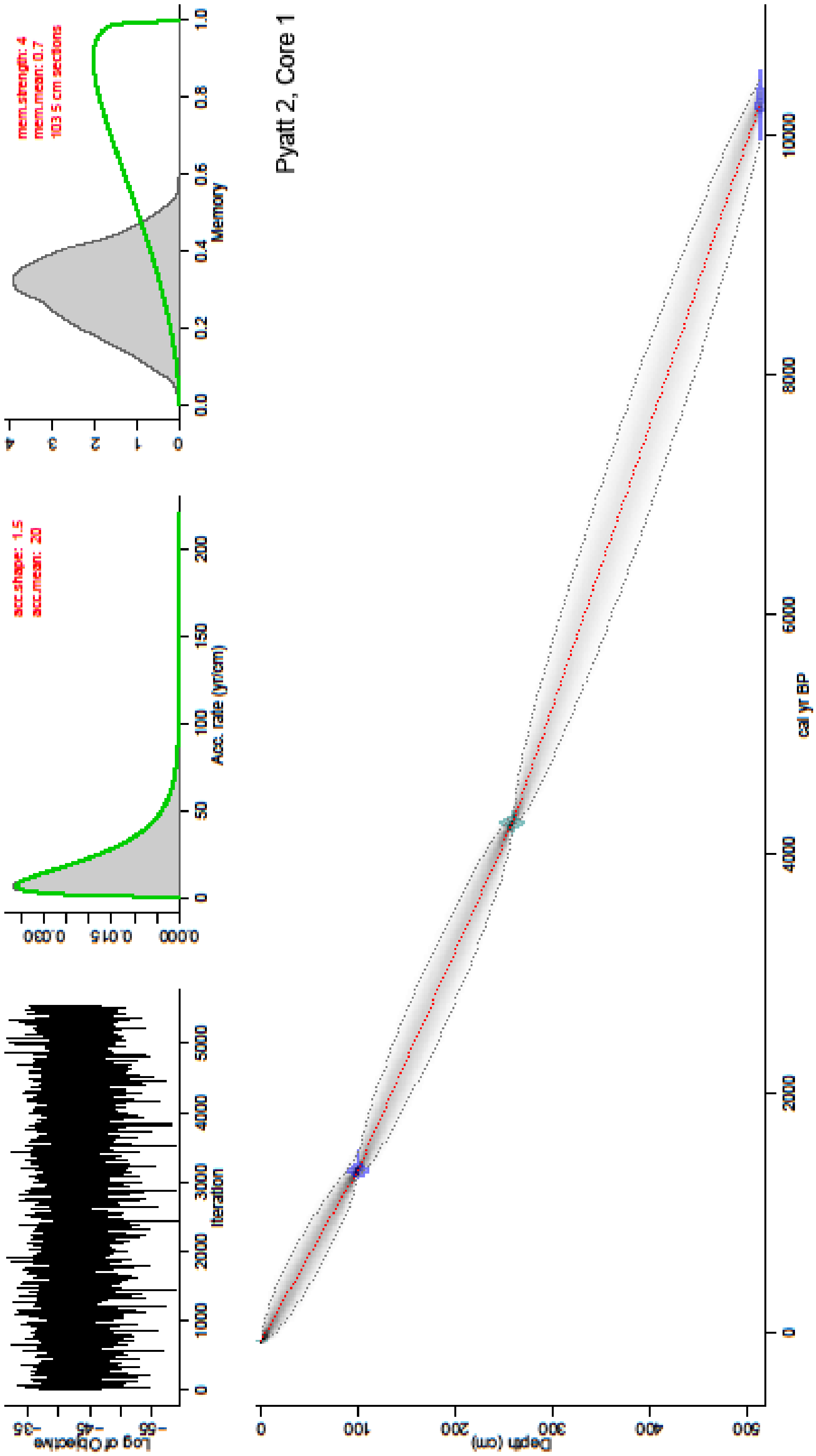
Broubster	4.9113	0.6822	14.464	63.149	1.8192	4.4768	0.2217	0.726	9.0809	0.288	99.819
Broubster	4.9901	0.0171	13.565	75.559	2.8272	1.3407	0.0072	0.1017	2.0188	0.0809	100.51
Broubster	5.167	0.4261	14.317	63.511	1.8938	3.7945	0.1771	0.5891	7.9036	0.284	98.063
Broubster	4.9221	0.0253	13.257	75.293	2.9699	1.3608	0.0124	0.1031	2.108	0.0726	100.12
Broubster	5.0501	1.6415	15.871	60.374	1.6192	5.0664	0.3999	0.2503	9.359	1.1772	100.81
Broubster	4.6766	0.3805	15.032	68.539	2.4548	2.5576	0.0991	0.1994	6.3453	0.4481	100.73
Broubster	5.5824	0.1961	14.673	68.066	2.031	3.3196	0.0542	0.2156	5.7168	0.3855	100.24
Broubster	5.0994	0.0029	14.138	75.598	3.1851	1.097	0.0017	0.0745	1.3082	0.0767	100.58
Broubster	5.418	0.0932	14.568	68.965	2.2953	2.8047	0.0433	0.2113	5.4104	0.3276	100.14
Broubster	4.9347	0.651	14.802	63.009	1.7476	4.2932	0.1823	0.2988	9.2391	0.6911	99.848
Broubster	5.6957	0.0422	14.798	68.876	2.2742	2.614	0.0262	0.1797	4.6782	0.258	99.442
Broubster	5.3487	0.459	14.668	63.639	1.8156	4.1566	0.1308	0.2733	8.1525	0.5829	99.227
Broubster	4.525	0.6477	14.359	63.446	1.7128	4.4889	0.181	0.2918	9.2687	0.701	99.622
Broubster	4.5882	1.469	16.001	61.667	1.8149	4.8539	0.3519	0.2254	8.3382	1.0938	100.4
Dalchork	5.6153	0.0376	11.858	70.875	4.402	0.3998	0.0052	0.0754	2.9001	0.1923	96.361
Dalchork	13.114	0.0048	19.702	69.179	0.0772	0.0273	0.0028	0.0037	0.0127	0.0018	102.12
Dalchork	5.6893	0.046	12.287	71.048	4.4243	0.3719	0.0131	0.0669	2.8932	0.2042	97.044
Dalchork	5.5538	0.0404	12.429	70.121	4.5511	0.3859	0.0124	0.0689	2.9076	0.1942	96.264
Forsinard	5.2852	0.0201	12.644	73.755	2.7948	1.4006	0.0104	0.0968	1.8729	0.0855	97.966
Forsinard	4.7446	0.2679	13.294	66.681	2.0411	3.5454	0.0863	0.46	6.7915	0.2627	98.175
Forsinard	4.7505	0.0834	13.496	69.231	2.3063	2.7141	0.0516	0.2706	4.8726	0.1947	97.97
Forsinard	4.7395	0.0231	13.157	72.827	2.819	1.4019	0.0087	0.0862	1.8302	0.0918	96.984
Forsinard	5.2233	0.0343	14.968	72.852	2.5639	1.9429	0.0055	0.1682	3.1157	0.1173	100.99
Forsinard	4.4671	0.1231	12.437	71.39	4.3567	0.4835	0.0187	0.3059	2.0658	0.1038	95.752
Forsinard	5.1074	0.0087	12.849	73.734	2.8644	1.4101	0.0072	0.0958	2.0301	0.085	98.192
Forsinard	5.6129	0.558	16.149	63.381	1.3009	4.7868	0.2104	0.6412	8.0408	0.2606	100.94
Forsinard	4.897	0.4617	14.873	64.399	1.8089	4.3542	0.1428	0.5727	8.4908	0.2759	100.28
Forsinard	4.8778	0.63	15.528	64.339	1.7808	4.212	0.2175	0.7166	8.5069	0.2887	101.1
Rosal	0.3376	0.015	13.936	78.108	2.2306	1.3414	0.0032	0.0903	2.1483	0.1012	98.312
Rosal	4.747	0.6091	14.393	62.775	1.8595	4.3276	0.1745	0.2751	8.8722	0.7769	98.81
Rosal	4.3948	0.6276	14.428	64.811	1.7099	4.3218	0.1687	0.2675	9.0451	0.6956	100.47
Rosal	4.9072	0.0039	13.981	71.588	2.4764	2.3038	0.0232	0.1493	3.8264	0.2157	99.475
Rosal	4.76	0.0095	12.703	75.244	2.9771	1.3469	0	0.0802	2.2345	0.0942	99.449
Rosal	5.0898	0.0531	13.694	70.16	2.3236	2.4088	0.0246	0.1561	4.1765	0.2342	98.32
Rosal	4.7127	0.7044	14.234	63.337	1.8327	4.5136	0.1756	0.2929	9.0648	0.708	99.576
Rosal	5.0463	0.0193	13.991	71.063	2.2899	2.4481	0.0299	0.1598	4.1176	0.2274	99.392
Rosal	4.5893	0.0117	12.35	72.339	2.8579	1.2881	0.0137	0.0662	1.9571	0.0926	95.566
Rosal	4.3986	0.743	14.577	63.092	1.7634	4.5117	0.1961	0.3045	9.1202	0.749	99.456
Rosal	4.5321	0.0154	12.622	73.324	2.8744	1.3485	0.0032	0.0935	1.9942	0.0964	96.904
Rosal	4.4313	0.0119	12.289	71.966	2.6851	1.2651	0.0361	0.0605	1.8417	0.0898	94.676
Rosal	4.9983	0.0603	13.897	69.45	2.3059	2.7355	0.0314	0.1799	4.5817	0.273	98.514
Rosal	5.5215	0.0239	14.034	74.724	2.6467	1.5608	0.0008	0.086	1.6991	0.092	100.39
Rosal	5.0058	0.066	13.877	70.599	2.5419	2.2969	0.0224	0.1294	3.7952	0.2107	98.544
Rosal	4.9524	0.0127	13.59	72.546	2.635	1.6862	0.0111	0.1047	2.8283	0.1453	98.512
Rosal	4.1862	0.0139	12.134	70.878	2.8793	1.1389	0.0227	0.0606	1.7469	0.0865	93.146
Rosal	4.7644	0.0172	12.817	74.305	2.9349	1.371	0.0027	0.0817	2.0197	0.0958	98.41
Rosal	4.6589	0.0534	12.764	73.859	2.8668	1.3074	0.003	0.0963	2.1115	0.0964	97.816
Rosal	4.6506	0.0089	12.479	73.812	2.8961	1.4124	0.0018	0.0817	2.0597	0.0984	97.5
Rosal	4.9181	0.0221	12.359	72.362	2.8193	1.3162	0.0099	0.099	1.9372	0.0924	95.935
Rosal	4.5644	0.0193	12.322	73.7	2.7913	1.2896	0.0051	0.0785	1.878	0.0948	96.743
Rosal	4.8054	0.0098	13.369	74.615	2.8529	1.3539	0.0064	0.072	1.973	0.1039	99.162
Rosal	4.5354	0.6916	12.649	67.357	1.8474	3.0904	0.1517	0.1168	5.1923	0.7381	96.37
Rosal	4.6893	0.7338	13.155	70.527	1.8011	3.0964	0.1366	0.1348	5.1292	0.741	100.14
Rosal	4.9402	0.4061	12.514	72.452	2.0571	2.4375	0.0637	0.1029	3.793	0.5363	99.303
Rosal	4.7084	0.3644	12.453	73.617	2.0388	2.2963	0.0691	0.1048	3.9344	0.52	100.11
Rosal	4.4394	0.3976	12.819	74.565	2.0623	2.3844	0.0748	0.102	4.0795	0.5126	101.44
Rosal	6.1828	0.0154	27.635	57.184	0.1658	8.9467	0.0034	0.0014	0.4501	0.0347	100.62

Rosal	4.5593	0.4311	12.952	72.617	2.0463	2.3673	0.0745	0.0702	3.7496	0.5428	99.411
Rosal	4.2344	0.3775	12.433	71.678	2.1437	2.2078	0.0596	0.0982	3.6826	0.5217	97.437
Rosal	5.0608	0.4276	12.488	72.372	2.1589	2.2314	0.0603	0.1085	3.8642	0.4991	99.27
Rosal	4.5914	0.4338	12.413	72.5	2.0235	2.3835	0.0628	0.103	3.8714	0.513	98.895
Rosal	5.0345	0.8632	12.58	67.665	1.779	3.3949	0.144	0.1315	5.1894	0.7741	97.556
Rosal	4.2325	0.6469	13.412	71.977	1.9774	2.8496	0.1205	0.1228	4.5925	0.6368	100.57
Rosal	4.9249	0.3905	12.927	72.941	2.076	2.4523	0.0704	0.1185	4.0355	0.5158	100.45
Rosal	4.4403	0.4292	12.542	73.701	2.091	2.4849	0.0816	0.0875	3.9822	0.5003	100.34
Rosal	4.8906	1.277	13.011	69.194	1.828	3.6405	0.08	0.1262	5.1794	0.6713	99.897
Rosal	4.7843	0.4692	12.699	71.849	2.002	2.5365	0.0914	0.1037	4.4999	0.5949	99.629
Rosal	4.6717	0.4103	12.853	72.145	2.0969	2.342	0.0712	0.0852	3.8675	0.5083	99.052
Rosal	4.3486	0.4981	13.23	72.42	2.1219	2.6999	0.0938	0.1151	4.5401	0.5919	100.66
Rosal	4.3375	0.4064	12.732	73.772	1.9774	2.4556	0.0614	0.101	3.7761	0.5141	100.13
Rosal	4.3919	0.7033	13.36	70.848	1.7653	3.288	0.1361	0.1311	4.7359	0.7252	100.08
Rosal	4.9572	0.6254	13.197	73.985	2.003	2.9568	0.0843	0.0968	4.0804	0.5446	102.53
Rosal	4.5652	0.5739	12.692	71.911	1.9177	2.7999	0.1151	0.1086	4.4149	0.6718	99.77
Rosal	4.6721	0.5716	13.251	72.409	2.0259	2.7796	0.1081	0.1179	4.613	0.6517	101.2
Rosal	5.2531	0.0341	13.084	74.739	2.7945	1.3871	0.01	0.0867	2.0719	0.0975	99.557
Rosal	6.0848	0.0934	15.241	68.38	1.8382	2.3243	0.044	0.2266	6.6202	0.2516	101.1
Rosal	5.3247	0.0022	13.677	75.818	2.2779	1.395	0	0.0612	1.4079	0.0896	100.05
Rosal	5.2082	0.0186	13.831	74.997	2.8597	1.3182	0.0021	0.0654	2.0603	0.0927	100.45
Rosal	5.1493	0.0781	15.034	68.722	2.1915	3.082	0.0344	0.2156	5.3286	0.2979	100.13
Rosal	4.9232	0.0467	13.506	69.171	2.3302	2.3996	0.0315	0.1615	4.4984	0.227	97.294
Rosal	5.0524	0.0291	12.921	74.083	2.8532	1.4876	0.0087	0.092	1.8992	0.0898	98.515
Rosal	4.9112	0.0187	13.655	71.149	2.8252	1.2754	0.0388	0.0754	1.8105	0.0883	95.847
Rosal	5.4896	0.139	14.293	67.501	2.1658	2.9747	0.039	0.2169	5.7288	0.3165	98.864
Rosal	5.2788	0.5068	13.934	64.468	1.8172	4.2243	0.1538	0.2749	8.5343	0.623	99.815

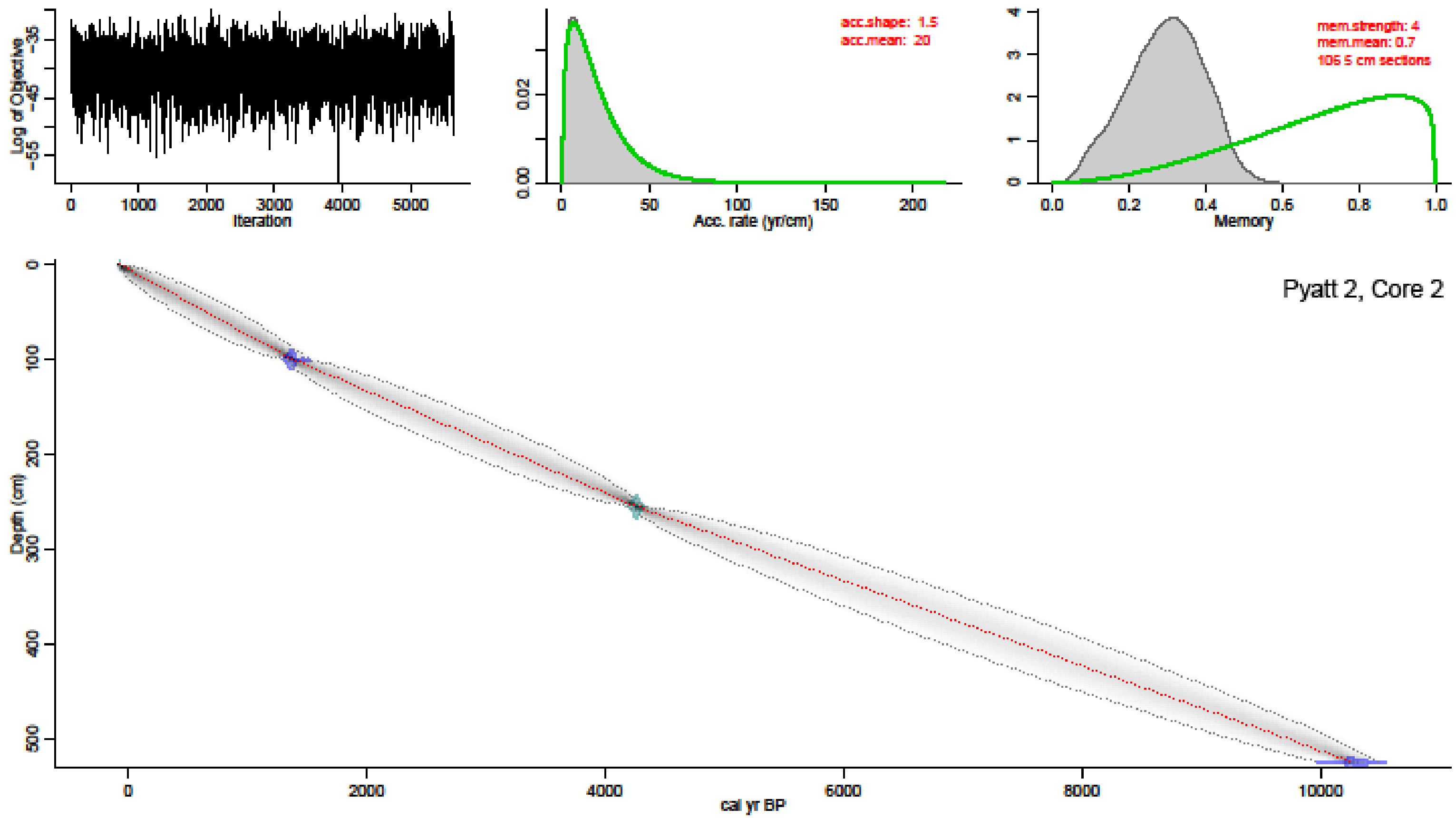
Supplementary data table 2. EMP data for Glen Garry tephra deposits with site designations.

