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# Potential for large losses of carbon from non-native conifer plantations on deep peat over decadal timescales

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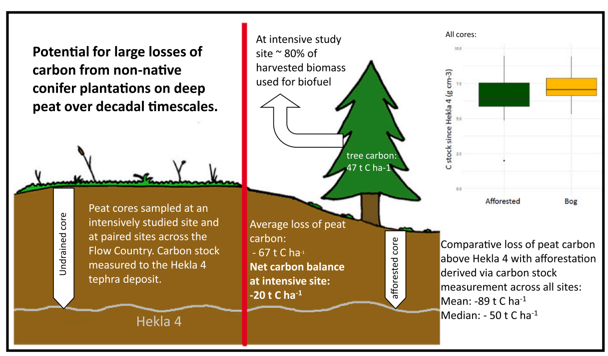
## HIGHLIGHTS

- In the 20th Century, large areas of deep peat were drained and afforested with conifers.
- This study measured carbon stocks in afforested and undrained peat.
- The Hekla 4 tephra deposit was used as a stratigraphic marker for assessing carbon accumulated.
- C Total carbon in afforested areas was on average 20 t C ha$^{-1}$ less than undrained peat after 60 years.
- This stock-based study is of relevance to the ongoing policy debate on deep peat restoration.

## GRAPHICAL ABSTRACT

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## ABSTRACT

Peatland drainage is a large source of anthropogenic $CO_2$ emissions. While conversion to agriculture is widely acknowledged to lead to “irrecoverable” carbon (C) losses, in contrast the C impacts of peatland forestry are poorly understood, especially in intensively managed plantations. Losses of C from peat oxidation are highly variable and can be compensated for by gains of C in trees, depending on the lifecycle of the timber and timescale

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Bulk density
C stocks
Drainage
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considered. Here, we used ITRAX scanning to enable rapid detection of the Hekla 4 cryptotephra layer as a reliable chronological marker above which peat properties and C stocks could be compared between open and afforested blanket bog cores in the Flow Country of Northern Scotland. At one site, Bad a' Cheò, we combine replicated core pair comparisons ($n = 19$) with timber extraction data to derive net ecosystem C balance over the lifetime of the plantation. Here the reduction in peat C carbon storage above Hekla 4 in afforested samples (67 t C ha$^{-1}$) is only partially compensated by tree C sequestration (47 t C ha$^{-1}$), leading to a net ecosystem C balance indicating a loss of 20 t C ha$^{-1}$ over the 50 years since the plantation was established. At that site, ~65 % of tree C rapidly returned to the atmosphere, as it was primarily used for heat and power generation. Across the wider Flow country region, a simplified paired sampling method was adopted at eight further sites, finding a either a loss or negligible change in peat C storage above Hekla 4 in afforested samples with a mean loss of 86 t C ha$^{-1}$ and median loss of 50 t C ha$^{-1}$. This study suggests that potentially substantial C losses have been an unintended consequence of non-native conifer afforestation over deep blanket bogs.

## 1. Introduction

The global soil carbon pool is significant but disproportionally distributed, with peatlands estimated to store between 400 and 600 Gt globally (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 organic matter, which is then preserved 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 (Houghton et al., 2012; Joosten, 2016; Page and 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) as a result of drainage (Leifeld et al., 2019). While peatland drainage for conversion to agriculture is acknowledged to lead to "irrecoverable" carbon losses (Goldstein et al., 2020), the carbon balance of commercial peatland afforestation is more difficult to constrain (Ojanen et al., 2013; Krüger et al., 2016; Sloan et al., 2018).

In the second half of the twentieth century, post-war timber shortages and 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 rapid and extensive afforestation of naturally treeless, deep peatlands that were otherwise not financially attractive for forestry and typically not planted commercially (Warren, 2000). By the late 1980s, 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 and Jessop, 2018) and in Ireland (Farrell, 1990; Renou and Farrell, 2005), respectively. In the UK, 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; Wilson et al., 2014). As a result, restoration initiatives driven by biodiversity and changes in policy (Forestry Commission Scotland, 2015, 2016) have already led to large scale removal of conifers on deep peat in areas adjacent to blanket bog habitats designated for conservation.

Now, with many of the initial plantations approaching 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. On one hand, it has been estimated that between 320 million ha and 1 trillion ha of additional forest cover, or an area roughly the size of Brazil, is required by most emissions scenarios in order to have a 50 % chance of meeting a 2° climate warming goal (Smith et al., 2016). As such, the emphasis on afforestation as a strategy for climate change mitigation (Bastin et al., 2019) is associated with political pledges at the national level towards large-scale tree planting and with policies supporting increased forest cover (e.g. UK Government, 2021a, 2021b). On the other hand, the increased recognition of the importance of peatlands to carbon storage and climate cooling (Dise and Phoenix, 2011; Harris et al., 2022) has put peatland protection and restoration high on the agenda and is supported by other sets of policies (e.g. NatureScot, 2015; UK Government, 2021a, 2021b). Guidance to inform decisions about the best strategies for UK carbon land management has been published for England (UK Government, 2023), but robust empirical data on the overall carbon impact of afforestation on peatlands are urgently needed.

Drainage and afforestation of peatland lead to gains and losses of carbon through a variety of processes (Swindles et al., 2019). The drainage and ploughing of previously open bogs required for the initial planting of a conifer plantation leads to compression and compaction of the peat, along with some oxidation and loss of carbon (Hargreaves et al., 2003), also causing ground level subsidence (Anderson, 2010) and ultimately, where the water table is drawn down sufficiently, peat cracking (Pyatt and John, 1989). As the water table lowers, often through the evaporative effect of the trees, more peat is exposed to oxidative loss of peat carbon through respiration (Eggelsmann, 1975), increasing emissions of $CO_2$, but reducing $CH_4$ emissions (Drosler et al., 2008; Hermans et al., 2019a). 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 warming 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 (Billett et al., 2007; Dinsmore et al., 2010; Williamson et al., 2021). The closing of the forest canopy progressively increases interception of rainwater and light and lowers the water table further (Cummins and Farrell, 2003), which in turn can lead to the loss of specialist peatland plant communities and a species poor understorey dominated by needle litter (Hancock et al., 2018).

The impact of draining boreal peatlands for forestry on the belowground (i.e. peat including fine roots) and ecosystem (combined above- and belowground) C balance has been the subject of many studies in Fennoscandia since the 1990s. These studies reveal a broad range of values from net gains to net losses in belowground C across mire types of varying nutrient status and under different climates (Minkkinen and Laine, 1998; Minkkinen, 1999, Turunen, 2008; Simola et al., 2012, Pitkanen et al., 2013; Krüger et al., 2016). As full forestry rotations are measured in decades, carbon accumulation in developing tree biomass and litter may be expected to counter the loss of peat carbon to the atmosphere. Indeed, 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 over multiple rotations (Lal, 2005; Clemmensen et al., 2013). In Fennoscandia, in some sites, post-drainage increases in wood, root and litter production in treed peatlands has been shown to compensate for C loss from the peat, suggesting that boreal peatlands drained for forestry could maintain a net positive ecosystem C balance over time (Minkkinen, 1999; Minkkinen et al., 2002). This finding was reinforced by further studies looking at contemporary GHG fluxes (Lohila et al., 2011, Ojanen et al., 2013: Tong, 2022) suggesting that drained forested peat soils were shown to be a net GHG sink. However, this may not be the case at all sites, and it may be that in some instances, the increased C emissions from degrading peat are not fully compensated for by tree growth, leading to net ecosystem C losses (Meyer et al., 2013; Krüger et al., 2016). Importantly, many of the peatlands reported in these studies were already naturally tree covered, with drainage used to boost timber yield (Laiho and Laine, 1997; Minkkinen et al., 2002; Laine et al., 2009). These factors, along with differences in the climates and geology of Fennoscandia and the British Isles, mean that such findings may not all be readily transferable to naturally open deep peatlands like those targeted in the UK context, such as the blanket bogs of the Flow Country (Sloan et al., 2018).