Site	Na_2O	MgO	Al_2O_3	SiO_2	K_2O	CaO	P_2O_5	MnO	FeO	TiO_2	Total
Rosal	4.5354	0.6916	12.649	67.357	1.8474	3.0904	0.1517	0.1168	5.1923	0.7381	96.37
Rosal	4.6893	0.7338	13.155	70.527	1.8011	3.0964	0.1366	0.1348	5.1292	0.741	100.14
Rosal	4.9402	0.4061	12.514	72.452	2.0571	2.4375	0.0637	0.1029	3.793	0.5363	99.303
Rosal	4.7084	0.3644	12.453	73.617	2.0388	2.2963	0.0691	0.1048	3.9344	0.52	100.11
Rosal	4.4394	0.3976	12.819	74.565	2.0623	2.3844	0.0748	0.102	4.0795	0.5126	101.44
Rosal	6.1828	0.0154	27.635	57.184	0.1658	8.9467	0.0034	0.0014	0.4501	0.0347	100.62
Rosal	4.5593	0.4311	12.952	72.617	2.0463	2.3673	0.0745	0.0702	3.7496	0.5428	99.411
Rosal	4.2344	0.3775	12.433	71.678	2.1437	2.2078	0.0596	0.0982	3.6826	0.5217	97.437
Rosal	5.0608	0.4276	12.488	72.372	2.1589	2.2314	0.0603	0.1085	3.8642	0.4991	99.27
Rosal	4.5914	0.4338	12.413	72.5	2.0235	2.3835	0.0628	0.103	3.8714	0.513	98.895
Rosal	5.0345	0.8632	12.58	67.665	1.779	3.3949	0.144	0.1315	5.1894	0.7741	97.556
Rosal	4.2325	0.6469	13.412	71.977	1.9774	2.8496	0.1205	0.1228	4.5925	0.6368	100.57
Rosal	4.9249	0.3905	12.927	72.941	2.076	2.4523	0.0704	0.1185	4.0355	0.5158	100.45
Rosal	4.4403	0.4292	12.542	73.701	2.091	2.4849	0.0816	0.0875	3.9822	0.5003	100.34
Rosal	4.8906	1.277	13.011	69.194	1.828	3.6405	0.08	0.1262	5.1794	0.6713	99.897
Rosal	4.7843	0.4692	12.699	71.849	2.002	2.5365	0.0914	0.1037	4.4999	0.5949	99.629
Rosal	4.6717	0.4103	12.853	72.145	2.0969	2.342	0.0712	0.0852	3.8675	0.5083	99.052
Rosal	4.3486	0.4981	13.23	72.42	2.1219	2.6999	0.0938	0.1151	4.5401	0.5919	100.66
Rosal	4.3375	0.4064	12.732	73.772	1.9774	2.4556	0.0614	0.101	3.7761	0.5141	100.13
Rosal	4.3919	0.7033	13.36	70.848	1.7653	3.288	0.1361	0.1311	4.7359	0.7252	100.08
Rosal	4.9572	0.6254	13.197	73.985	2.003	2.9568	0.0843	0.0968	4.0804	0.5446	102.53
Rosal	4.5652	0.5739	12.692	71.911	1.9177	2.7999	0.1151	0.1086	4.4149	0.6718	99.77
Rosal	4.6721	0.5716	13.251	72.409	2.0259	2.7796	0.1081	0.1179	4.613	0.6517	101.2

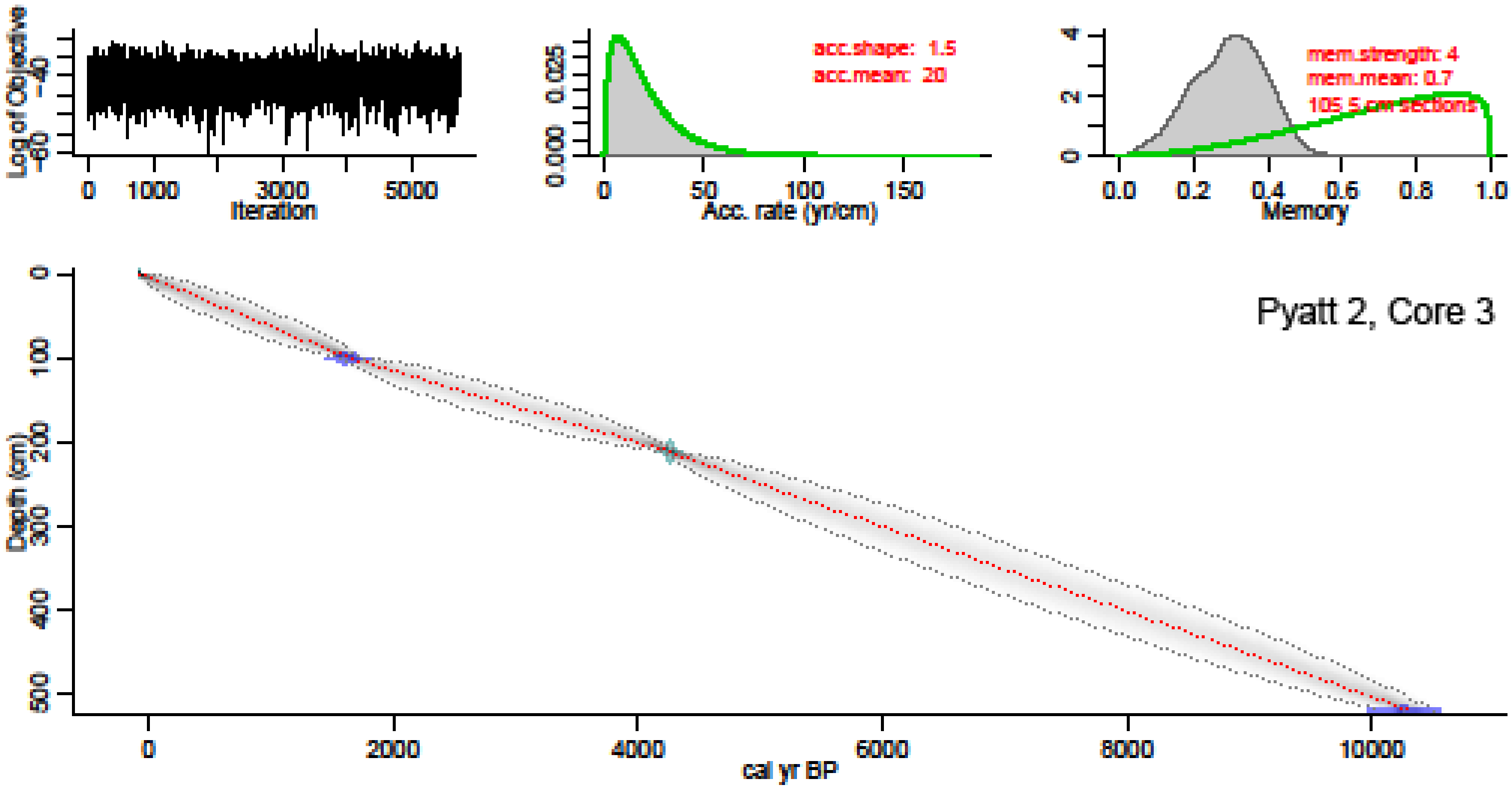
Age depth models



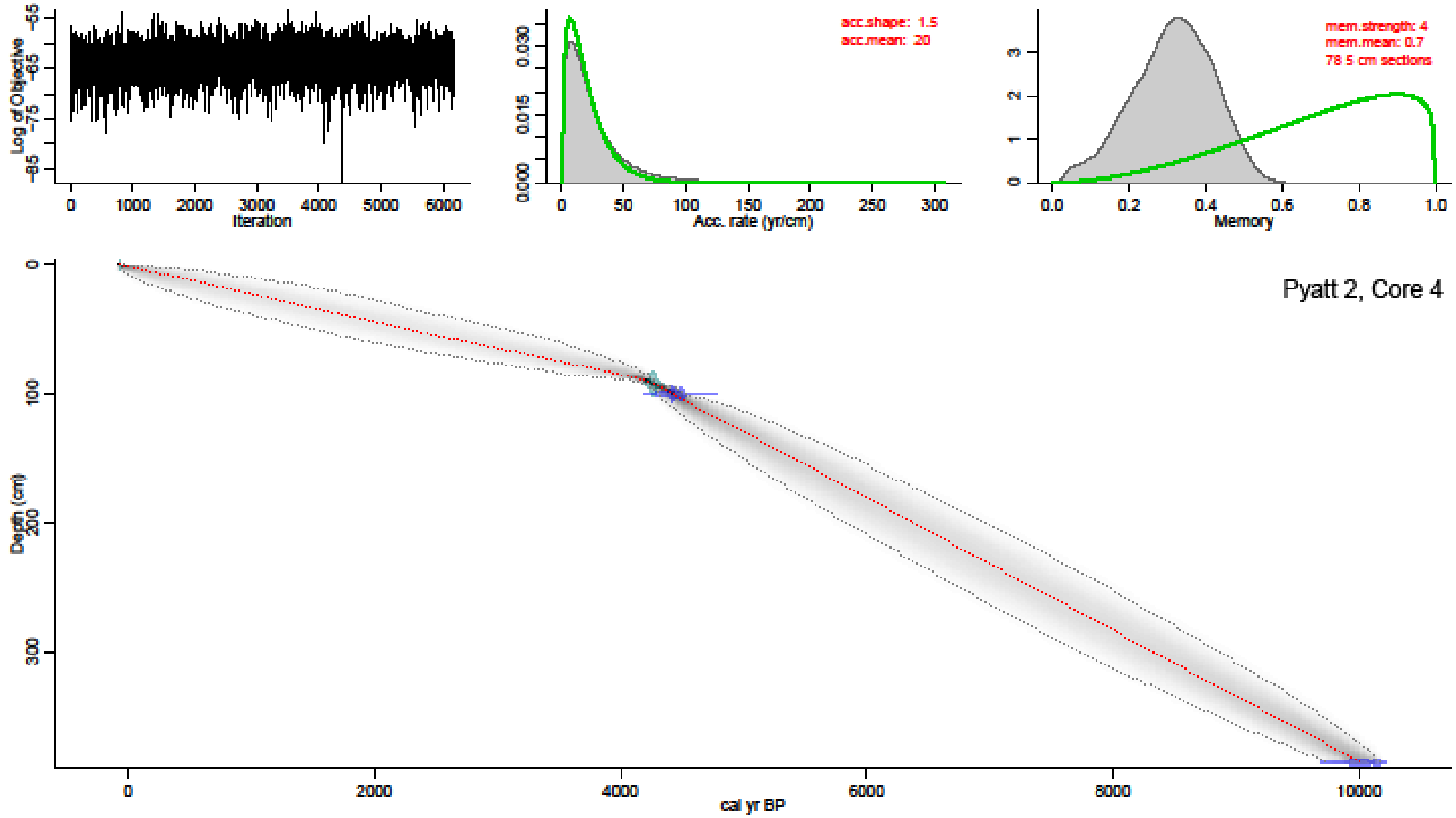
Supplementary Figure 1. Age depth model derived from radiocarbon dating and tephra deposits for Pyatt 2, Core 1.



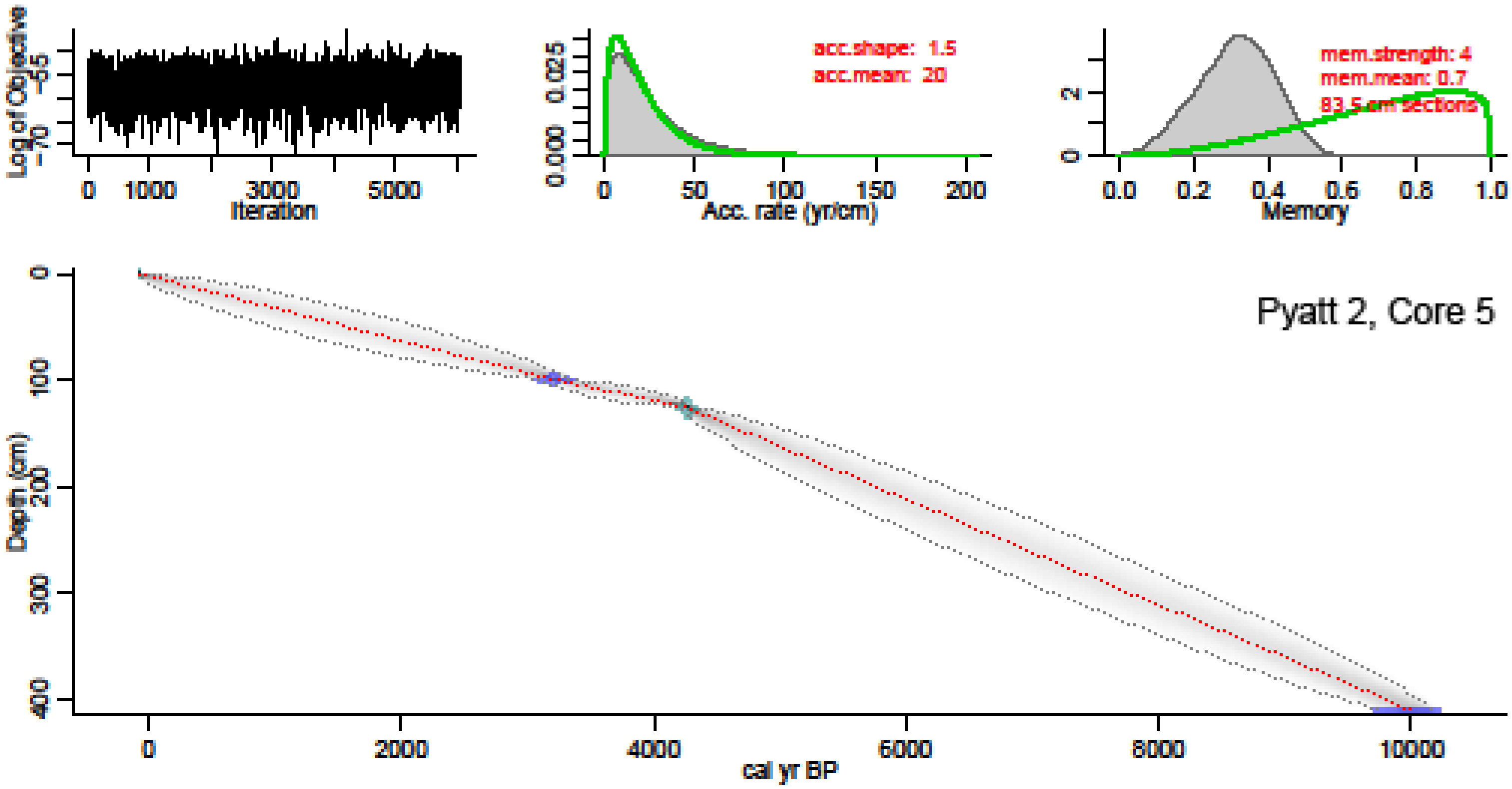
Supplementary Figure 2. Age depth model derived from radiocarbon dating and tephra deposits for Pyatt 2, Core 2.



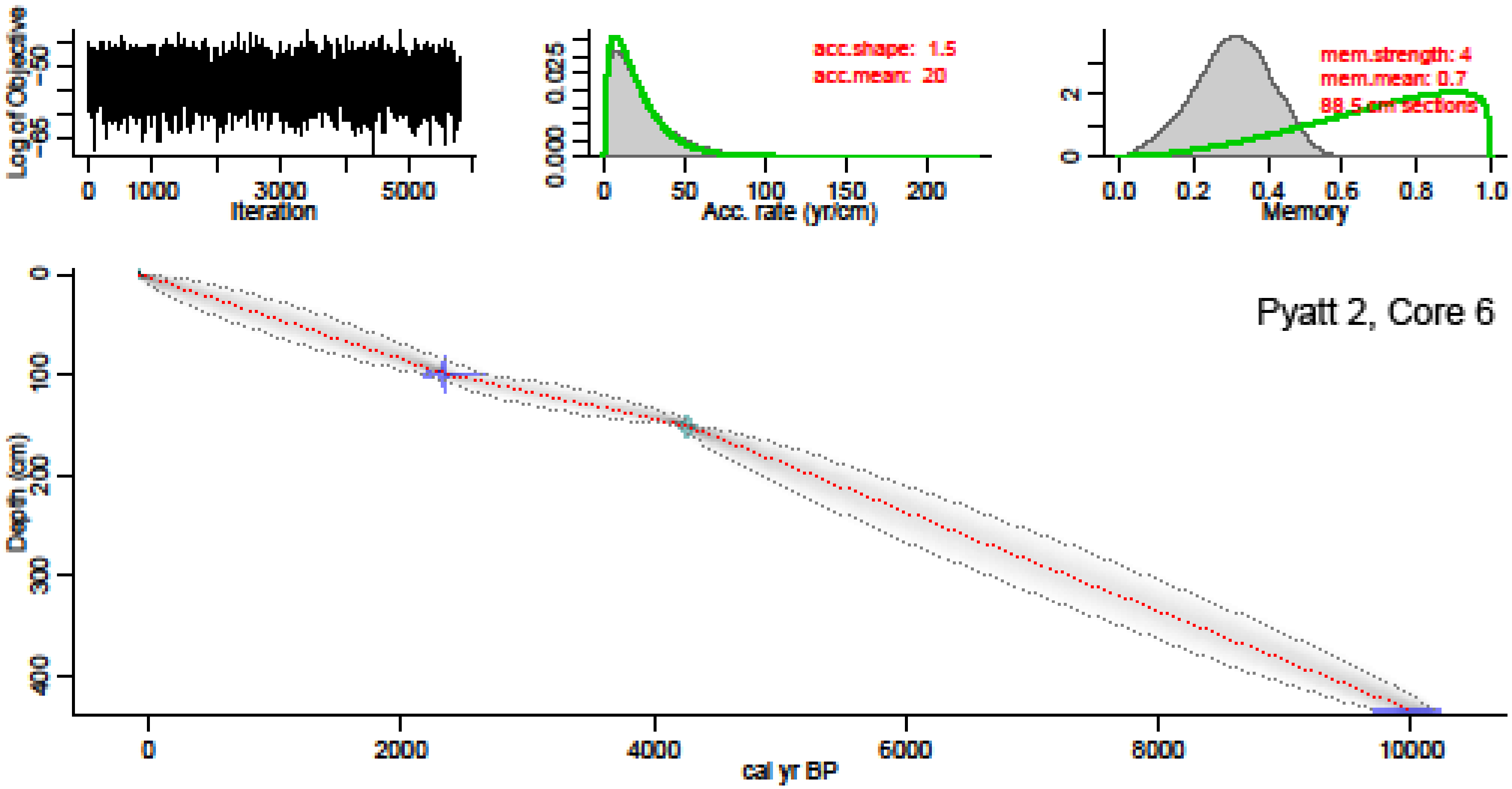
Supplementary Figure 3. Age depth model derived from radiocarbon dating and tephra deposits for Pyatt 2, Core 3.



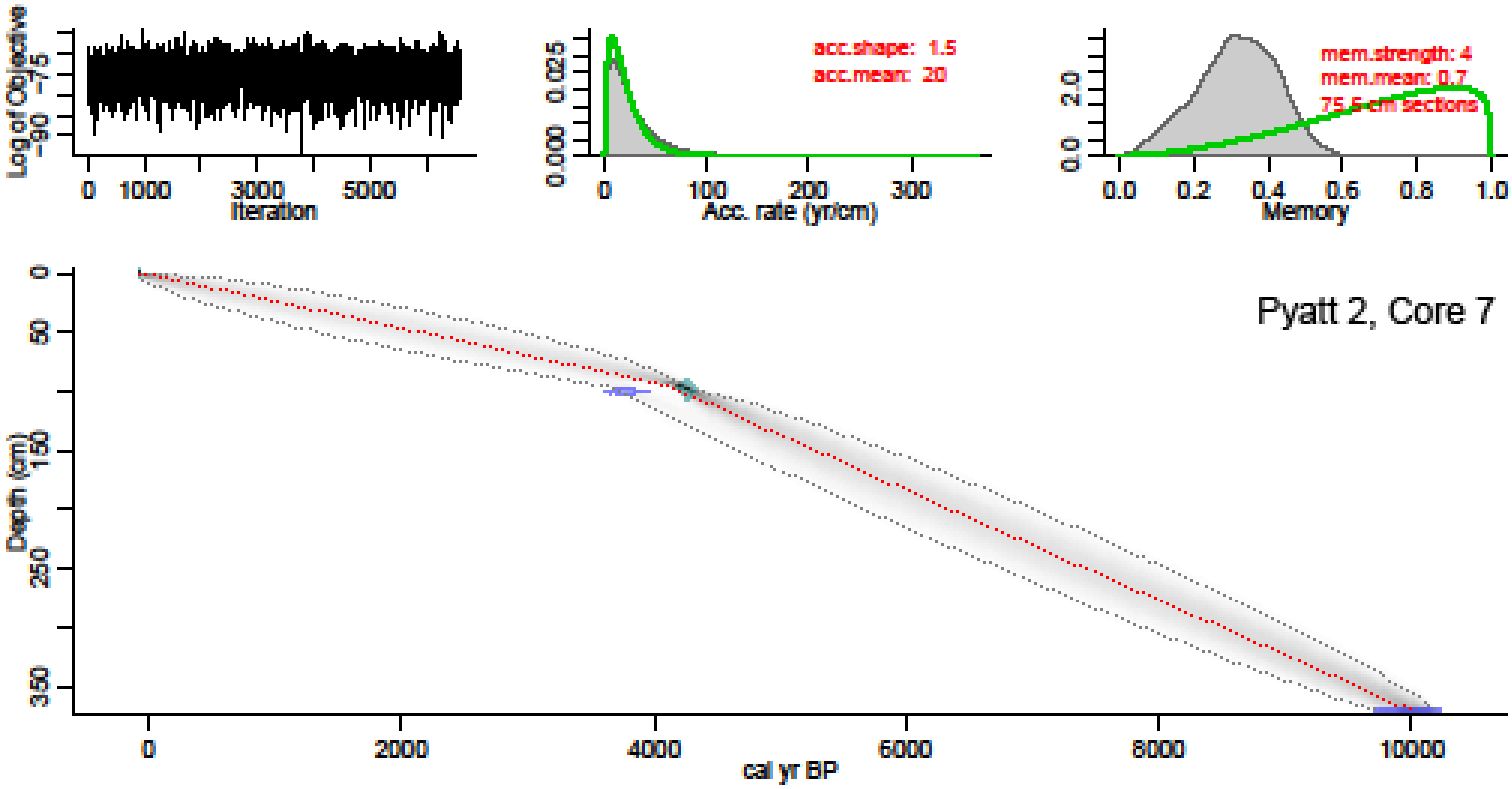
Supplementary Figure 4. Age depth model derived from radiocarbon dating and tephra deposits for Pyatt 2, Core 4.



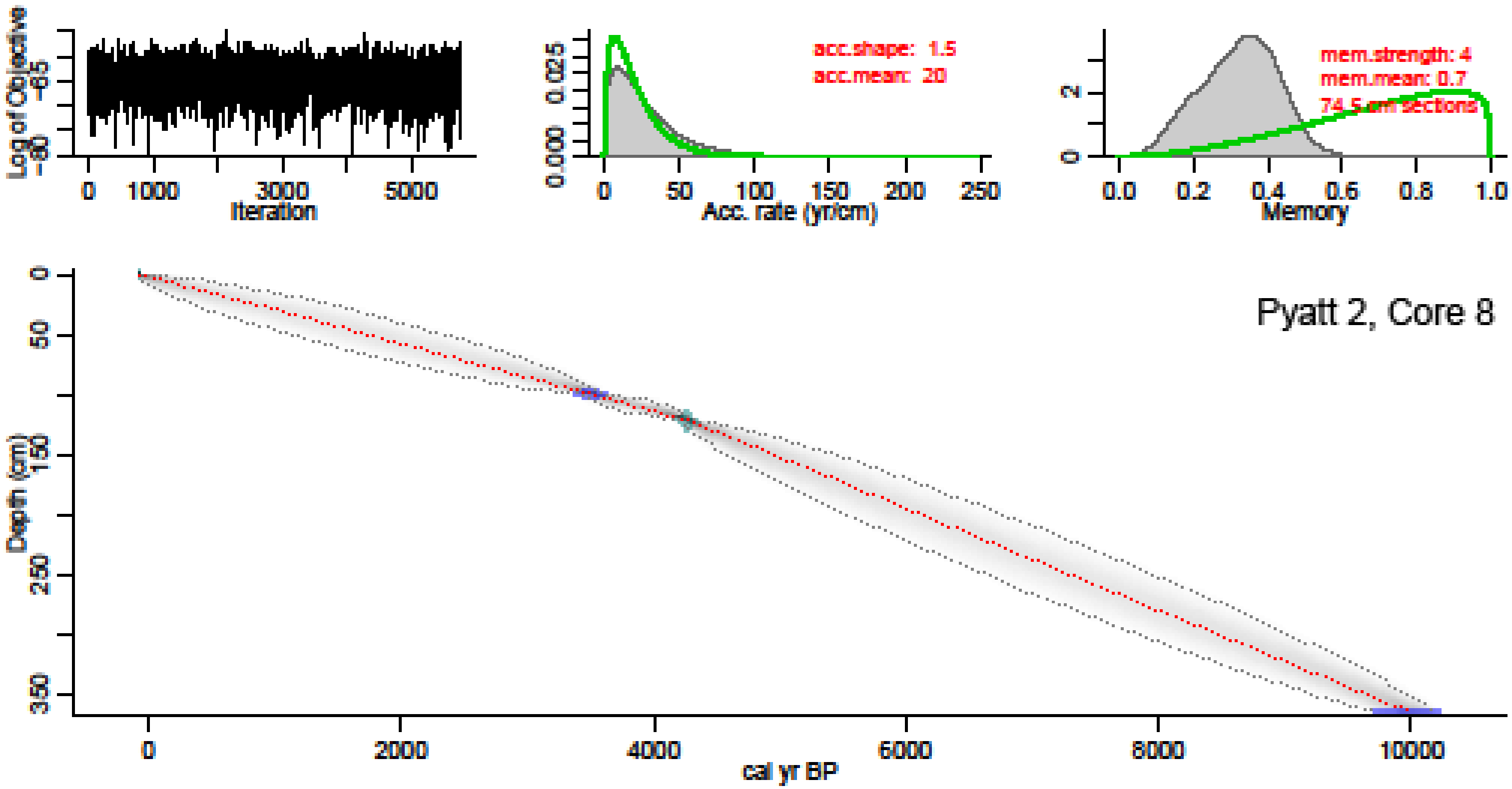
Supplementary Figure 5. Age depth model derived from radiocarbon dating and tephra deposits for Pyatt 2, Core 5.



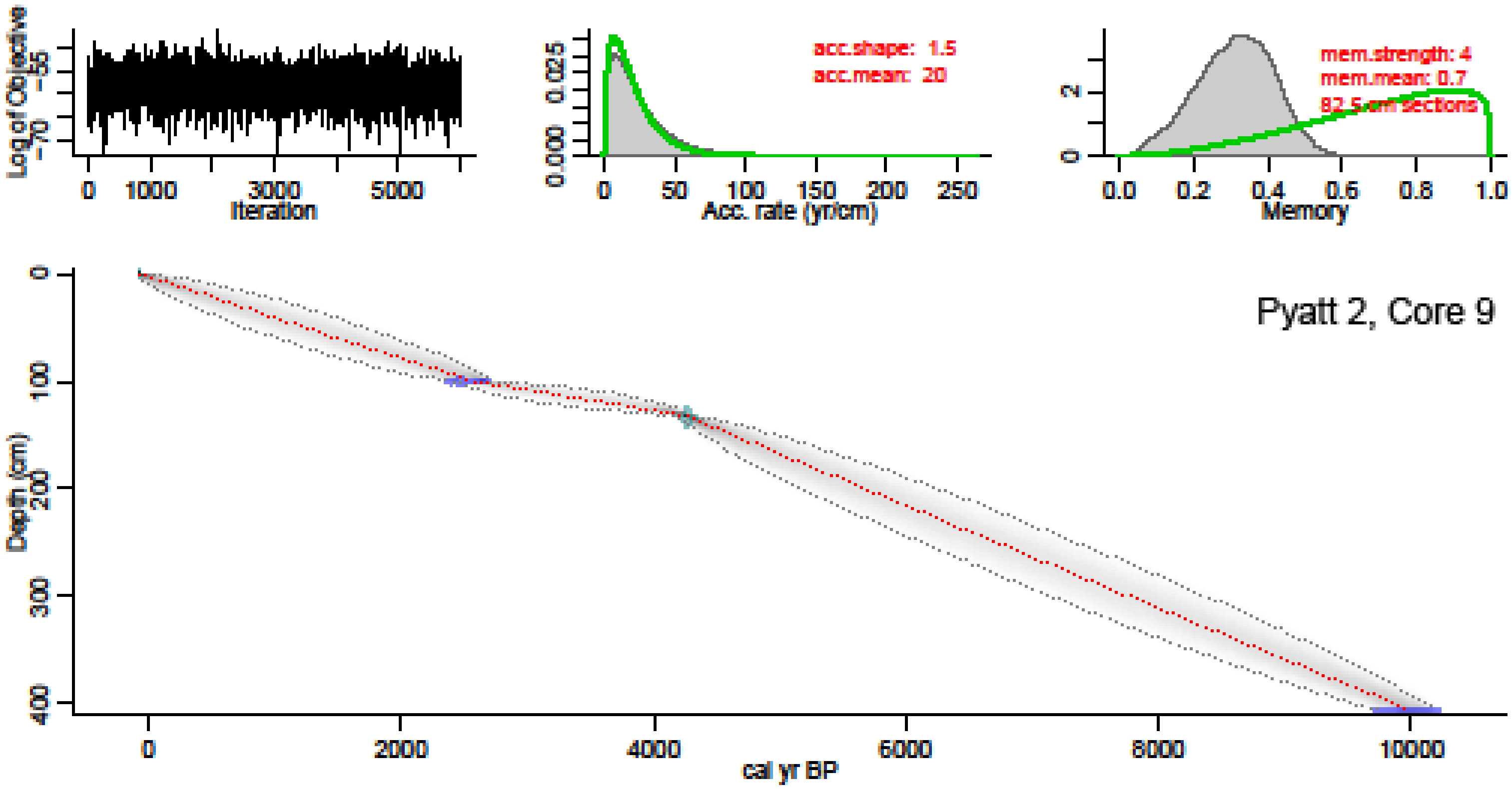
Supplementary Figure 6. Age depth model derived from radiocarbon dating and tephra deposits for Pyatt 2, Core 6.



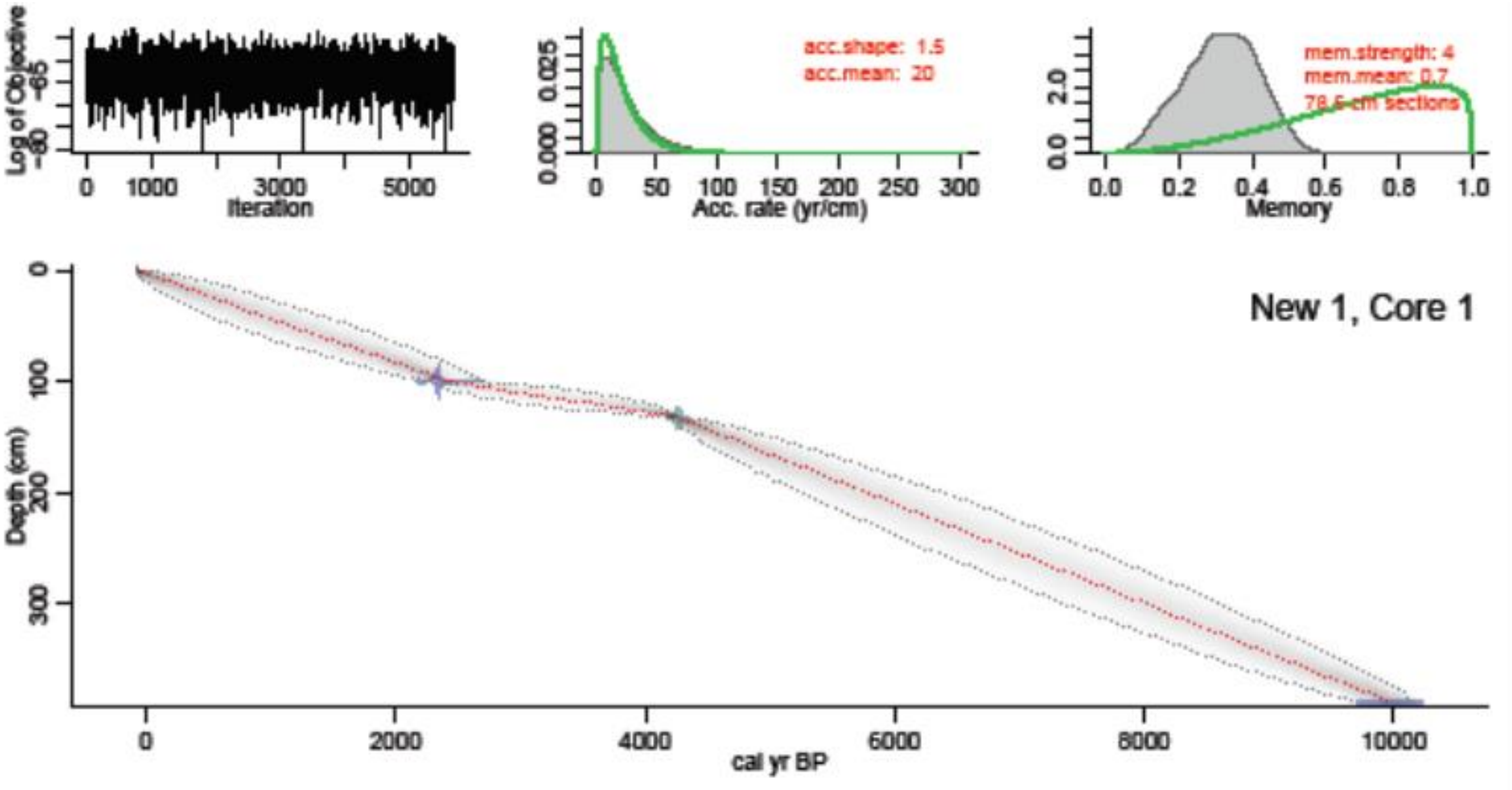
Supplementary Figure 7. Age depth model derived from radiocarbon dating and tephra deposits for Pyatt 2, Core 7.



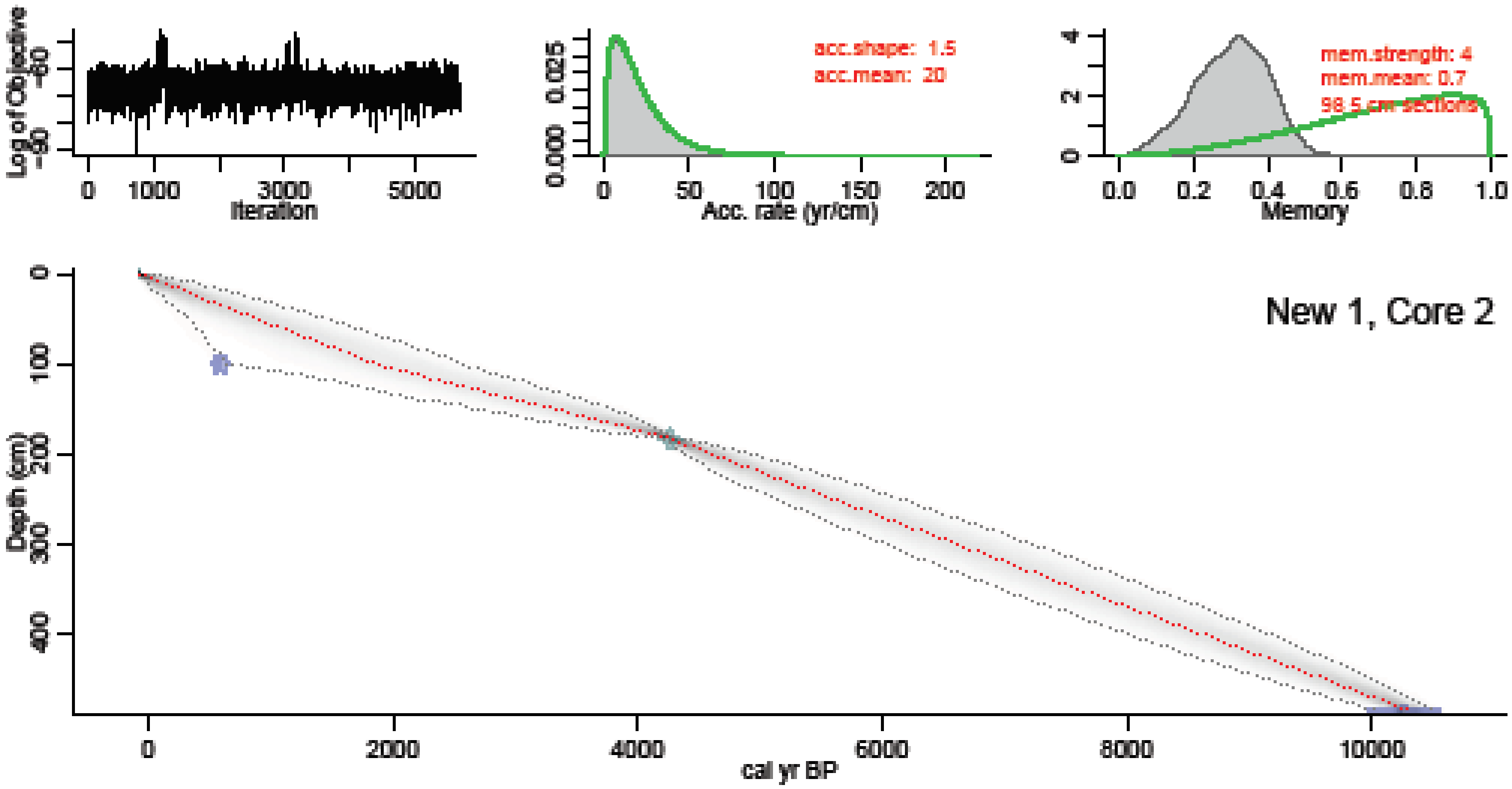
Supplementary Figure 8. Age depth model derived from radiocarbon dating and tephra deposits for Pyatt 2, Core 8.