For example, these data have been used in modelling studies in the British Isles that have implied a net increase in carbon storage under afforestation due to tree growth, and that have identified the important influence of the ground preparation methods used and productivity of the forest on net C balance (Cannell, 1999; Vanguelova et al., 2018; Vanguelova et al., 2021). To date, it is still unclear what the net belowground and ecosystem C balances of forestry on deep peat are, where lower yields of wood could be expected, owing to wetter and more nutrient-poor soil conditions, particularly where drains have not been well-maintained (Oosthoek, 2013). Clearly, there is scope to fill this gap with empirically derived assessments of changes in C stocks over the lifetime of these deep peat plantations.

In this study, we used a stock-based approach to provide empirical measures of the changes in carbon associated with afforestation of deep peat with non-native conifers in northern Scotland. More specifically, the objectives of the study are to 1) use pairs of cores from undrained and adjacent afforested peatlands to measure differences in C stock in the peat (i.e. belowground C); 2) empirically measure biomass and carbon content from trees to derive a net ecosystem C balance within a single site; 3) contextualise the C balance with peat properties and peat C balance derived from core pairs collected in a range of open peatlands and afforested peatlands across the region. We hypothesized that we would measure compaction, subsidence and carbon losses from the peat from afforested peatlands, and that these losses would be partially, but not fully, compensated by tree growth. We also hypothesized that changes in carbon balance would vary regionally because of differences in local setting and management practices.

## 2. Material and methods

### 2.1. Study site and design

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 with non-native conifers between the 1960s and 1980s (Stroud et al., 1987; Lindsay et al., 1988). Eight afforested peat plantations were surveyed, along with an adjacent open blanket bog area (Fig. 1, Table 1). One site, Bad a' Cheò, was used as an intensively sampled site in which multiple cores were taken and changes in carbon storage were compared to carbon uptake in timber biomass. In the remaining seven sites a simplified paired sampling was carried out to put the results from Bad a' Cheò in a regional context.

The Bad a' Cheò site, a 50 ha area of a much larger blanket bog at approximately 90 m elevation above sea level was used for the more intensive, within-site comparison of peat and tree C stocks. The site was drained and planted in 1968, mostly as a cultivation and drainage experiment testing different combinations of ploughing type and cross-drainage intensity, 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 under forest stands by up to 0.5 m from interpolated 1966 pre-planting values, or 13 % of total depth (Sloan et al., 2019), presumably due to some combination of peat compaction and oxidation. In the Bad a' Cheò plantation, three transects had been established from prior studies (Pyatt et al., 1992; Sloan et al., 2019), running from the undrained, unplanted bog into the drained and planted blocks of forestry (Fig. 1b). Along each of the transects, three locations were identified: (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

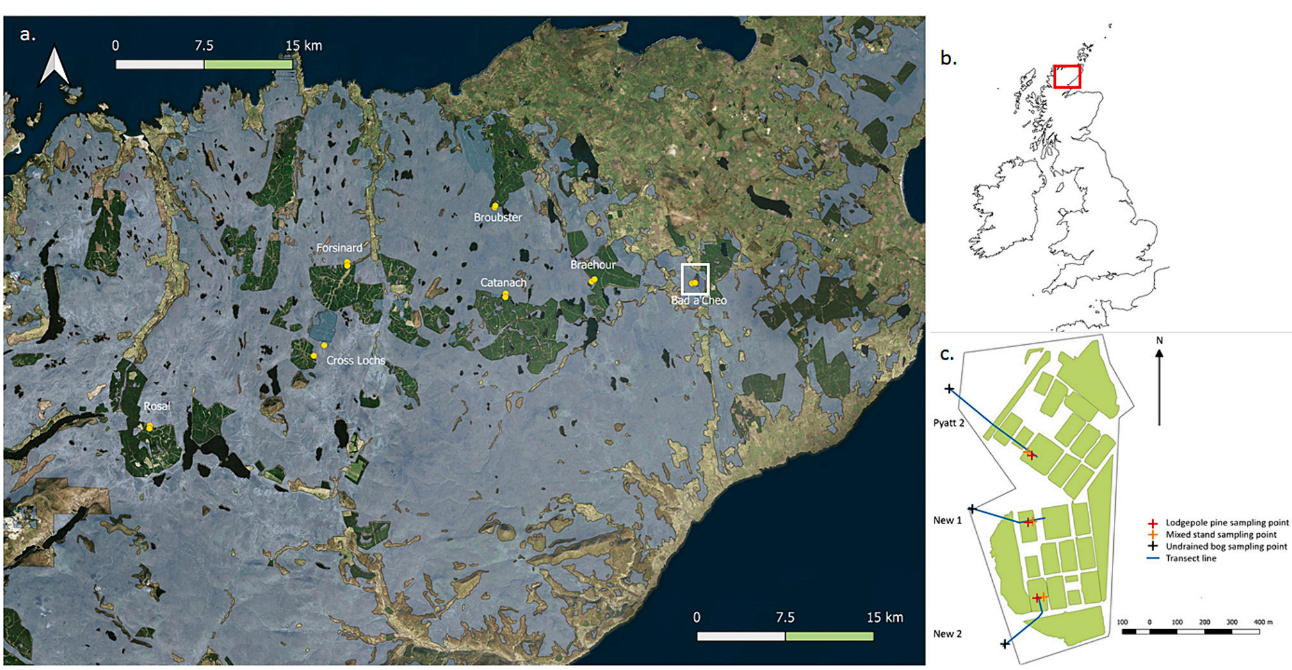


**Fig. 1.** a) Sampling locations across the Flow Country. Pale blue/grey indicates area of peatland (from PeatWind GIS Layer). B) Inset location of highlighted area within the UK. C) Inset site map of Bad a' Cheò with sampling points.