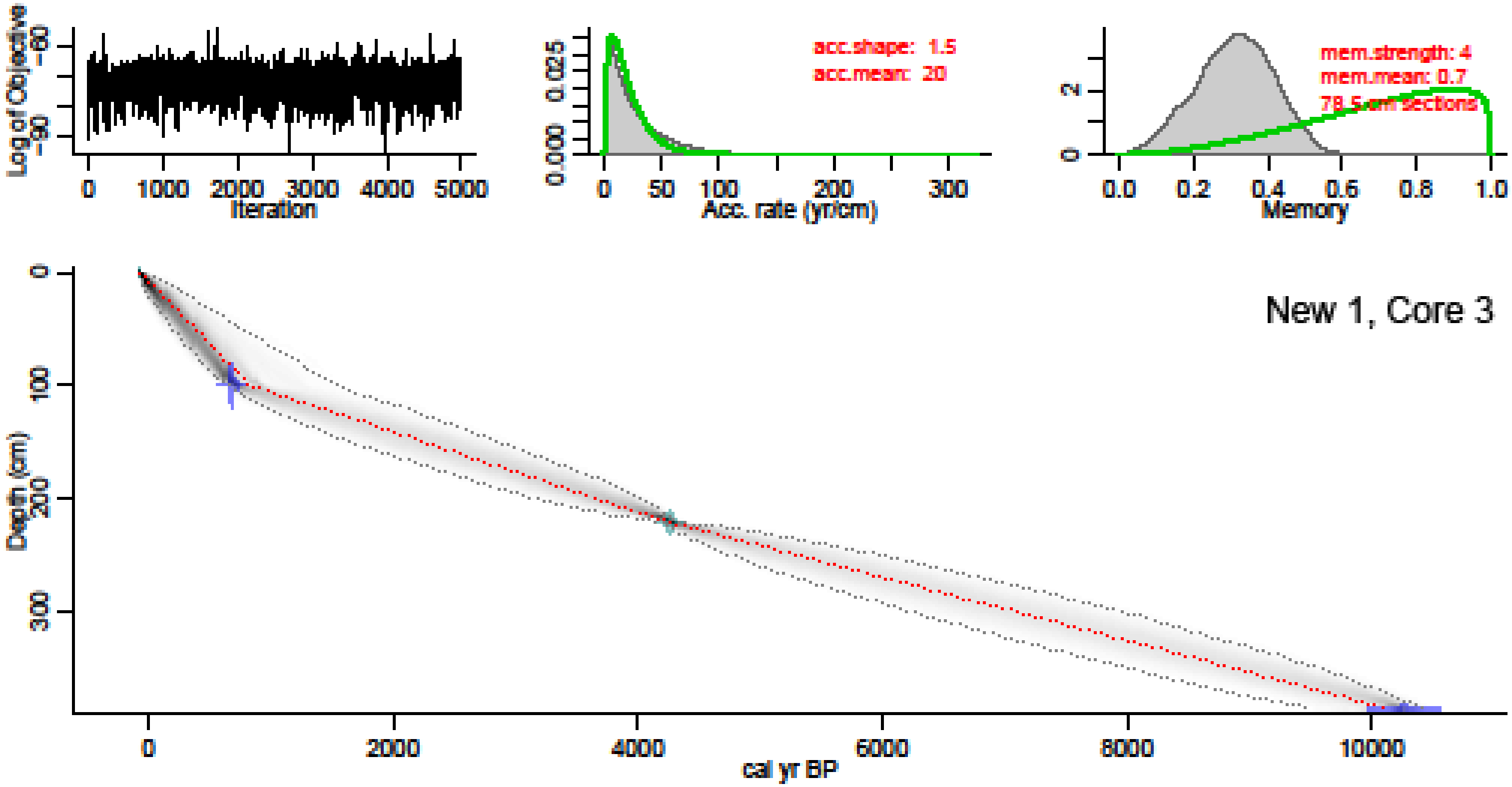
Supplementary Figure 9. Age depth model derived from radiocarbon dating and tephra deposits for Pyatt 2, Core 9.



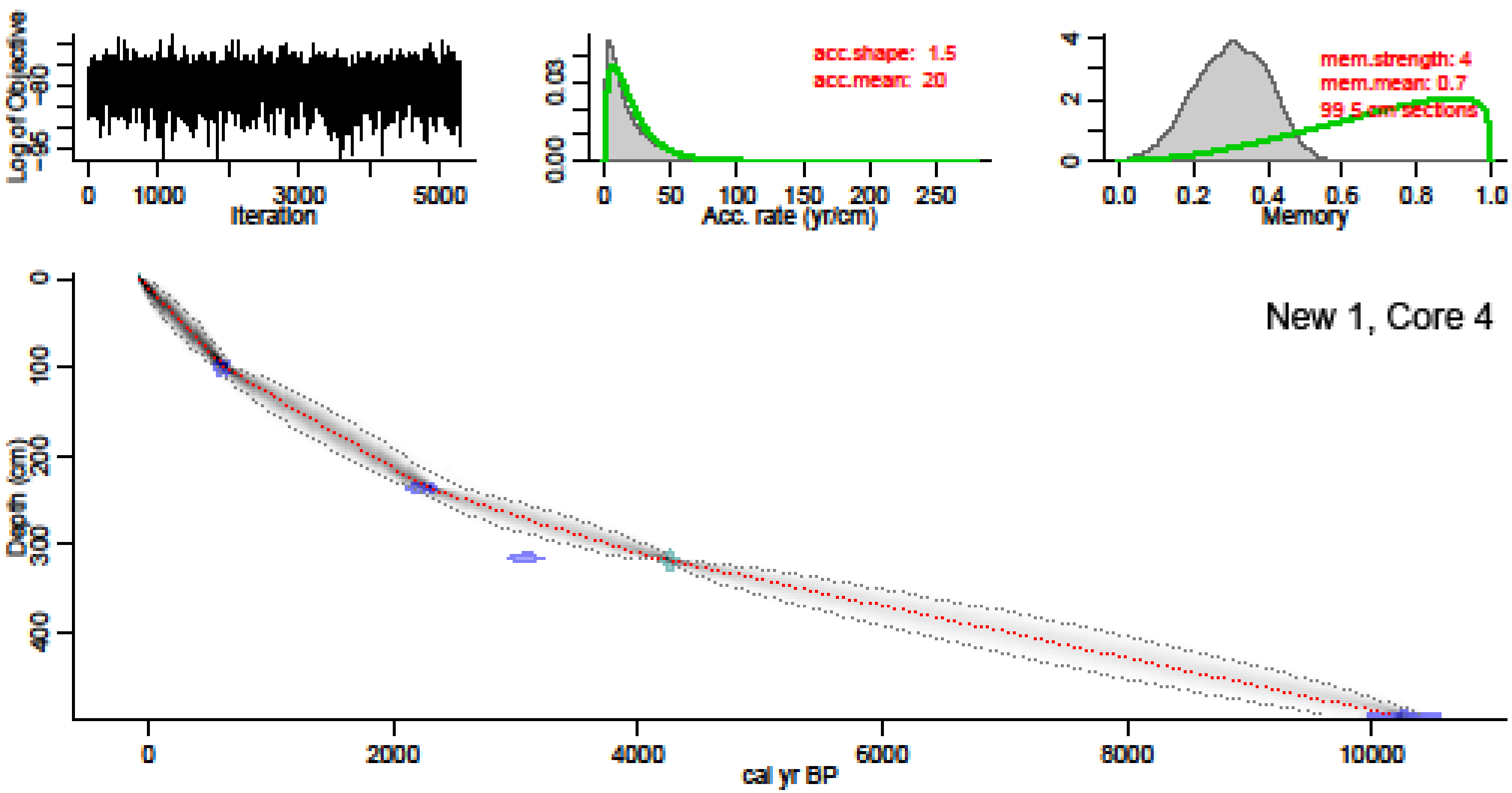
Supplementary Figure 10. Age depth model derived from radiocarbon dating and tephra deposits for New 1, Core 1.



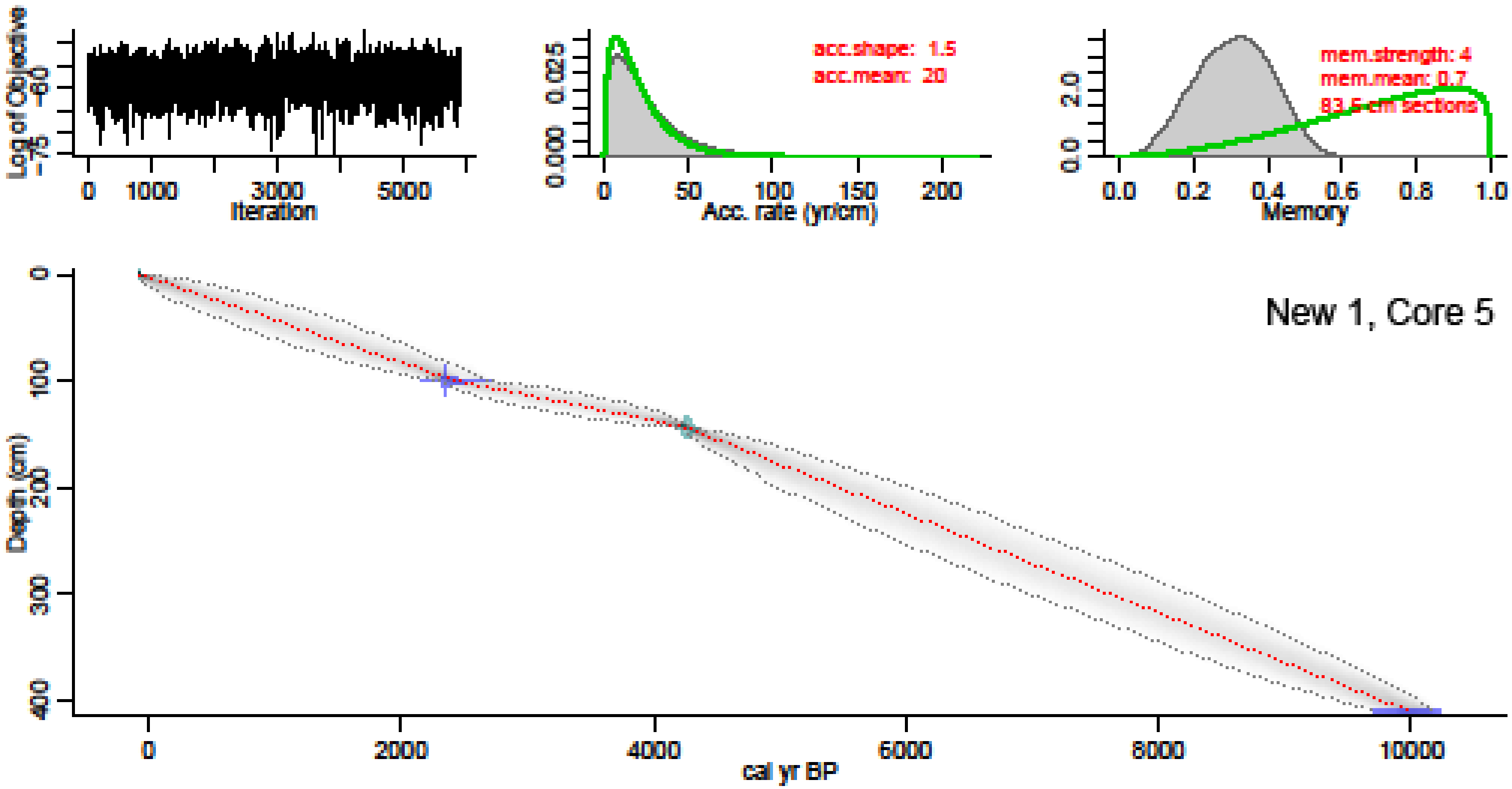
Supplementary Figure 11. Age depth model derived from radiocarbon dating and tephra deposits for New 1, Core 2.



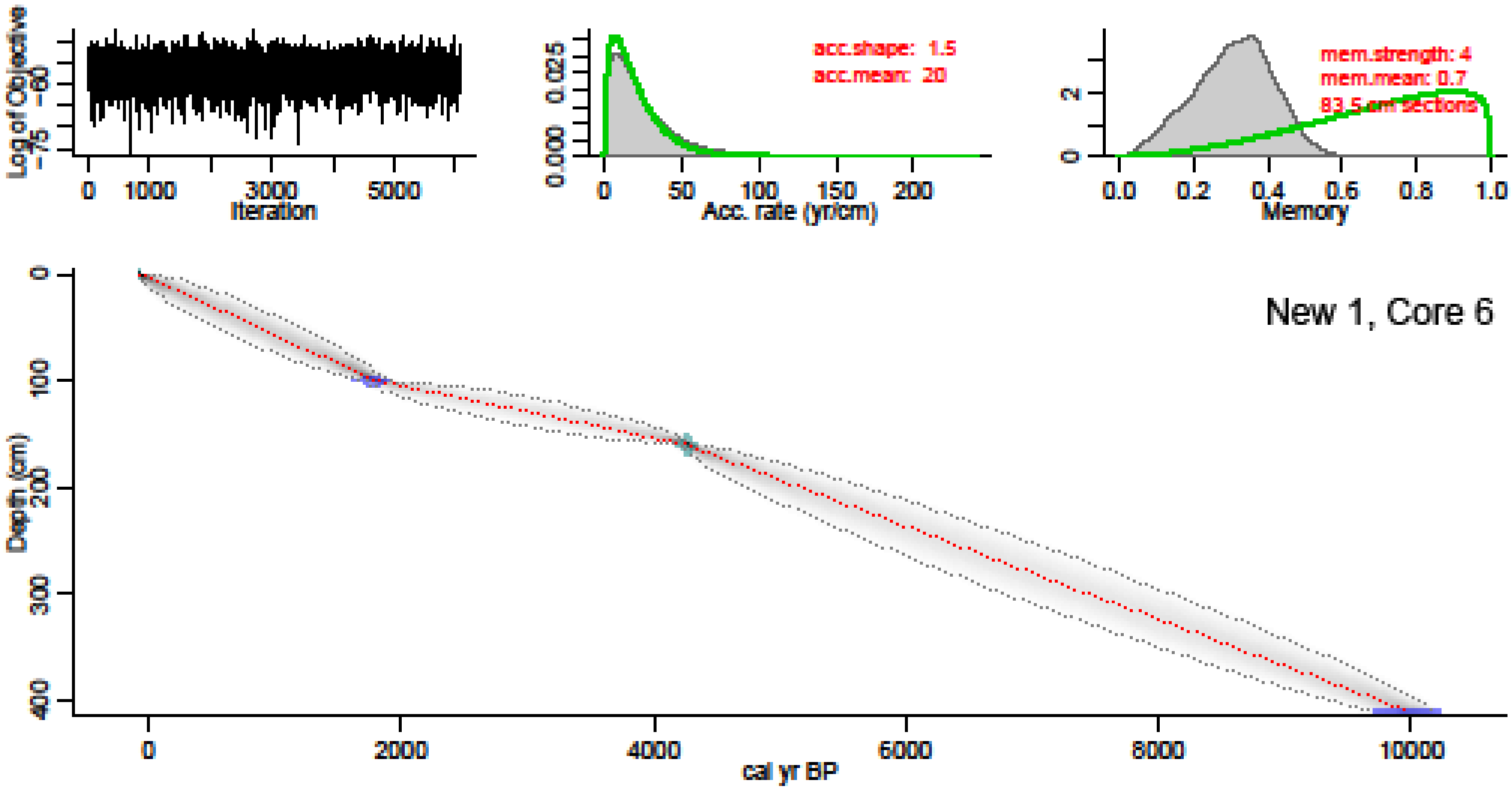
Supplementary Figure 12. Age depth model derived from radiocarbon dating and tephra deposits for New 1, Core 3.



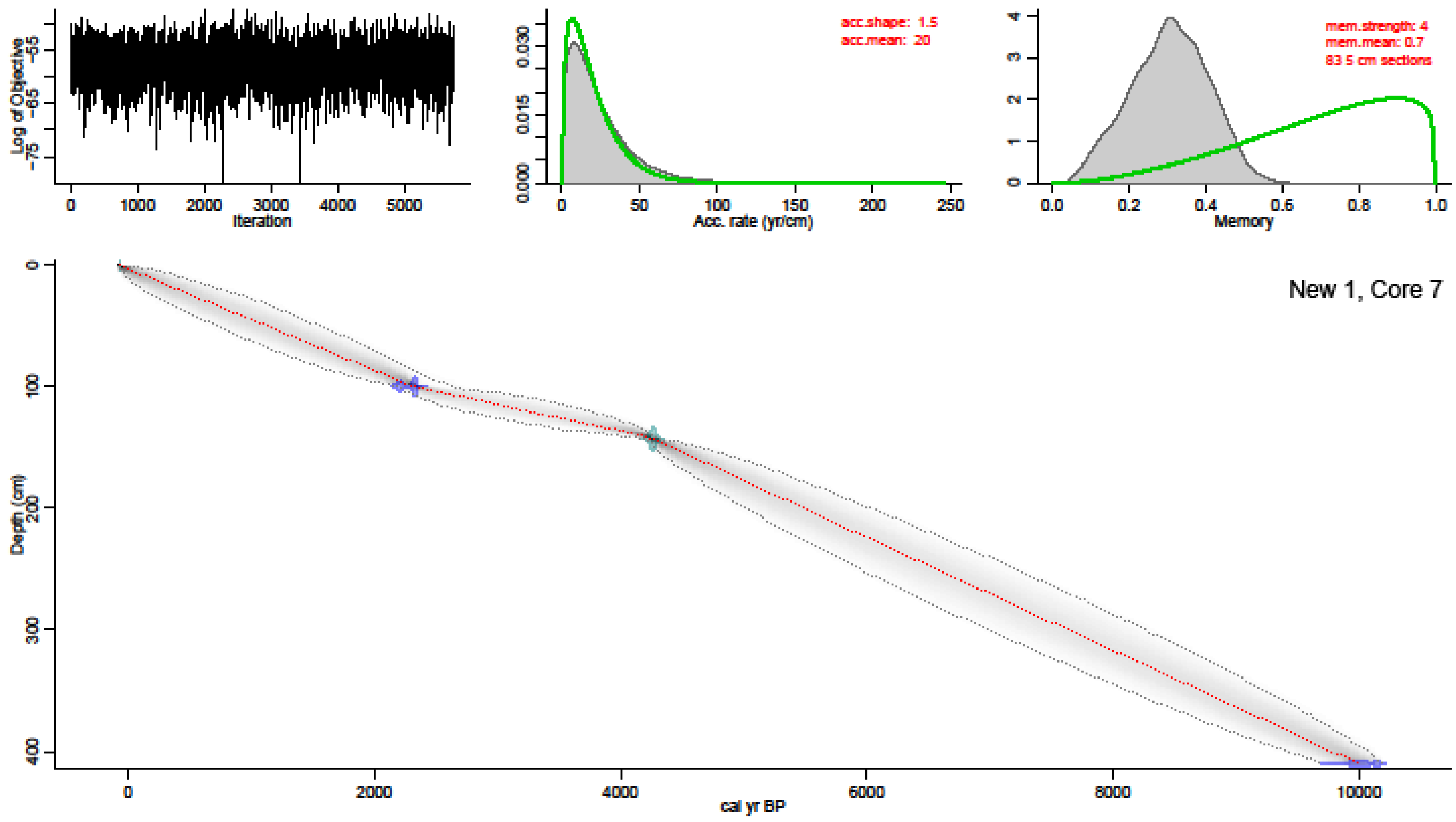
Supplementary Figure 13. Age depth model derived from radiocarbon dating and tephra deposits for New 1, Core 4.