**Table 1**
Sites used in the study, across the Caithness and Sutherland counties of northern Scotland. For each site, the location, elevation above sea level, peat depth of both the undrained (bog) and drained (forestry) sites and details about the plantation type are provided. Values for peat depth at Bad a' Cheò are derived from the average of cores taken across the site.

| Site name (year sampled) | Location | Elevation (m.a.s.l) | Details |
|---|---|---|---|
| Bad a' Cheò (2014 & 2017) | 58°25′49.28″N 3°25′41.00″W | 90 | One of the oldest plantations in the Flow Country set up as an experimental site by Forestry commission in 1968, converted to a windfarm in 2017. This site was used for the first study and for the intensive replicated study. |
| Braehour (2014) | 58°25′27.31″N 3°34′24.37″W | 121 | Publicly owned forestry plantation planted in the early 1980s, with some areas targeted for restoration after 2015. **Site cored but lack of Hekla 4 deposit excluded it from analysis**. |
| Broubster (2017) | 58°28′43.98″N 3°43′32.28″W | 210 | Publicly owned, typical of the 1980s. |
| Catanach (2014) | 58°24′57.08″N 3°42′11.41″W | 185 | Private forestry, one of the most remote plantation locations in the Flow Country, 9 miles from the nearest paved road, typical of 1970s–1980s. |
| Cross Lochs (2014) | 58°22′14.43″N 3°57′58.27″W | 207 | The open bog is part of RSPB Forsinard Flows National Nature Reserve and is a well-studied area, including an eddy covariance flux tower. The plantation is privately owned, typical of the 1980s but with a double-ploughed ridge rather than single ploughed. |
| Dalchork (2017) | 58°10′8.94″ 4°30′43.68″ | 185 | Publicly owned. Part of the plantation has been restored back to open bog since 2015. Only planted with Lodgepole Pine in the 1980s, plus some pre-existing plots (not cored for this study) in the 1960s. |
| Forsinard (2017) | 58°25′59.41″N 3°56′2.54″W | 119 | The area includes the RSPB Forsinard Flows National Nature Reserve and the neighbouring Big House estate. The plantation, largely afforested in the 1980s, is now being restored back to open bog. |
| Rosal (2017) | 58°18′14.04″N 4°12′43.14″W | 333 | Publicly owned, planted in the 1970s with Lodgepole Pine. |

sampling locations, cores were taken from ridge, plough furrow and original surface. Thus, in total, 27 cores were taken for this within-site comparison – providing a uniquely large sample size and level of replication for this kind of study.

A coring campaign across the wider Flow Country was undertaken to provide additional regional context. Four additional sites (Broubster, Forsinard, Rosal and Dalchork) from around the Flow Country were also sampled using a simplified paired sampling design in which a single pair of cores were sampled, one within (from the original surface microtopography) and one outside (from the lawn microtopography) of an afforested bog. The same design had also been used previously at Bad a' Cheò and at three additional sites (Braehour, Catanach and Cross lochs) as part of a pilot study completed in 2014 examining tephra and C accumulation rates in the region (Ratclifffe, 2016; Ratcliffe et al., 2018). These sites covered a range of peat depths and elevations, plus a mixture of plantation ages and tree mixes and 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. All sites selected were in their first forestry rotation and had been planted in the second half of the 20th century.

In those sites, a sampling point was identified approximately midway across the forestry block (to avoid edge effect) along the forest margin matching the ownership boundary criteria mentioned above. A tape was run out at right angles to the forest boundary, into the plantation and into the undrained bog. At least 50 m either side of the forest boundary the nearest sampling point with the target microtopography, either unploughed original surface in the forestry plantation or lawn in the open bog was sampled. In all the sampling sites, a 0.5 m long, 5 cm diameter Russian corer (Aaby and Digerfeldt, 1986) was used to sample peat to the underlying bedrock. The cores included fine roots (<~5 mm), but not the larger woody roots. At each sampling point two cores were obtained within approximately 20 cm of each other, and 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 and Granlund, 1926).

### 2.2. Peat carbon content

Instead of using basal dates to obtain total accumulation (which will vary between locations), we measured C accumulation above a reliable stratigraphic marker known to be present in the Flow Country; a band of Hekla 4 cryptotephra. Cryptotephra are distal deposits of volcanic glass shards which form fine layers not visible to the naked eye and are abundant throughout Scottish peat (Dugmore et al., 1995; Lawson et al., 2012). Here, we used ITRAX scanning techniques, which combine radio chromatograph and XRF elemental analysis, to allow for the rapid identification of tephra layers (Croudace et al., 2006). Based on findings from the pilot study by Ratclifffe (2016), the top 320 cm of each core (or the total length of the core where peat depth was <320 cm) were XRF scanned and the ITRAX results compared to LOI data to indicate where deposits of inorganic material of interest were located. A prominent 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, 2019). This layer of Hekla 4 tephra, dated as 2310 ± 20 BCE (Pilcher et al., 1995), serves as the stratigraphic marker that forms the basis for like-for-like carbon comparisons. Unlike traditional microscopy-based approaches or reliance of C dating, the use of tephrochronology enabled us to process a high volume of cores, including all the replicated cores from Bad a' Cheò, which is rare in palaeoecological studies.

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 one instance where a layer of Hekla 4 tephra could not be identified (the Dalchork bog core). In the Braehour afforested core a layer of Hekla 4 could not be examined, nor was there enough radiocarbon data for an accurate age depth model to be generated. The pair of Braehor cores were therefore excluded from further analysis. 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. Samples for radiocarbon analysis were pre-treated using the acid-base-acid method, and vertical rootlets were picked out prior to radiocarbon analysis using bulk samples (using AMSDirect). Age depth modelling was completed with the BACON package in R (Blaauw and Christen, 2011), with IntCal20 used to calibrate the radiocarbon dates

(Reimer et al., 2020).

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 h) and weighed to calculate bulk density (Chambers et al., 2011). Loss on ignition (LOI) was determined by combusting the subsamples at 550 °C for 4 h 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 Soil C above Hekla 4 (CHek4). This uses a formula adapted from Mäkilä and Goslar (2008):

$$\mathrm{CHek4} = \sum \mathrm{inc} \times \mathrm{BD} \times \mathrm{C}$$

where 'CHek4' 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 as a proportion of dry mass. The sum of all the carbon in each 5 cm increment to the depth of the Hekla 4 tephra gives the final carbon accumulation figure. The effect of afforestation on carbon balance was derived from the difference between CHek4 from undrained bog and afforested bog and assumed that there was no difference in CHek4 prior to afforestation.

For this study, the comparison is done on the basis of changes in total stock of carbon rather than on apparent yearly rate of C accumulation. Indeed, a potential criticism of a whole column inventory approach for peat carbon stocks is that 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). In the case of afforested peat, the redistribution of peat by ploughing means that top 50 cm of the forestry sites may be dense former catotelm peat brought back to surface. Such disruption to the peat surface would make the removal of an arbitrary 50 cm unjustified.

### 2.3. Tree carbon

To assess tree biomass within the Bad a' Cheò forestry plantations we used two approaches. Firstly, a non-destructive survey of tree morphometrics combined with felling of sample trees to directly measure carbon content. Within Bad a' Cheò, 20 plots of 16 trees were established along the transects in blocks of 4 × 4 trees, with ten plots each of Lodgepole pine monocultures and mixed stands of Lodgepole pine and Sitka spruce. 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' Cheò, 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, cut into sections and brought back to the laboratory, alongside soil samples (for fine root, later excluded from the analysis – see Appendix) and root plates also cut into sections. From these samples from both species, we directly measured fresh biomass weight, dry biomass weight, and carbon content (see Appendix).

In 2017 the Bad a' Cheò plantation was felled when the site was taken over by a renewable energy company for a wind farm development. As part of this 21.43 ha were mechanically felled, with brash and stumps mulched, and the remaining areas 0.24 ha trees were hand felled. On the remaining 5.86 ha of afforested bog, the timber was not extracted but the woody debris were mulched on site, because of windthrow or misshapen and dead wood, or problems of accessing portions of the areas with machinery. Information on timber yield and destination and use of the wood was obtained from the logging company. Our directly measured carbon content was used to estimate the total mass of carbon in the biomass removed from the site and empirically derived measurements described above, and assuming that loss of moisture between felling and loading onto lorries, where the wood was weighed, was negligible. Combining this direct measure of biomass with our morphometric measurements and assessment of C in different unharvested parts of the trees, we were able to estimate the total C accumulation by the trees from plantation to felling.

### 2.4. Statistical analyses

All the statistical analyses were conducted in R version 4.3.1 (R Core Team, 2023). The data was analysed in two groups:

a) The multiple cores collected from Bad a' Cheò were analysed to give a detailed description of the changes at this single intensively studied site. Due to our low number of replicates, non-parametric analyses of variance (function *anova* on rank transformed data) were used to compare means BD and CHek4 values between site type (Bog or Afforested) and micro-topographic positions (High: hummock or ridge, Medium: lawn or original surface, Low: furrow or hollow) within the Bad a' Cheò site. Within the afforested section, we used Mann-Whitney test to compare the BD, CHek4 between the mixed and single species stands, and the surveyed DBH between the Lodgepole pine and Sitka spruce.
b) In the sets of paired cores taken across the wider Flow Country region (Table 1) we used a paired *t*-test to determine whether there was a significant difference in BD and CHek4 between Bog and Afforested sites overall. As well as using the individual core pairs, a single average value from bog and afforested cores from the Bad a' Cheò intensive study was used. For the purposes of this analysis the averaged value from Bad a' Cheò was given the same weight as an individual core pair to mitigate what would otherwise be a nested and imbalanced design.

## 3. Results

### 3.1. Peat carbon

Within the more intensively studied Bad a' Cheò plantation, the mean bulk density down to Hekla 4 was lower ($F = 20.05$, $p = 0.002$) in the undrained bog sites (0.064 ± 0.016 g $cm^{-3}$) than in the afforested sites (0.10 ± 0.03 g $cm^{-3}$), but there were no significant differences between microtopographic positions ($F = 1.27$, $p = 0.30$) (Fig. 2a, Table 2). Within the Bad a' Cheò plantation, the stand type did not affect

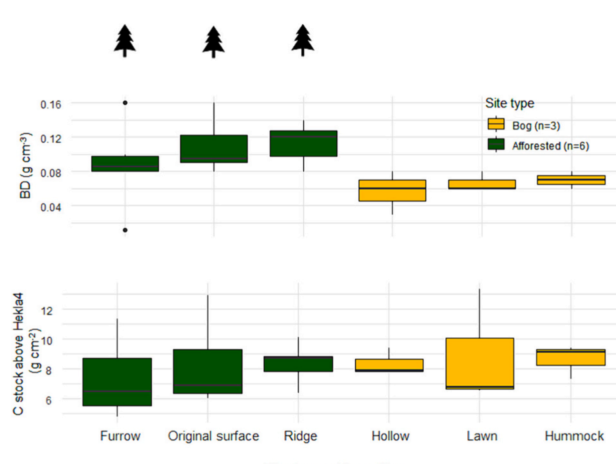


**Fig. 2.** a) Peat bulk density and b) Apparent carbon accumulation since Hekla 4 at the Bad a' Cheò plantation across different microtopographical positions. Significant differences are highlighted with an asterisk (*). Dots represent outlying data.

**Table 2**
Peat characteristics from all sampled cores. For sites, BadaCheo1 was sampled in 2014, while BadaCheo was sampled in 2017.For microtopography, Hu = hummock, Ho = hollow, L = lawn, F = furrow, OS = original surface, R = ridge. For trees, LP = Lodgepole pine, SS = Sitka spruce. CHek 4 is carbon stored above Hekla 4. Values for BD, % moisture, % LOI (loss on ignition), and % C are the averaged values for each core to Hekla 4.

| Site | Sample type | Transect (micro-topography) | Tree | Peat depth (cm) | Hekla4 depth (cm) | BD (g $cm^{-3}$) | Moisture content (%) | % LOI | % C | CHek4 (g $cm^{-2}$) |
|---|---|---|---|---|---|---|---|---|---|---|
| BadaCheo | Bog | New1 (Hu) | NA | 387 | 221 | 0.08 | 91.58 | 96.73 | 51.81 | 9.42 |
| BadaCheo | Bog | New1 (Ho) | NA | 485 | 183 | 0.08 | 91.95 | 95.81 | 49.57 | 7.33 |
| BadaCheo | Bog | New1 (L) | NA | 493 | 318 | 0.08 | 91.69 | 95.84 | 48.04 | 13.32 |
| BadaCheo | Bog | New2 (Hu) | NA | 536 | 234 | 0.06 | 93.34 | 95.39 | 50.97 | 7.8 |
| BadaCheo | Bog | New2 (Ho) | NA | 494 | 210 | 0.06 | 93.58 | 97.3 | 51.35 | 6.74 |
| BadaCheo | Bog | New2 (L) | NA | 478 | 304 | 0.06 | 94.23 | 97.79 | 51.01 | 9.13 |
| BadaCheo | Bog | Pyatt2 (Hu) | NA | 514 | 259 | 0.07 | 93.10 | 97.41 | 52.53 | 9.39 |
| BadaCheo | Bog | Pyatt2 (Ho) | NA | 525 | 255 | 0.06 | 94.06 | 97.26 | 51.94 | 7.87 |
| BadaCheo | Bog | Pyatt2 (L) | NA | 520 | 212 | 0.06 | 93.31 | 97.58 | 53.96 | 6.59 |
| BadaCheo | Forest | New1 (F) | LPSS | 388 | 133 | 0.09 | 90.59 | 97.24 | 52.23 | 6.47 |
| BadaCheo | Forest | New1 (OS) | LPSS | 410 | 144 | 0.09 | 91.09 | 97.76 | 52.65 | 6.83 |
| BadaCheo | Forest | New1 (R) | LPSS | 410 | 160 | 0.09 | 90.65 | 96.94 | 52.04 | 7.53 |
| BadaCheo | Forest | New1 (F) | LP | 105 | 139 | 0.08 | 91.85 | 97.5 | 53.07 | 6.06 |
| BadaCheo | Forest | New1 (OS) | LP | 410 | 143 | 0.08 | 91.56 | 97.43 | 52.35 | 6.5 |
| BadaCheo | Forest | New1 (R) | LP | 405 | 150 | 0.08 | 91.79 | 97.31 | 51.58 | 6.4 |
| BadaCheo | Forest | New2 (F) | LPSS | 262 | 121 | 0.14 | 84.88 | 96.87 | 52.16 | 8.65 |
| BadaCheo | Forest | New2 (OS) | LPSS | 260 | 131 | 0.16 | 83.78 | 96.77 | 52.27 | 11.34 |
| BadaCheo | Forest | New2 (R) | LPSS | 284 | 149 | 0.16 | 81.09 | 96.23 | 52.22 | 12.92 |
| BadaCheo | Forest | New2 (F) | LP | 292 | 134 | 0.12 | 87.13 | 97.01 | 52 | 8.79 |
| BadaCheo | Forest | New2 (OS) | LP | 298 | 146 | 0.12 | 87.50 | 96.79 | 53.7 | 9.46 |
| BadaCheo | Forest | New2 (R) | LP | 310 | 148 | 0.13 | 85.93 | 97.67 | 51.34 | 10.07 |
| BadaCheo | Forest | Pyatt2 (F) | LPSS | 386 | 92 | 0.09 | 90.30 | 97.66 | 50.48 | 4.85 |
| BadaCheo | Forest | Pyatt2 (OS) | LPSS | 410 | 126 | 0.09 | 89.31 | 97.44 | 54.14 | 6.2 |
| BadaCheo | Forest | Pyatt2 (R) | LPSS | 435 | 151 | 0.13 | 86.53 | 95.11 | 51.62 | 10.11 |
| BadaCheo | Forest | Pyatt2 (F) | LP | 370 | 98 | 0.1 | 89.95 | 96.95 | 51.97 | 5.22 |
| BadaCheo | Forest | Pyatt2 (OS) | LP | 365 | 122 | 0.1 | 87.84 | 96.43 | 51.93 | 6.95 |
| BadaCheo | Forest | Pyatt2 (R) | LP | 406 | 133 | 0.12 | 86.74 | 92.85 | 51.59 | 8.82 |
| BadaCheo1 | Bog | NA (L) | NA | 436 | 200 | 0.07 | 93.71 | 97.78 | 52.6 | 6.27 |
| BadaCheo1 | Forest | NA (M) | LPSS | 375 | 114 | 0.09 | 88.38 | 97.62 | 51.9 | 6.32 |
| Braehour | Bog | NA (L) | NA | 146 | 102 | 0.12 | 88.11 | 95.95 | 50.3 | NA |
| Braehour | Forest | NA (OS) | LPSS | 141 | NA | 0.12 | NA | NA | NA | NA |
| Broubster | Bog | NA (L) | NA | 276 | 136 | 0.09 | 91.58 | 97.53 | 51.76 | 6.23 |
| Broubster | Forest | NA (OS) | LPSS | 71 | 21 | 0.18 | 83.12 | 95.16 | 48.88 | 1.99 |
| Catanach | Bog | NA (L) | NA | 166 | 117 | 0.1 | 89.92 | 97.53 | 56.8 | 6.76 |
| Catanach | Forest | NA (OS) | LPSS | 100 | 97.4 | 0.11 | 88.63 | 97.84 | 50.5 | 5.48 |
| CrossLochs | Bog | NA (L) | NA | 358 | 166 | 0.08 | 93.77 | 97.92 | 51.8 | 6.72 |
| CrossLochs | Forest | NA (OS) | LPSS | 255 | 123 | 0.09 | 93.13 | 97.71 | 54.3 | 5.99 |
| Dalchork | Bog | NA (L) | NA | 260 | 136 | 0.07 | 92.64 | 98.26 | 53.89 | 5.3 |
| Dalchork | Forest | NA (OS) | LP | 151 | 98 | 0.1 | 90.25 | 92.63 | 51.13 | 5.18 |
| Forsinard | Bog | NA (L) | NA | 267 | 189 | 0.1 | 90.07 | 96.86 | 53.53 | 10.91 |
| Forsinard | Forest | NA (OS) | LPSS | 271 | 171 | 0.12 | 87.75 | 97.26 | 53.94 | 11.1 |
| Rosal | Bog | NA (L) | NA | 333 | 187 | 0.08 | 92.14 | 97.49 | 52.97 | 7.85 |
| Rosal | Forest | NA (OS) | LP | 474 | 154 | 0.09 | 91.14 | 97.16 | 53.13 | 7.53 |