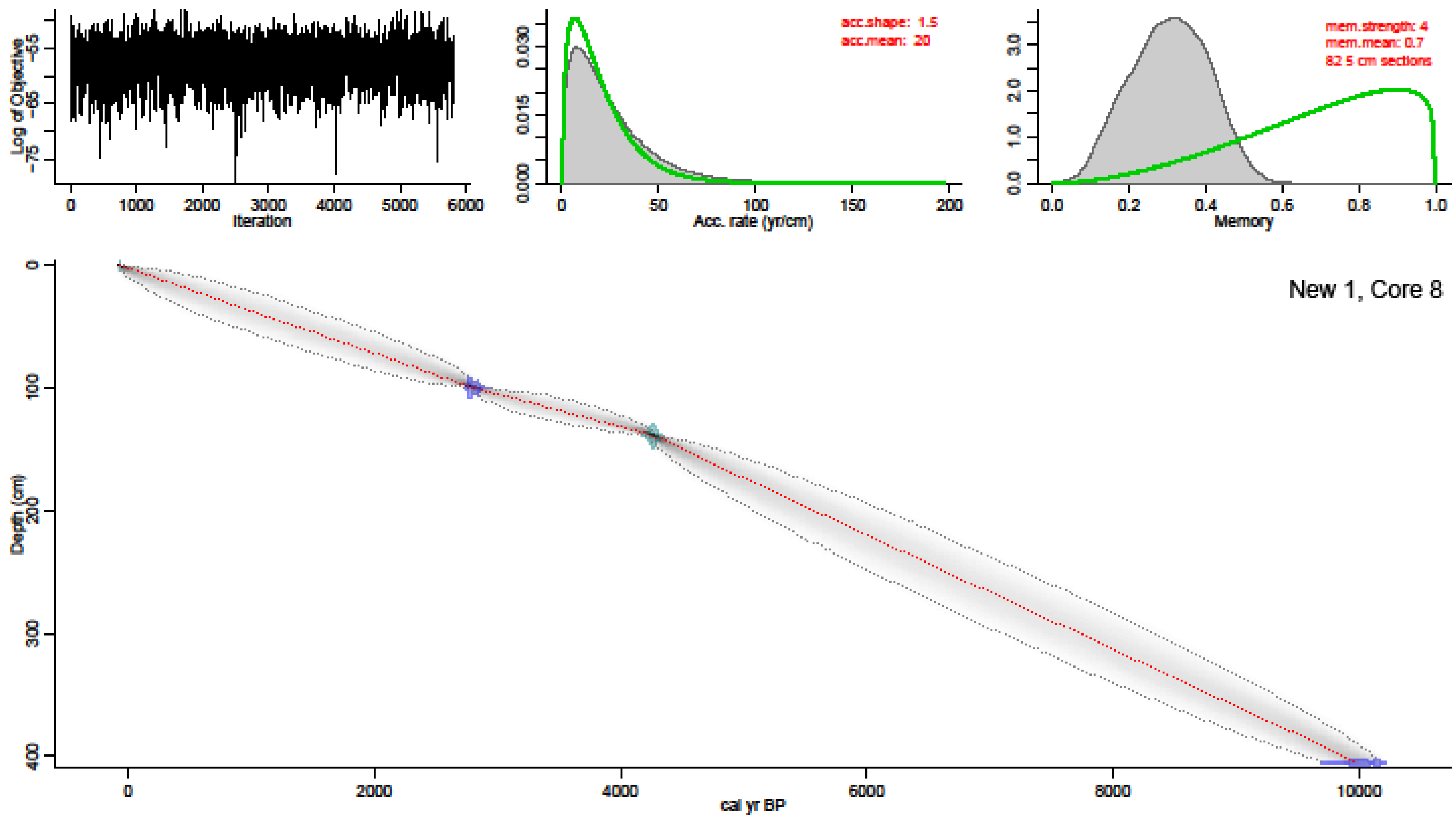
Supplementary Figure 14. Age depth model derived from radiocarbon dating and tephra deposits for New 1, Core 5.



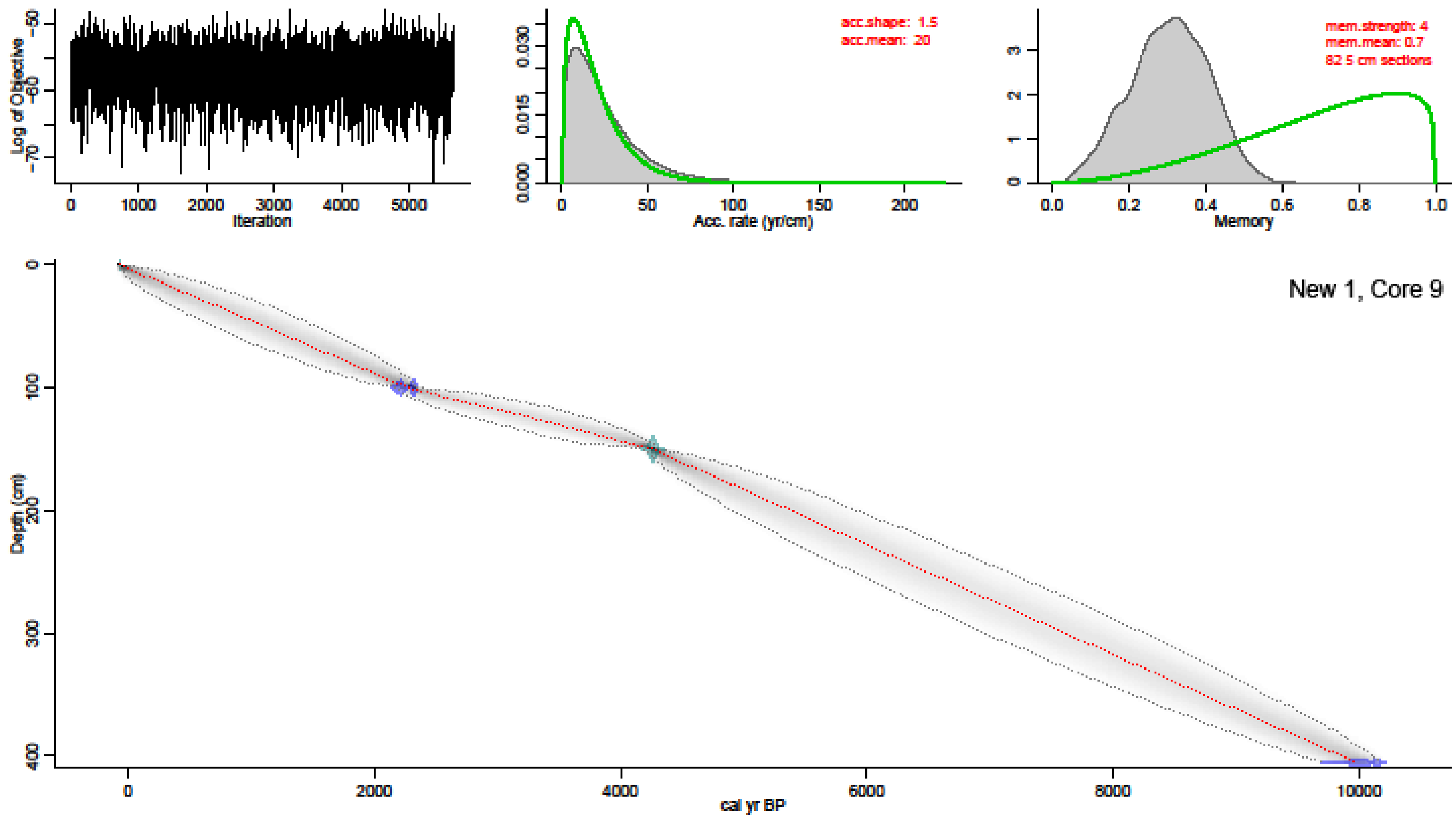
Supplementary Figure 15. Age depth model derived from radiocarbon dating and tephra deposits for New 1, Core 6.



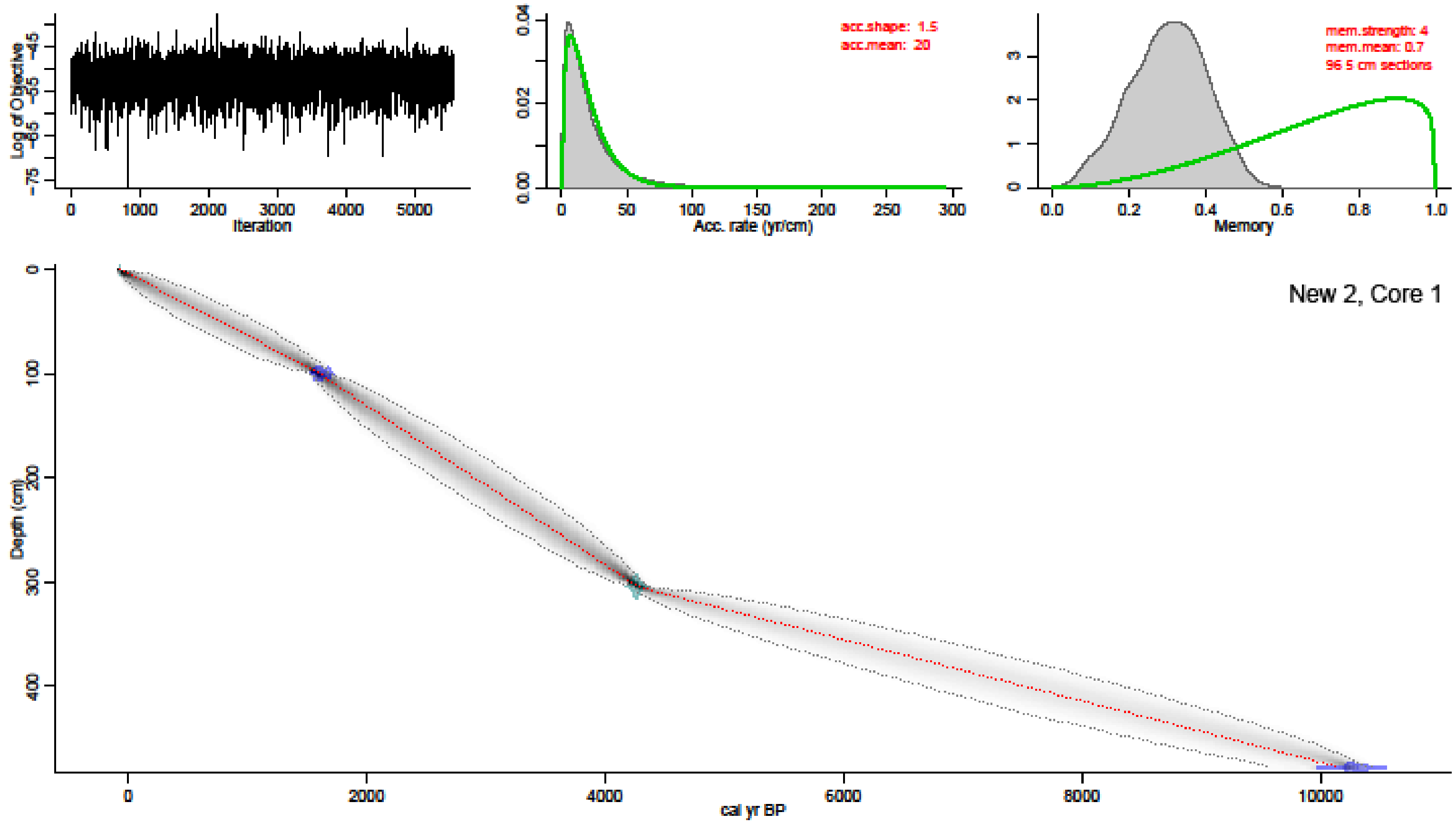
Supplementary Figure 16. Age depth model derived from radiocarbon dating and tephra deposits for New 1, Core 7.



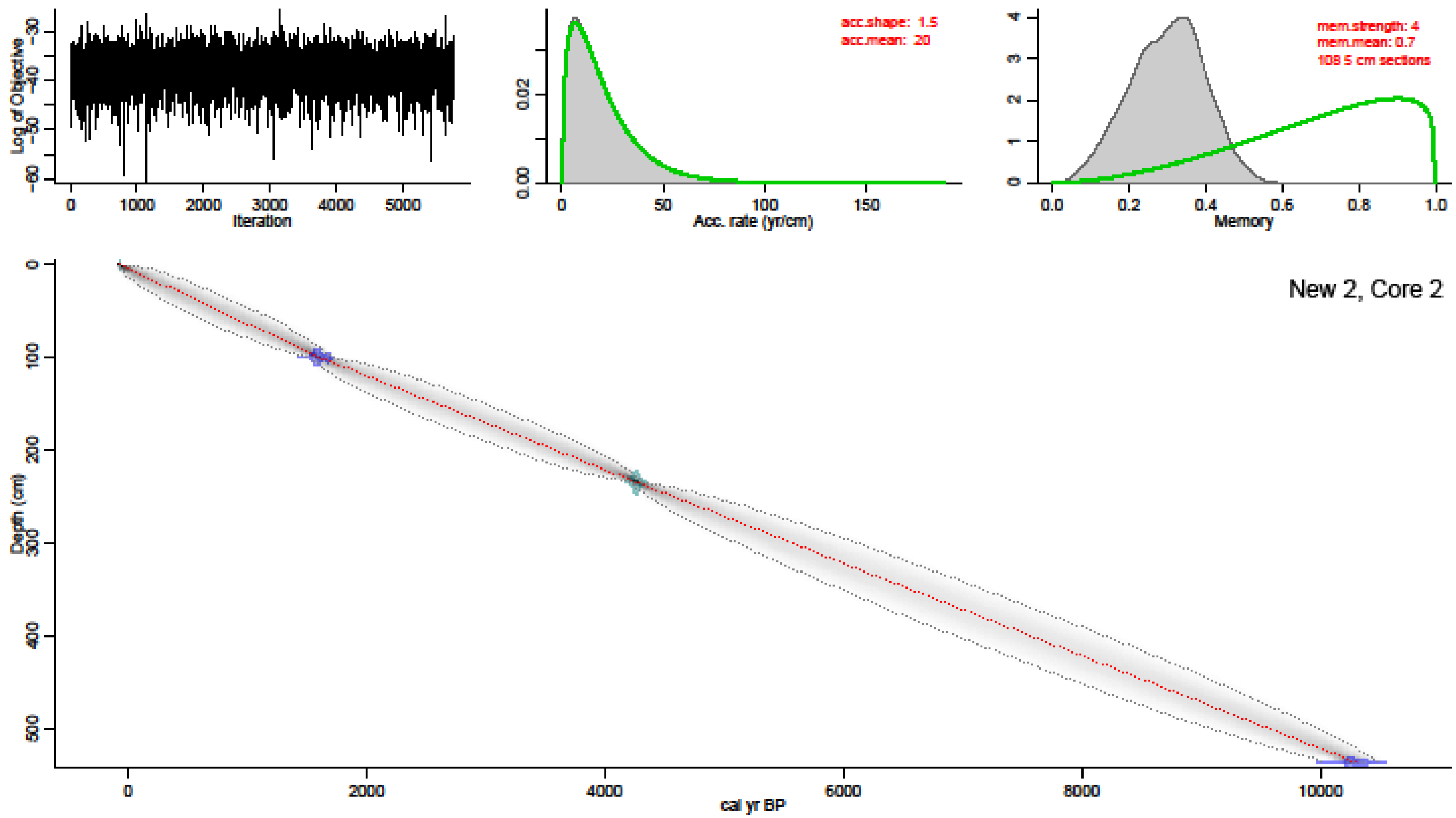
Supplementary Figure 17. Age depth model derived from radiocarbon dating and tephra deposits for New 1, Core 8.



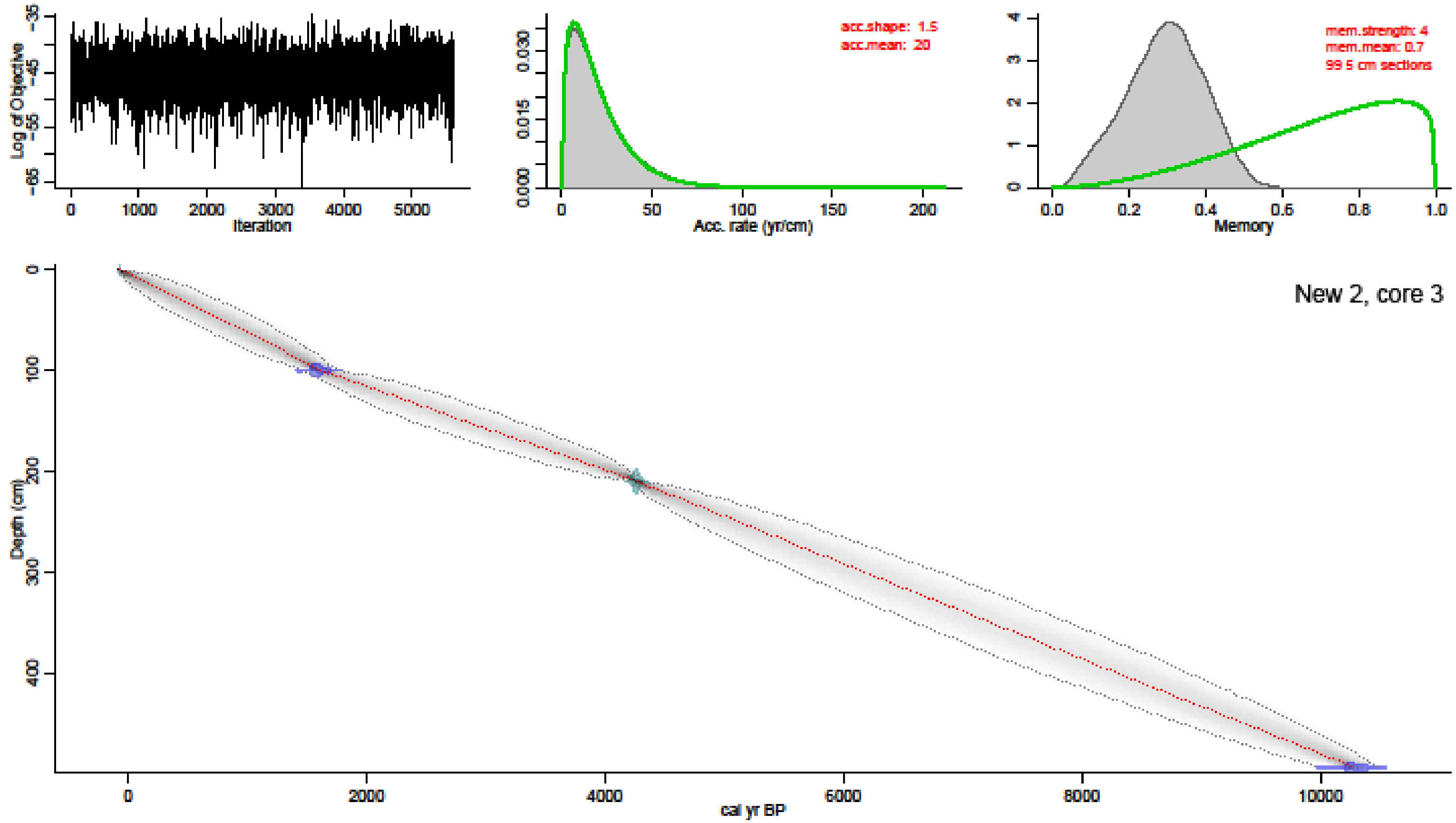
Supplementary Figure 18. Age depth model derived from radiocarbon dating and tephra deposits for New 1, Core 9.