the bulk density significantly (W = 50, $p = 0.42$), with mean BD under Lodgepole Pine/Sitka Spruce mix (0.09 ± 0.03 g $cm^{-3}$) similar to that under Lodgepole Pine as a single species (0.11 ± 0.03 g $cm^{-3}$). Accumulation of carbon stock since Hekla 4 (CHek4) at the Bad a' Cheò site was greater in undrained bog (8.62 ± 2.06 g $cm^{-2}$) than in afforested sites (7.95 ± 2.20 g $cm^{-2}$), but this difference was not found to be significant (F = 0.84, $p = 0.370$). This lack of significance is partly attributable to the large within-site variation (Fig. 2b)which was especially pronounced in the"New 2"transect which, contrary to the other two transects, showed a a higher C stock above Hekla 4 under forestry compared with the undrained bog (Fig. 3a).

Across the three transect groups at Bad a' Cheò, the difference in CHek4 between the afforested and undrained plots is a net loss of 0.93 ± 1.35 g C $cm^{-2}$ at Pyatt 2, a net loss of 3.39 ± 1.19 g C $cm^{-2}$ at New 1, and a net gain of 2.32 ± 1.09 g C $cm^{-2}$ at New 2. Overall, at the intensively studied Bad a' Cheò plantation, afforestation led to a net loss of 0.67 ± 2.0.86 g C $cm^{-2}$.

The same overall trends observed in Bad a' Cheò apply when looking at the regional variation in the paired cores (including the above results from Bad a' Cheò 2017 as an additional value, carrying the same weight as the other pairs), with peat under afforested sites significantly denser (0.11 ± 0.03 g $cm^{-3}$) than in undrained bogs (0.08 ± 0.02; $t = -3.65$, $p = 0.0007$) and with values of CHek4 lower on average in afforested sites than in undrained bogs, although not significantly so ($t = 1.75$, $p = 0.123$) (Fig. 3b and c, Table 2). However, there is again a large variation in values. Three sites (Bad a' Cheò 2017, Broubster, Catanach) indicated a net loss of C, and three core pairs (Dalchork, Bad a' Cheò 2014, Forsinard) indicated a near neutral effect (Fig. 4). Across all paired samples the average reduction in carbon stock in the wider study region of 0.86 ± 1.44 g C $cm^{-2}$ (median of 0.5 g C $cm^{-2}$ reduction in carbon stock) in the peat accumulated since Hekla 4 in afforested sites compared to their undrained equivalent (Table 3).

### 3.2. Tree carbon

From allometric measurements, we found no significant variation in average DBH of Lodgepole pine trees across the three transects ($n = 241$, Chi square = 1.390, $p = 0.499$), with values of 19.6 cm on Pyatt 2, 19.3 cm on New 1 and 19.6 cm on New 2. On the other hand, Sitka spruce size did vary significantly across the site ($n = 80$, F = 8.564, $p \leq 0.001$) with the average of Pyatt 2 (16.4 cm) and New 1 (12.6 cm) significantly smaller than the average DBH of New 2 (24.6 cm).

The destructive sampling of tree biomass showed that on average dry tree biomass is 56 % (Lodgepole pine) and 50 % (Sitka spruce) of fresh

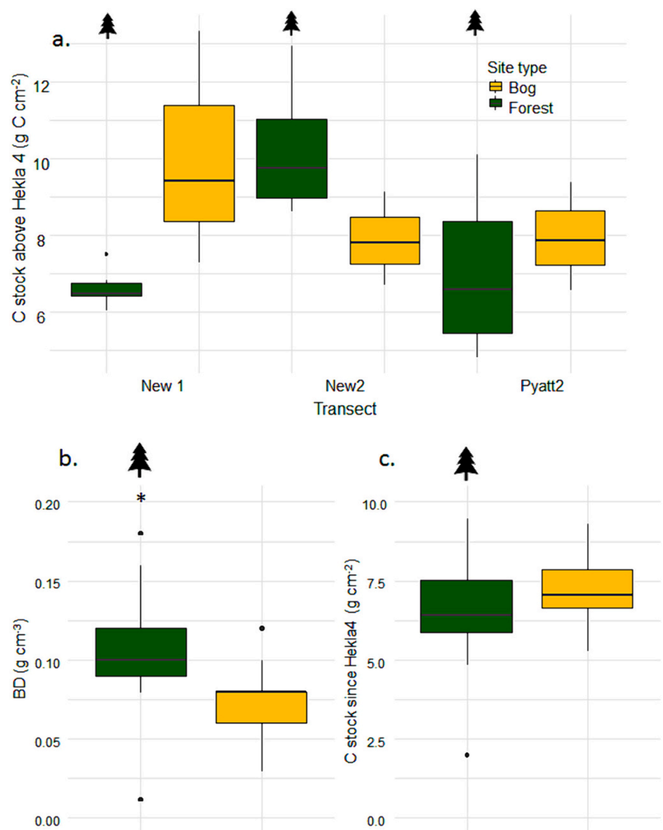


**Fig. 3.** a) Mean and distribution of C stocks above Hekla 4 from the Bog ($n = 3$) and Forest ($n = 6$) cores at the three transects from the Bad a' Cheò site. b) Mean and distribution of bulk density and c) Mean and distribution of C stock above Hekla 4 from all the Forest and Bog cores taken across all the sites ($n = 8$ pairs), with significant differences highlighted by asterisks ($^{**}p < 0.01$).