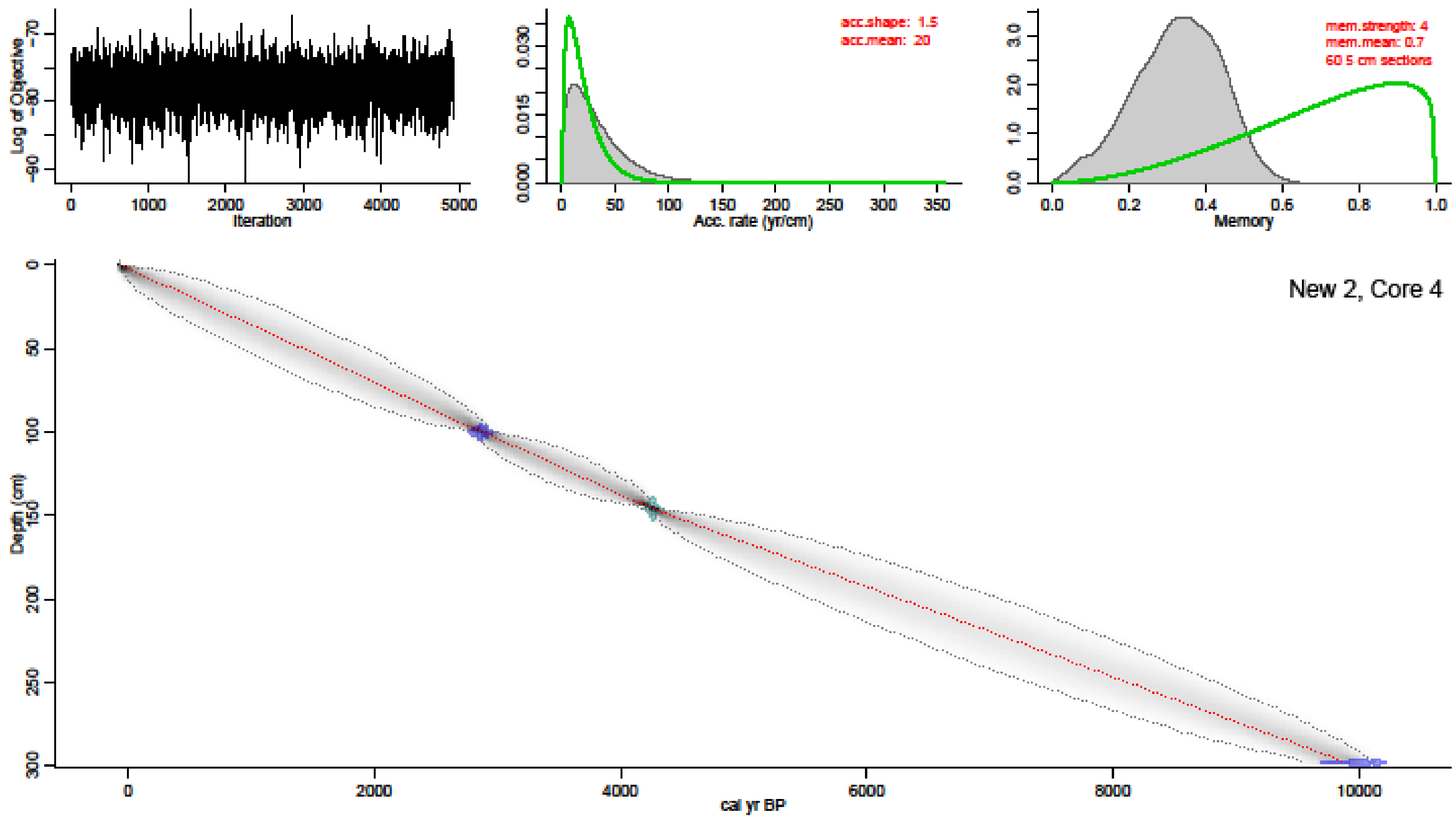
Supplementary Figure 19. Age depth model derived from radiocarbon dating and tephra deposits for New 2, Core 1.



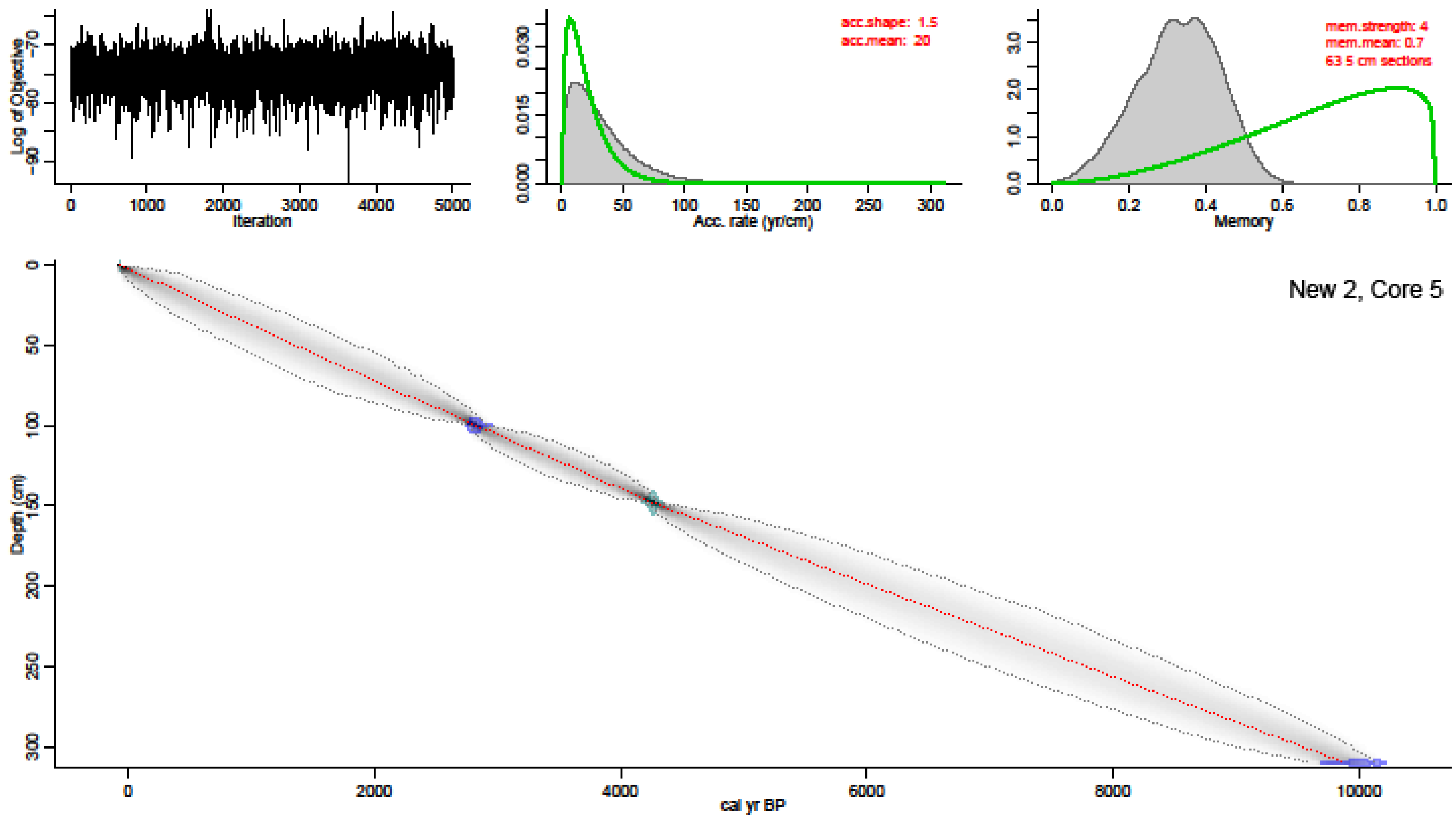
Supplementary Figure 20. Age depth model derived from radiocarbon dating and tephra deposits for New 2, Core 2.



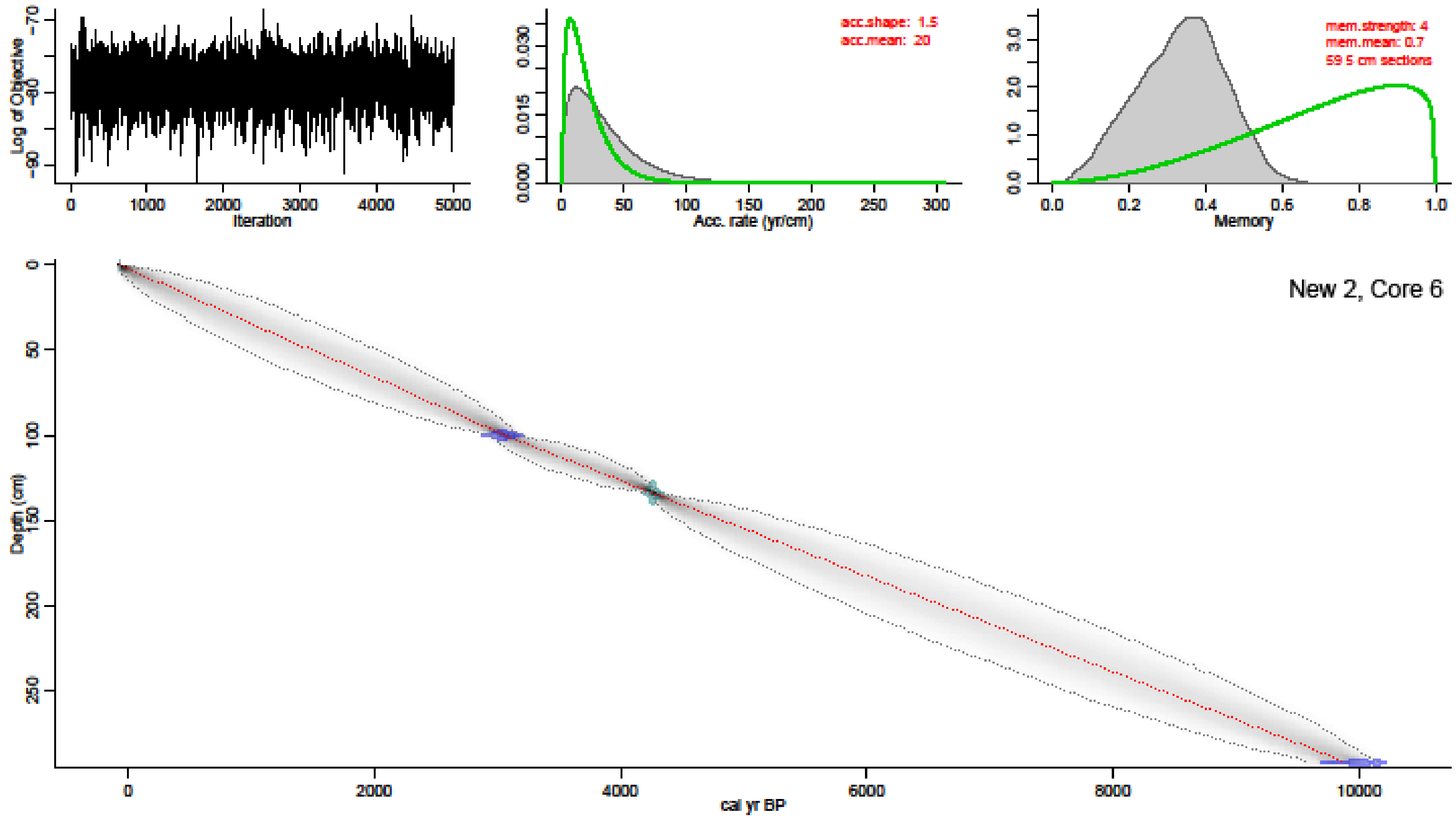
Supplementary Figure 21. Age depth model derived from radiocarbon dating and tephra deposits for New 2, Core 3.



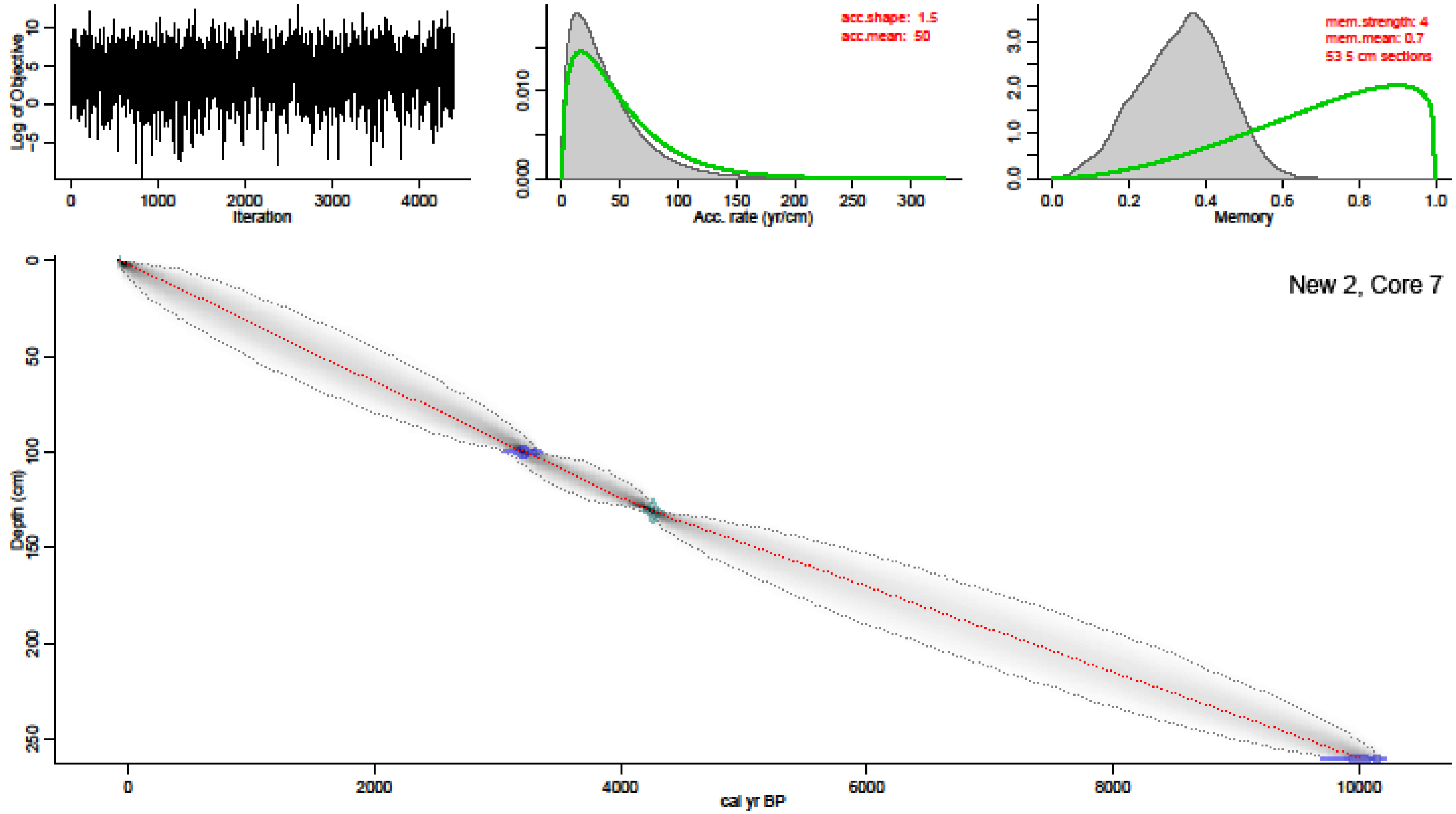
Supplementary Figure 22. Age depth model derived from radiocarbon dating and tephra deposits for New 2, Core 4.