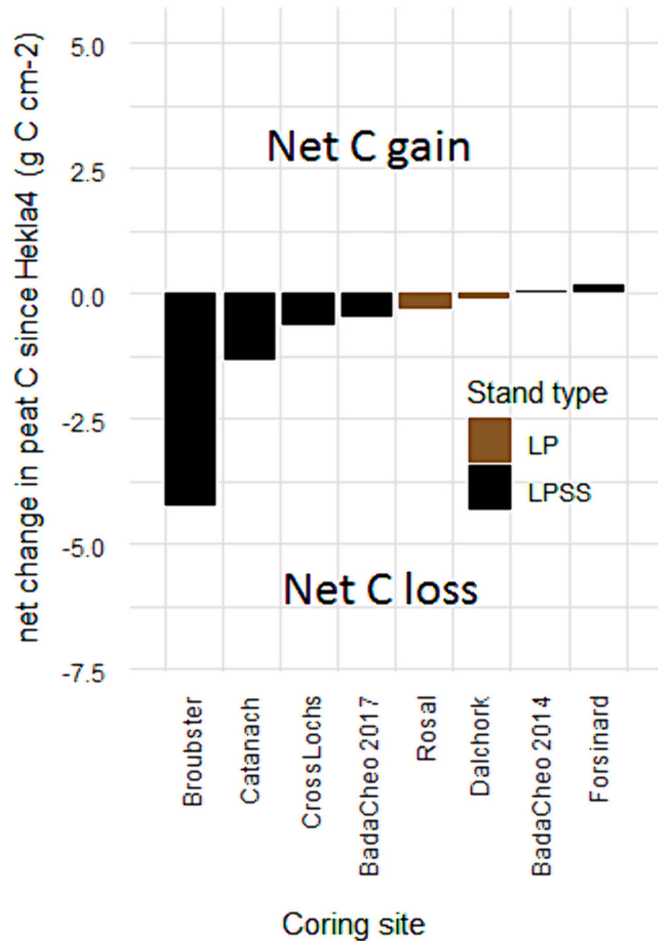


**Fig. 4.** Net change in peat C stock since Helka4 based on the difference between each pair of forest and bog cores. The sites identified as "New1", "New2" and "Pyatt2" were all sampled within the Bad a' Cheò plantation in 2017, while the site identified as Bad a' Cheò1 was sampled in 2014, in the same plantation. LP stand type is Lodgepole pine, while LPSS is a mixed stand of Lodgepole pine and Sitka spruce.

biomass (Appendix Table 1). 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 %.

From the felled 21.67 ha, timber weighing 2946 t was extracted, which is a little below what would be expected for a yield class 8 over the ~50 year period (yield class is the Forestry Commission index for tree productivity in timber biomass, used in the UK ans measured in cubic meters per hectare per year; Matthews et al., 2016). Of this, 2363.16 t were used for wood fuel, power generation and wood pellets. The remaining 583.28 t were split between fuel use and the manufacture of boards. Using the allometrically derived ratios of carbon to fresh biomass and assuming that moisture loss between felling and weighing was negligible, this extracted fresh weight is the equivalent of 681 t of carbon (t C). When adding the other (non-harvestable) components of the tree this equates to 1024 t C, or 47.1 t C ha, with woody roots left in the peat accounting for 173 t C and brash or portions of stem not removed making up the remaining 170 t C. Within the 5.86 ha of plantation where the trees were not harvested but mulched on site, the amount of wood biomass unharvested in this way is unknown, but assuming a similar productivity of ~47 t C ha$^{-1}$, this could represent a further 275 t C.

**Table 3**
Values of the change in peat C stock calculated for each site based on the difference between C stocks above Hekla 4 under Forest and Bog cores. For Bad a' Cheò 2017, the value represents an average of the three transects (±standard deviation). This result from Bad a' Cheò has been weighted equally to the paired samples in calculating the average.

| Site | Change in peat C stock above Hekla 4 (g C cm$^{-2}$) |
|---|---|
| Bad a' Cheò 2017 | −0.67 ± 2.86 |
| Bad a' Cheò 2014 | 0.05 |
| Braehour | Hekla 4 not found |
| Broubster | −4.24 |
| Catanach | −1.28 |
| Cross Lochs | −0.73 |
| Dalchork | −0.12 |
| Forsinard | 0.19 |
| Rosal | −0.32 |
| Mean | −0.86 |
| SD | 1.44 |
| Median | −0.50 |

### 3.3. Total carbon budget

At Bad a' Cheò, the average difference of carbon storage of −0.67 g C cm$^{-2}$ in the peat under afforested bog is equivalent to a loss of 67 t C ha$^{-1}$, which, when applied to the 21.67 ha of Bad a' Cheò where wood products were extracted, equates to losses of 1450 t C over the ~50 years between plantation and harvest. This loss is only partially compensated by the 1024 t C that was estimated to have been stored in tree biomass, leading to a potential net loss of 426 t C for that section of the plantation, or the equivalent of 20 t C ha$^{-1}$.

## 4. Discussion

### 4.1. Peat carbon changes following drainage and afforestation

In this study, we assessed the impacts of afforestation on peat C stock using a C-stock comparison between pairs of cores, which is considered to be a robust approach in terms of estimating differences in C accumulation or loss among sites (Krüger et al., 2016). As was hypothesized, we find that drainage and afforestation lead to compaction of the peat, measured as an increase in the bulk density of the peat and generally shallower peat, and smaller average depth to Hekla 4 in afforested sites. This is in line with studies measuring subsidence (Sloan et al., 2019) and changes in peat properties associated with afforestation (Minkkinen, 1999; Krüger et al., 2016; Ratcliffe et al., 2018; Sloan, 2019).

We find both net gains and losses of carbon accumulation since Hekla 4 (CHek4). Despite high variation, our results still point to average C losses from the peat following drainage and afforestation of 67 t C ha$^{-1}$ at Bad a' Cheò, with a median of 50 t C ha$^{-1}$ in the wider region. This figure is comparable with Finnish studies that reported losses over fifty years equivalent to 75 t C ha$^{-1}$ (Simola et al., 2012), 65.5 t C ha$^{-1}$ (Pitkanen et al., 2013) and 42–135 t C ha$^{-1}$ (Krüger et al., 2016).

In our study, the values presented for CHek4 and the resulting changes in C stocks from the peat are highly variable. This variation would not have been observed without the high level of replication in our study, made possible by the use of ITRAX to detect cryptotephra as a reliable stratigraphic marker (Balascio et al., 2015) and which enabled the rapid processing of ~125 m of peat from 43 cores. 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., 2011). In this case, the Hekla 4 tephra was the best choice, as it is widely distributed throughout the north of Scotland, relatively abundant (thick) for a cryptotephra, and deep enough in the profile to capture the main impacts of drainage and afforestation. However, estimating losses above Hekla 4 has the disadvantage compared to a younger stratigraphic marker, of incorporating several thousand years of paleoclimatically induced spatial variability. Indeed, different locations respond in contrasting ways to climate (e.g. Eppinga et al., 2010; Ratcliffe et al., 2018), which effectively equate to random noise in this method. A trade off also exists against older stratigraphic markers: by focusing on Hekla 4 (median depth bog: 200 cm; forest: 133 cm) it may also mean that some effects of the forestry on the deeper peat may have been missed, although these are likely to be small. Given the typical depth of Hekla 4 peat below the marker is not likely to have been directly aerated or disturbed, it has been compacted and likely subject to changes in temperature related to shading, snow cover and seasonal differences in surface moisture content and thermal conductivity. It is difficult to say whether these changes would be negative or positive for the C balance, but they are likely to be extremely small (Krüger et al., 2016).

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 (Barber et al., 2008; Pilcher et al., 1996; Watson et al., 2016). The more recent Glen Garry tephra, while frequently reported is more sporadically distributed and difficult to identify than Hekla 4. Lairg A is less frequently reported (it may predate 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 (Dugmore et al., 1995). 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 a tephra layer to fit the criteria of the study is a key disadvantage of our approach, presenting the possibility of a sampling design capturing more or less of the peat column than is required.

### 4.2. Sources of variation and error in belowground C assessment

The range of values observed in our study is in part attributable to the spatial variability in properties such as peat depth, bulk density and C content inherent in peatlands and has also been reported in studies of Finnish afforested peatlands (Simola et al., 2012), and un-drained high-latitude peatlands in Canada (Robinson and Moore, 1999) and Finland (Korhola et al., 1995). Peatland carbon accumulation can be highly sensitive to localised hydrological conditions, in turn impacted by geology and topography (Klein et al., 2013). This is well illustrated by Korhola et al. (1995) who found a more than three-fold variation in C accumulation rates across a single raised bog in Finland, with the highest and lowest rates occurring between sites <50 m apart. In peatlands with visible tephra layers, such variability in tephra depth can be seen even more clearly (Shearer, 1997). While such natural variability is itself a subject of study and debate, it is very probably linked to the underlying topography, which is notoriously variable in the blanket bogs which we are studying. We designed our study to minimise the potential variation between afforested and undrained sites e.g., by using legal boundaries representing landowners preferences for planting or not planting, instead of physical landscape boundaries where the underlying peat may have different characteristics, as well as by targeting equivalent micro-topographical positions in the regional study. Yet, it is possible that the paired coring sampling strategy may not have necessarily targeted peat with comparable Holocene carbon accumulation rates, despite their geographical proximity (Ratcliffe et al., 2018). For example, at Forsinard where we observed a small net gain of C under forestry, there was only a 4 cm difference in the total peat depths of the paired cores, which is much less than would have been expected from the observed compression and compaction. However, we believe that there were no practical or cost-effective alternative approaches to produce empirical data, and that our high level of replication which included between-site variation as well as within-site variation helps balance this potential source of error. It is also worth noting that the variability we have found here, is consistent with variability in the broader literature which consists of studies made up of multiple different methodological approaches. It is clear some peatland forests accumulate carbon (Lohila et al., 2011; Minkkinen et al., 2018; Minkkinen, 1999; Minkkinen and Laine, 1998), while others lose it (Minkkinen and Laine, 1998; Hargreaves et al., 2003; Lohila et al., 2007; Meyer et al., 2013; He et al., 2016; Korkiakoski, 2020).

One of the largest negative differences in carbon stocks between undrained and afforested peat was recorded in the Broubster core. In Broubster, 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, or that some of the peat has been lost. Domestic peat cutting for fuel plantation was a common practice across northern Scotland that could explain this discrepancy, but there are no visible signs of historical peat cutting in and around the Broubster plantation. The younger basal date of the afforested section relative to the undrained (3943 Cal BCE and 6056 Cal BCE respectively; Sloan, 2019) suggests a later initiation, or an unknown disruption of the peat post-initiation. It may be that the preparation of the afforested site could have caused an unusually large loss of peat through larger than usual mechanical disruption during the ploughing process, or that underlying hidden topography has led to different, and possibly not comparable peat stocks as discussed above.

Our highly replicated Bad a' Cheò site provides more insight in how the assessment of C stocks is affected by the disruption caused by ploughing, where dug out peat creating the furrows is redistributed to ridges, often as high as 50 cm above the original surface (Sloan et al., 2018). For instance, the relocation of peat creates a longer peat column in the ridges, leading to variable depth to Hekla 4 (Sloan, 2019), and thus potentially influences stocks of C measured using this chronological marker. On the other hand, peat on the ridges may be more likely to be oxidised as it is further away from the water table. The spatial heterogeneity of the balance between these processes is well exemplified by the fact that ridges within Bad a' Cheò were found to have both the highest net gains and the biggest net losses of C of all the core pairs in the study. And a third source of spatial variation, tree root distribution, may have contributed to the observed spatial heterogeneity in C stock. The cores from New 2 in Bad a' Cheò, unlike most of the other core pairs, suggested a net C gain under the forestry. The afforested samples on this transect were somewhat atypical of the Flow Country, characterised by dry, well drained peat, which was shallower than at the other afforested areas of Bad a' Cheò and had a significantly higher bulk density, and significantly larger Sitka spruce trees than elsewhere. It may be that an increase in the primary production of the bog vegetation post-drainage, resulting in enhanced carbon sequestration rates initially, compensating any losses from decomposition (Krüger et al., 2016). It is also possible for "new C" to be added in the soil, when increased litter input by trees is mixed with the mosses at nutrient-poor sites, building a secondary (raw) humus layer on top of the peat (Minkkinen et al., 2008). This increased input could have enabled some additional C storage in the peat at New 2 and other sites. Indeed, such a phenomenon where forestry soils are a contemporary net sink for GHG has been measured by Hermans et al. (2022) at the Forsinard site, as part of another study.

### 4.3. Carbon budget of forestry on deep peat and potential sources of error

This study presents the first empirically derived near-complete carbon budget for one drained afforested peatland site during the first forestry rotation, which implies a degree of compensation in tree biomass for carbon loss from peat, but an overall average net reduction of carbon stock of 20 t C ha$^{-1}$ relative to what would be expected if the peat had remained undrained. We readily acknowledge that this number comes with uncertainties, including about the volumes of brash (branches and tops) and roots left on site post extraction and the impact of roadside storage on moisture (and therefore on C-to-fresh weight ratios).