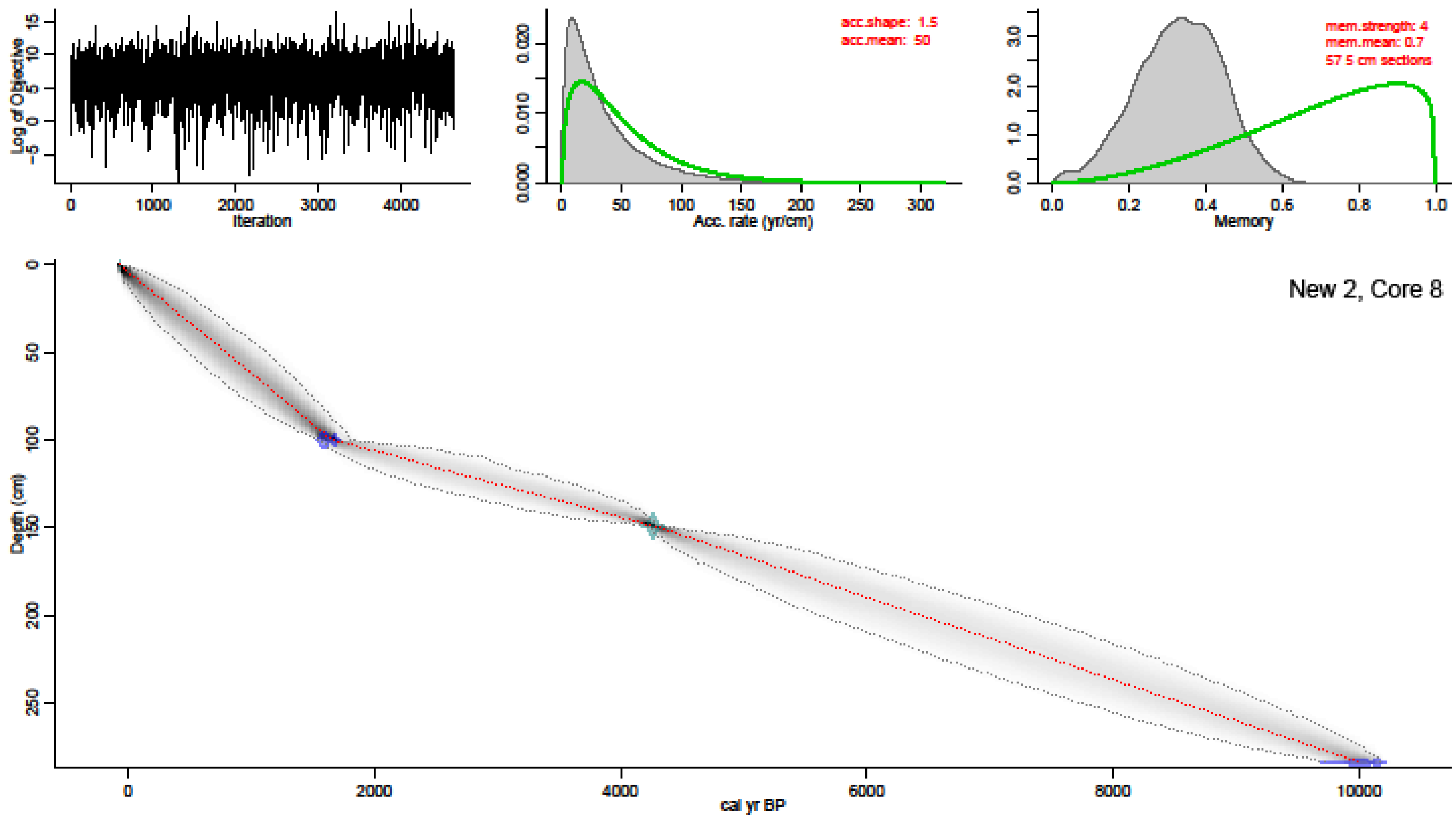
Supplementary Figure 23. Age depth model derived from radiocarbon dating and tephra deposits for New 2, Core 5.



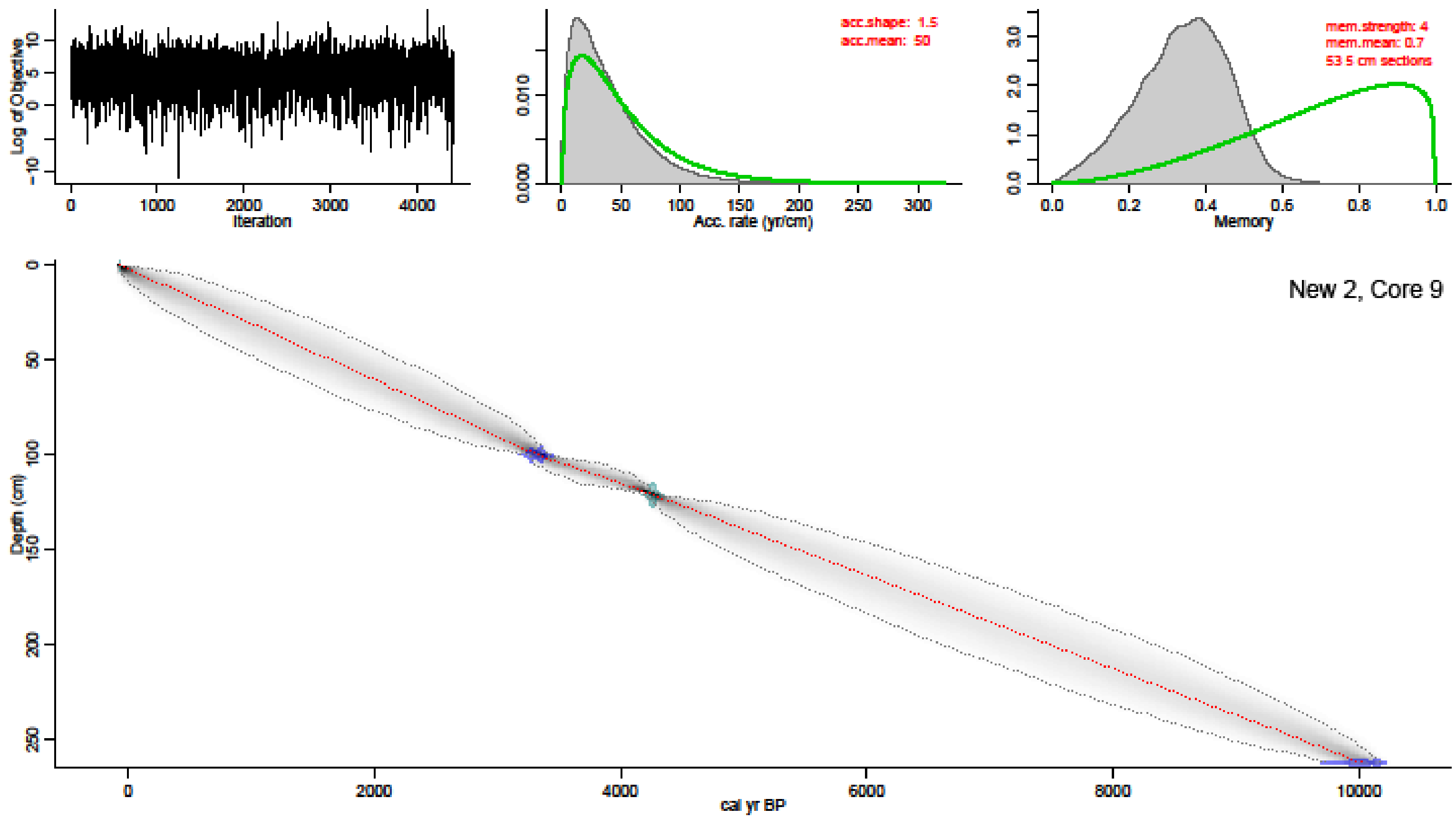
Supplementary Figure 24. Age depth model derived from radiocarbon dating and tephra deposits for New 2, Core 6.



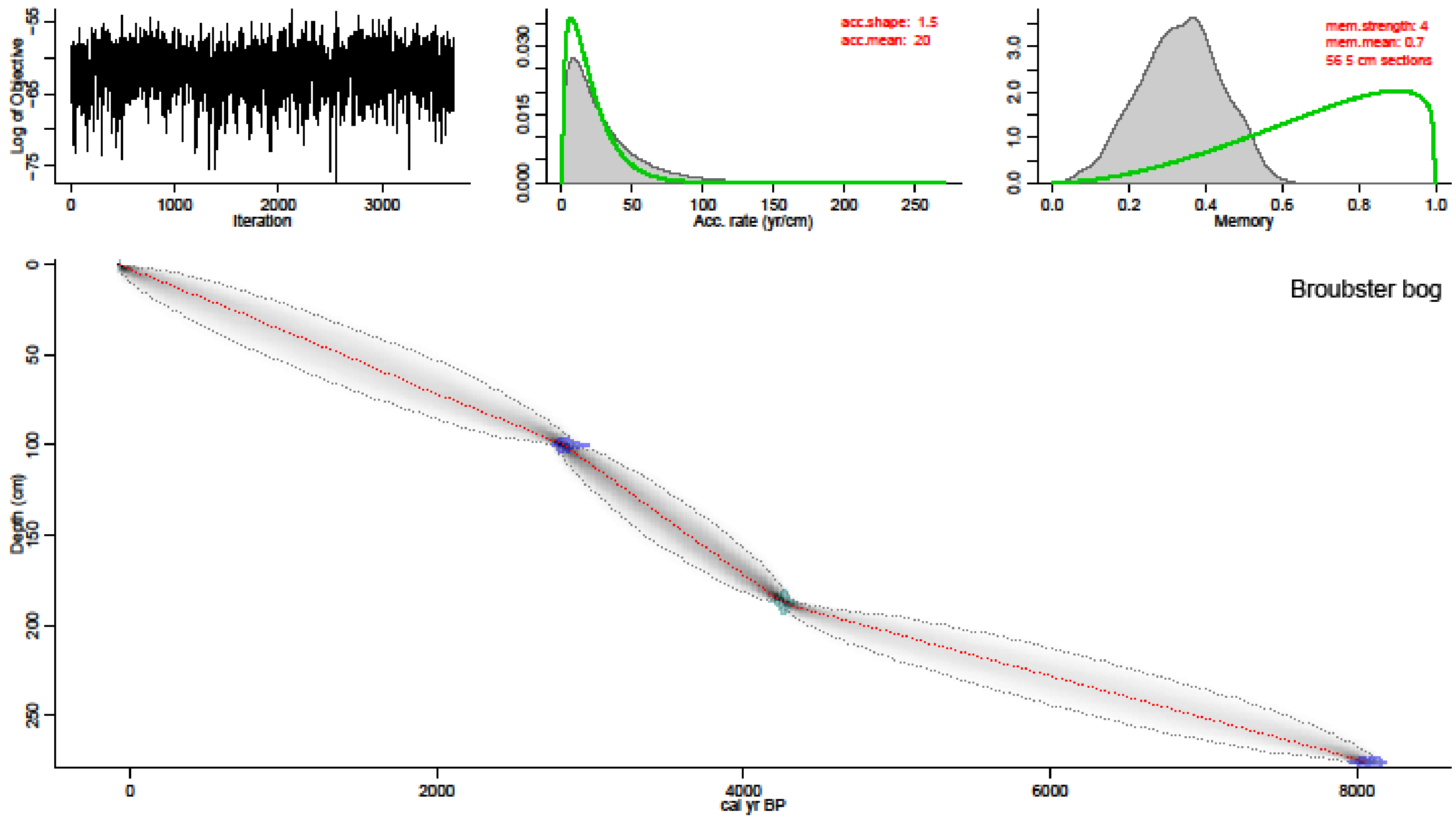
Supplementary Figure 25. Age depth model derived from radiocarbon dating and tephra deposits for New 2, Core 7.



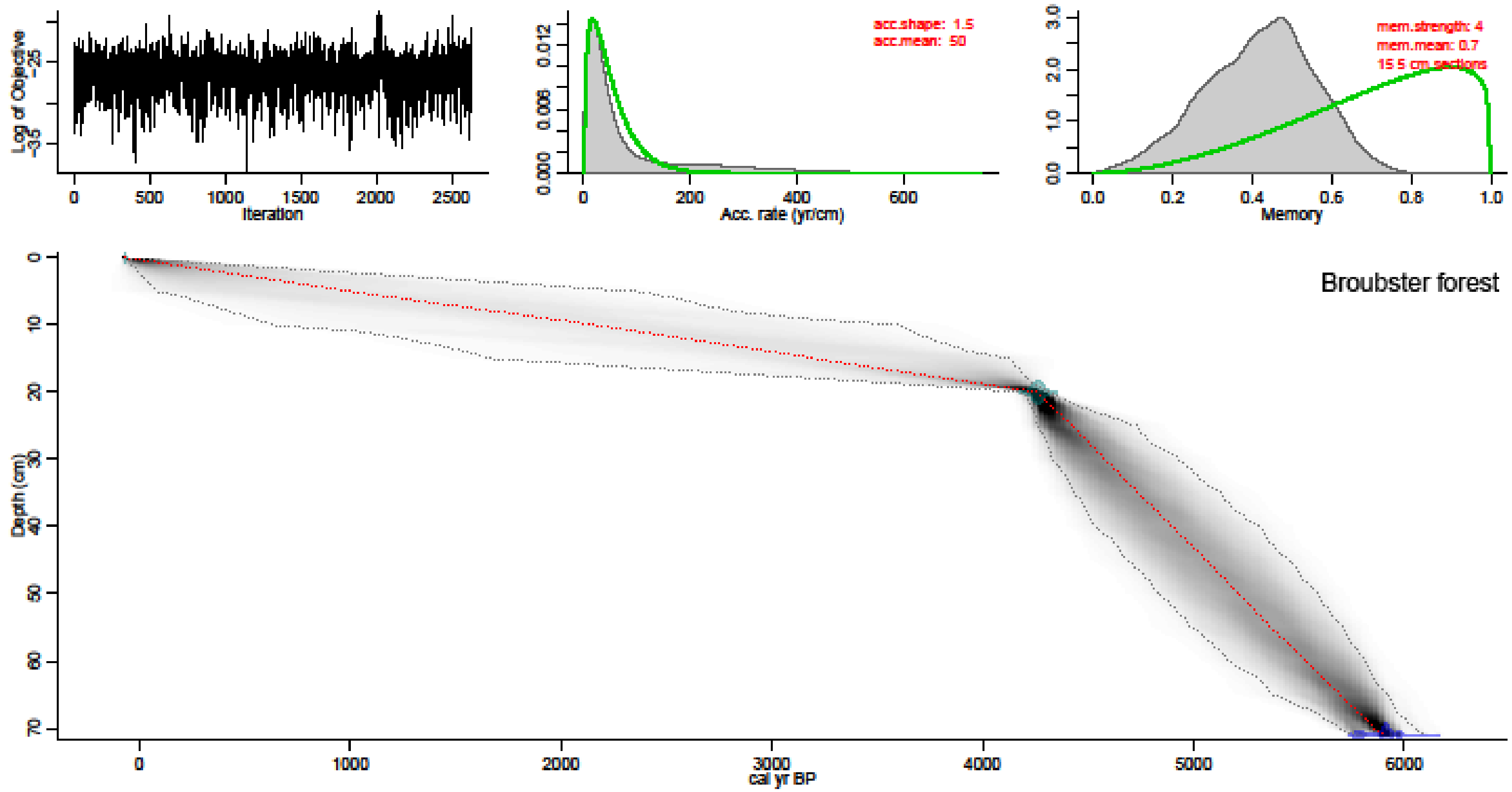
Supplementary Figure 26. Age depth model derived from radiocarbon dating and tephra deposits for New 2, Core 8.



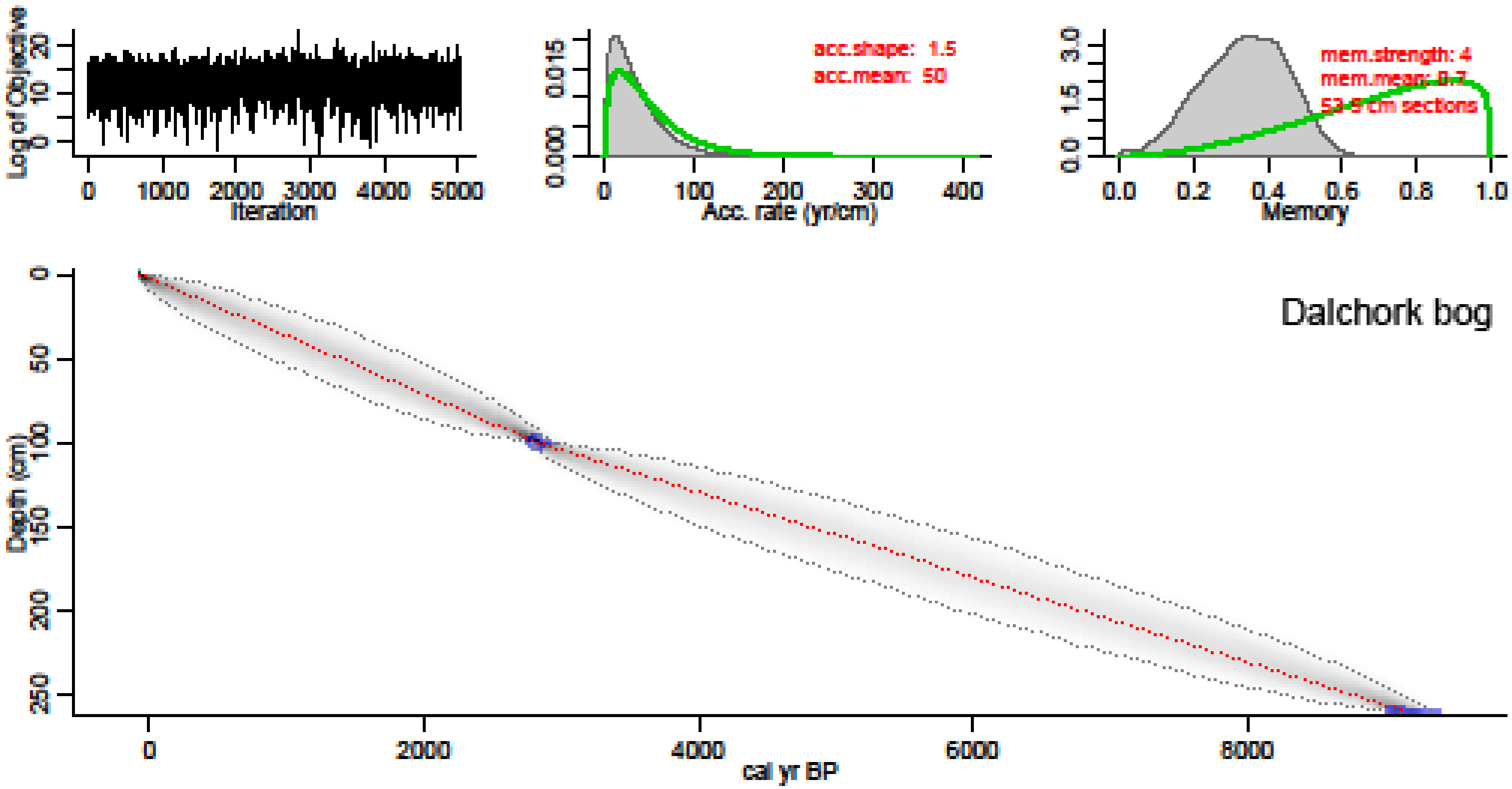
Supplementary Figure 27. Age depth model derived from radiocarbon dating and tephra deposits for New 2, Core 9.



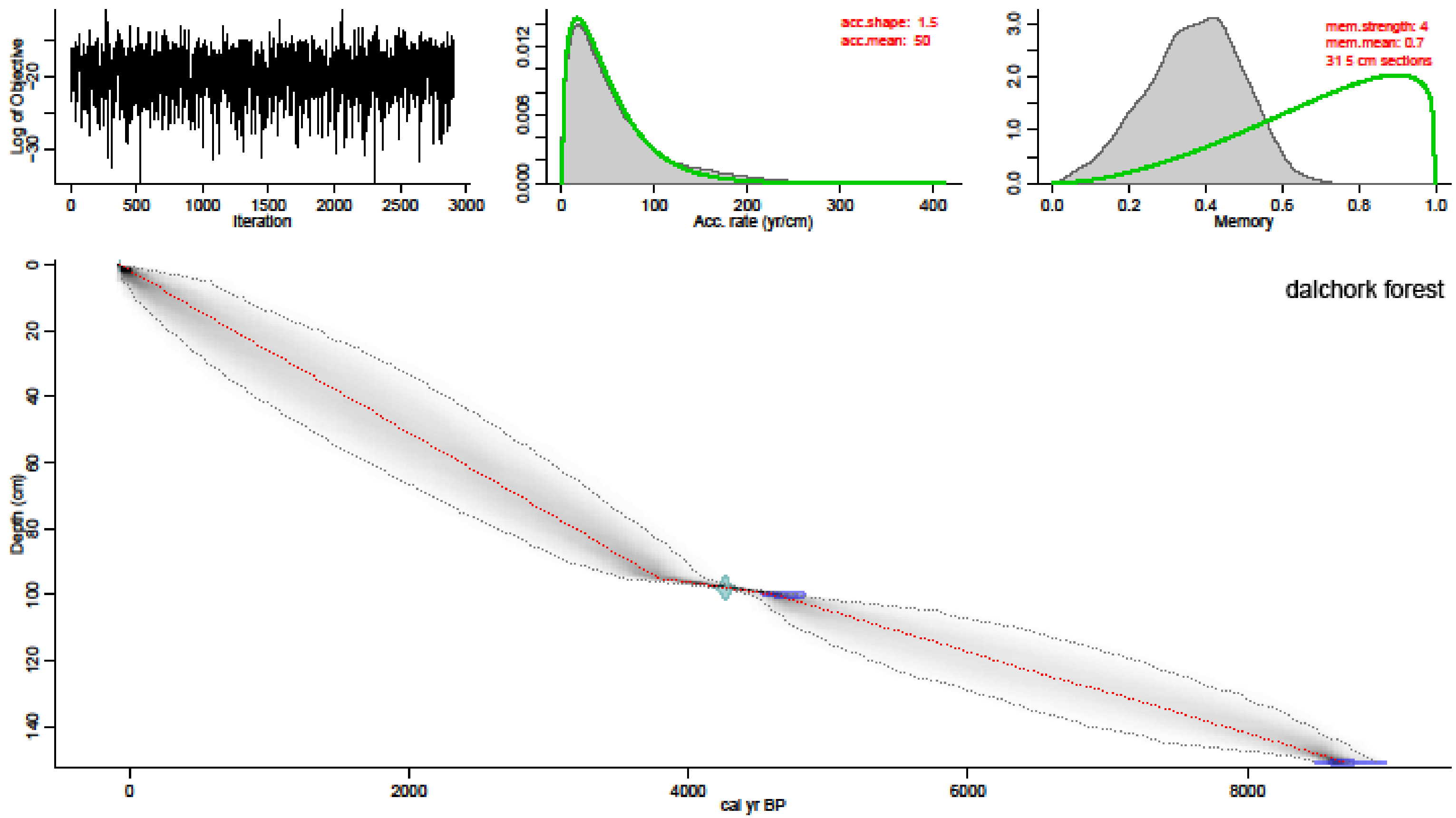
Supplementary Figure 38. Age depth model derived from radiocarbon dating and tephra deposits for Broubster undrained bog.



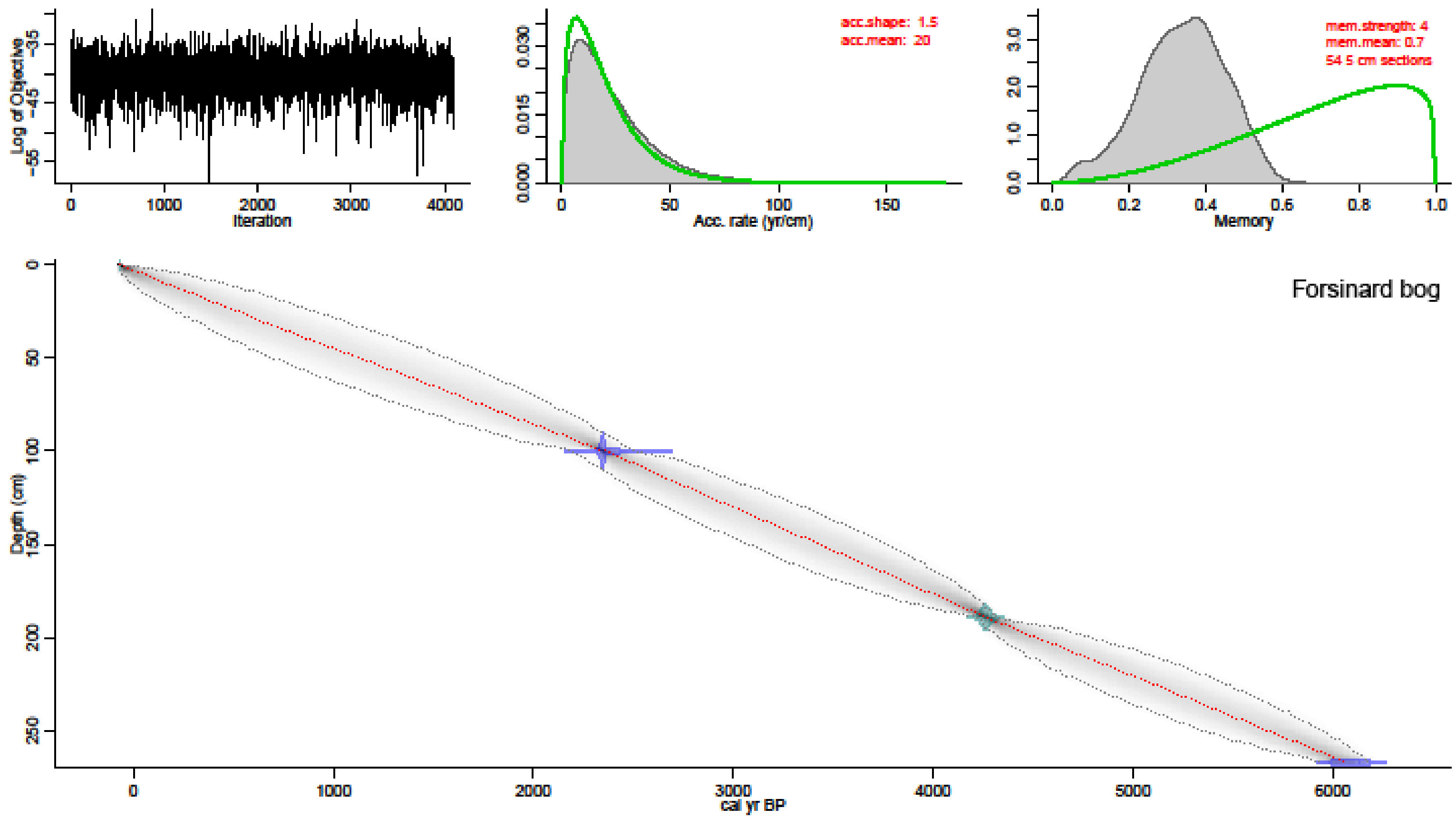
Supplementary Figure 29. Age depth model derived from radiocarbon dating and tephra deposits for Broubster afforested bog.



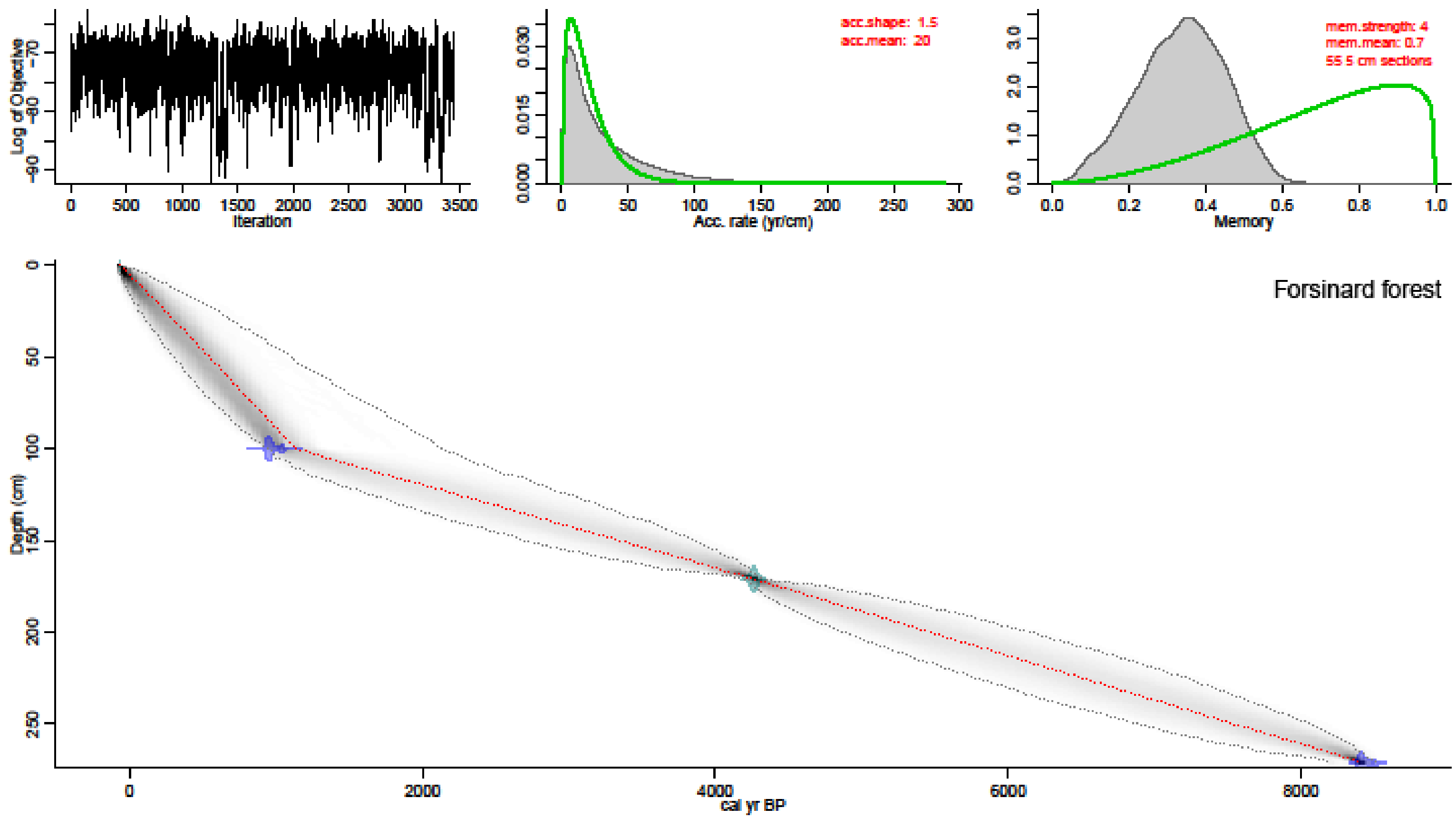
Supplementary Figure 30. Age depth model derived from radiocarbon dating for Dalchork undrained bog.



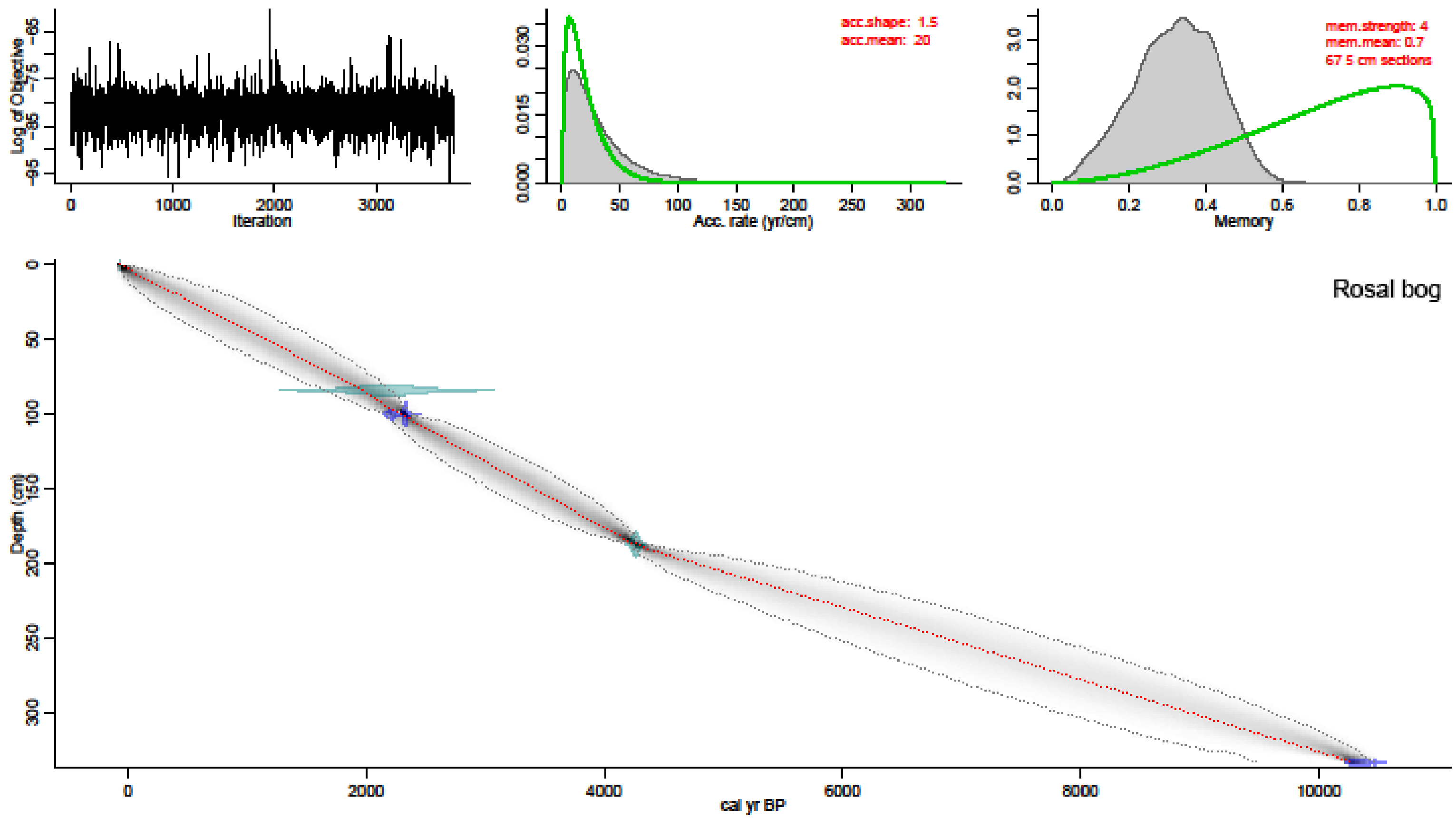
Supplementary Figure 31. Age depth model derived from radiocarbon dating and tephra deposits for Dalchork afforested bog.



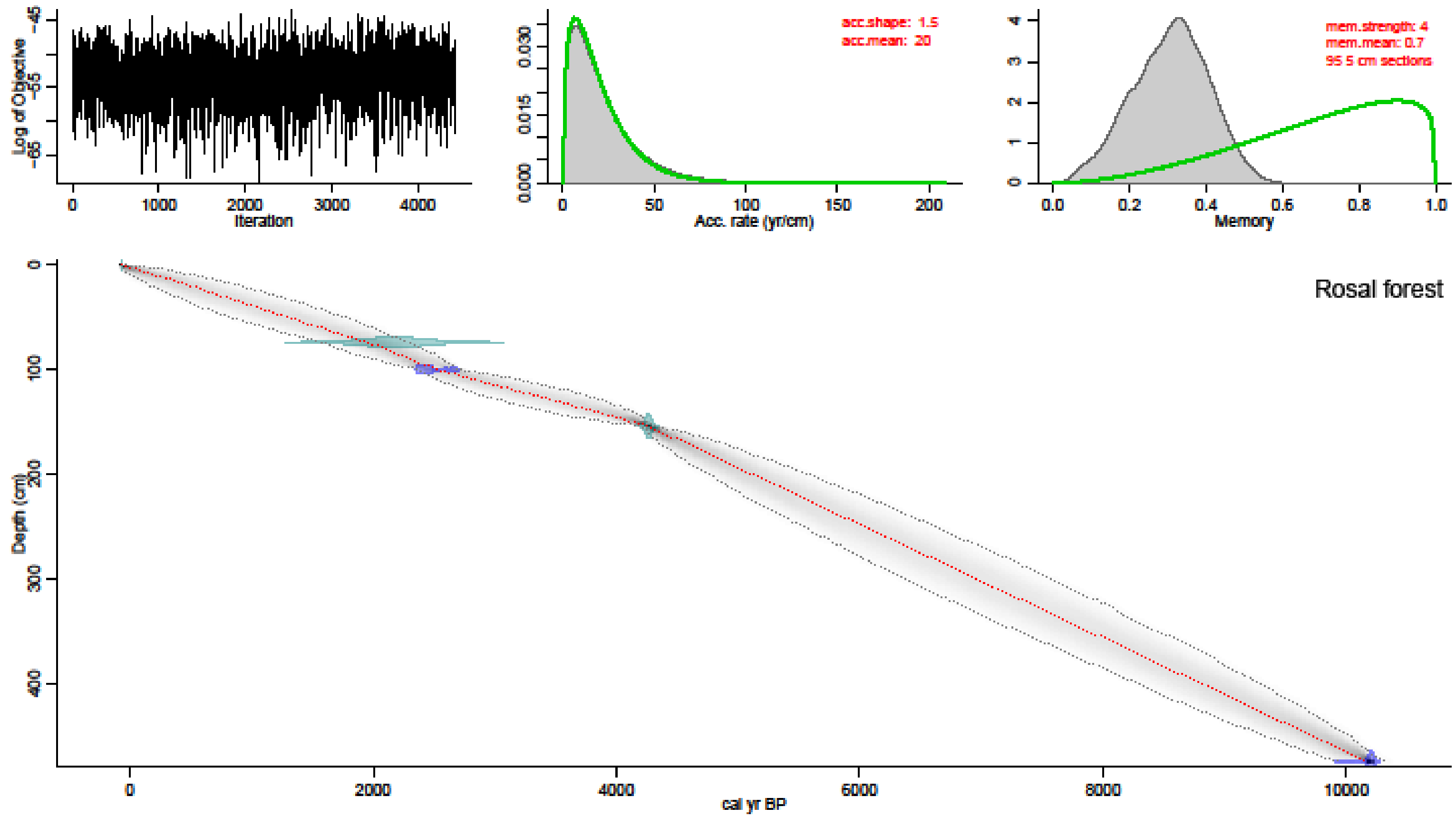
Supplementary Figure 32. Age depth model derived from radiocarbon dating and tephra deposits for Forsinard undrained bog.



Supplementary Figure 33. Age depth model derived from radiocarbon dating and tephra deposits for Forsinard afforested bog.



Supplementary Figure 34. Age depth model derived from radiocarbon dating and tephra deposits for Rosal undrained bog.



Supplementary Figure 35. Age depth model derived from radiocarbon dating and tephra deposits for Rosal afforested bog.