For the 5.7 million ha of afforested peat in Finland, Minkkinen et al. (2002) estimated that 50 Mt C has been gained over a century (a figure in line with much of the twentieth century literature) while Turunen (2008) calculated 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' Cheò, the former study would predict a gain of 4.4 t C ha$^{-1}$, while Turunen would predict a loss of 12.8 t C ha$^{-1}$. The differences between the observed results at Bad a' Cheò and what could have been expected from the previous studies mentioned above are attributable to many factors which suggest that Fennoscandian data are not directly applicable to the UK. Among these factors are the different climatic conditions, geology, and the abundance of minerotrophic fens with different hydrology and plant assemblages than blanket bogs in the British Isles. Such environmental and physical factors created conditions that in part contributed to differences in how the sites were prepared and under what incentives: In Fennoscandia, forestry is a more well-established practice on fen peats, which may already be naturally wooded (Laiho and Laine, 1997). While there are some plantations in open sites in the naturally afforested areas, relatively little work is required to bring these areas into commercial production (Minkkinen et al., 2002; Maljanen et al., 2010), and producing high quality hard wood products (Minkkinen et al., 2002; Drosler et al., 2008; Ojanen et al., 2013). On the other hand, within the UK, the tax incentives for afforestation meant that some of the land selected was extremely marginal, such as the deep peat blanket bogs studied here, and as a result required much more intensive preparation.

Acknowledging a high level of within-site and regional variation, our study conservatively suggests median losses of 50 t C ha$^{-1}$ from the peat following drainage and afforestation. Put it another way, over a 50 year period, to sequester enough carbon to be C neutral on average, conifer plantations on deep peat would have had to reach at least a yield class of 8–10, which many of the forestry plantations on deep peat in the far north of Scotland have not, particularly the earlier ones. Over a similar period, assuming a similar productivity to that of the Bad a' Cheò plantation and therefore a similar net loss of 20 t C ha$^{-1}$, the 67,000 ha of afforested peatlands of the Flow Country could have lost 1.3 Mt C.

There are reasons to expect that net carbon loss from the Bad a' Cheò, and losses from deep peat plantations in general, could have been larger than measured here – and that this estimate is somewhat conservative. For instance, this study does not consider areas which were outside the forest blocks but were still affected by the forestry drainage system. Areas peripheral to planting are known to have a significantly reduced peat depth (Sloan et al., 2019) and the effects of drainage may extend between 40 and 100 m from the forest blocks (Shotbolt et al., 1998; Lindsay, 2010). In these areas, a lowered water table may have led to small, but persistent, losses of carbon from oxidation and increased $CO_2$ emissions; however, this was never measured and cannot be implied from compaction alone. Further, a significant volume of woody material was mulched and left to decay or be preserved through integration in "new" peat formation, as the site was transitioned to a wind farm. The fate of the carbon in this remaining biomass and in the root biomass is dependent on the future drainage status of the peat. 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., 2019a). Part of the site was re-wetted and reprofiled, burying woody debris in the furrows as the plough throws are flattened, which 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 farms, which includes some form of forest-to-bog restoration interventions, but also the creation of tracks for which some drainage has to be maintained. A pulse of GHG production in the immediate aftermath of felling has also been documented, as tree photosynthesis is removed, and $CO_2$ is produced by decomposition of tree residues (Korkiakoski et al., 2019). Recent studies have also shown that forest-to-bog restoration can return to a net sink for C within 10–15 years or faster, but that sites under restoration can be net C sources for some years following tree removal, with brash contributing to emissions (Hambley et al., 2018; Hermans et al., 2019b) if not taken away from sites. Clear felling is also known to be associated with increase exports of dissolved organic carbon (DOC) (Nieminen et al., 2015). In other words, there may be an additional "legacy" loss associated with the decomposition of the remaining woody debris and losses of carbon through aquatic pathways after the forestry has been removed.

The fate of the extracted timber itself is another distinction between some Fennoscandian forestry and deep peat plantations in the UK. High quality wood products ensure that carbon remains in tree biomass for long periods (although a proportion of the biomass of this wood will be lost during the processing phase), as opposed to lower quality wood products intended for fuel use. For instance, the majority of the Bad a' Cheò biomass, and much of the biomass from the other Flow Country plantations undergoing forest-to-bog restoration, is destined for uses which will return carbon to the atmosphere rapidly, such as fuel wood and power generation (Darrell Stevens, RSPB Site Manager, personal communication). For carbon accounting purposes this loss would not be attributable to the plantation itself, but rather is "embedded" in the emissions from the industries that use the timber (Agostini et al., 2014). But while fuel from conifer biomass is likely to replace power generation from fossil fuels, once the full carbon balance of the forestry plantation on deep peat is considered, the greenhouse gas savings from substitution are rather less (Agostini et al., 2014; Fehrenbach et al., 2022).

### 4.4. Implications for peat restoration and forestry

Carbon positive drained peatland forests are reported internationally (Lohila et al., 2011; Minkkinen et al., 2018, Minkkinen, 1999;

Minkkinen and Laine, 1998; Turetsky et al., 2011; Tong, 2022), suggesting that it is possible in some instances to retain trees without the negative, unintended consequences of intensive commercial forestry on treeless peatlands (Payne and Jessop, 2018). Sadly, we are a long way from being able to reliably determine which peatland forests are likely to lose or gain carbon over their lifetimes. Our study measured the potential C impacts of a somewhat specific subset of forestry practice, the case of drainage and afforestation over deep treeless blanket peatland. With changes of policy, this practice is no longer allowed. For decades now, the forestry sector, alongside NGOs, renewable energy sector and private land managers have engaged in large-scale forest-to-bog restoration of many of these plantations, including some of our study sites (Catanach, Bad a' Cheò, Forsinard, Braehour, Dalchork, Rosal) in the years after our measurements were taken. It is acknowledged that in some of the deep peat plantations, the process of drainage may have led to the development of peat cracks which necessitate the more energy-intensive and costly backfill trenching rewetting technique (Pyatt and John, 1989; Artz et al., 2018). In such situations, it may not be practical or desirable to fully restore peat back towards bog and alternatives will be needed. Where second rotation of conifers or mixed native woodland plantations go ahead, then we suggest that it will be essential to document the greenhouse gas balance as the trees grow. Importantly, to develop resilient landscapes in the future, the role that peat has had in absorbing carbon over millennia must be recognised, and management should be implemented in such a way that will not result in unintended consequences, such as irrecoverable losses of carbon and enhanced GHG emissions especially coupled with declines in biodiversity. But just as critically, where trees have been lost in the landscape, policies should support their return and facilitate practices that will enable both peatlands and forests to contribute to a better future by cooling the climate and enhancing biodiversity.

## 5. Conclusion

This study has shown differences in responses to drainage and afforestation with conifers across blanket bogs in the Flow Country of Scotland ranging from a large loss of carbon stock, at some coring locations, little change or apparent accumulation. At one intensively studied site, the fifty-year-old Bad a' Cheò plantation, the changes in carbon stock varied between losses across two transects and a gain in carbon stock on a further transect. When the carbon accumulated in tree biomass is considered (although this carbon is likely to return to the atmosphere rapidly) Bad a' Cheò suggests a total net ecosystem loss of stored carbon of 20 t C ha$^{-1}$ over the initial forestry rotation. Restoration of many afforested bogs is underway, and there is a key need for future research into the extent to which methods of restoration are able to mitigate future loss of carbon from the peat store. Stock-based approaches, as used in this study, have an important and complimentary role alongside flux-based methods to investigate the impacts of such management changes in peatlands.

## CRediT authorship contribution statement

**Thomas J. Sloan:** Writing – review & editing, Writing – original draft, Visualization, Project administration, Methodology, Investigation, Formal analysis, Conceptualization. **Joshua Ratcliffe:** Writing – review & editing, Writing – original draft, Methodology, Investigation, Conceptualization. **Russell Anderson:** Writing – review & editing, Methodology, Investigation, Funding acquisition, Conceptualization. **W. Roland Gehrels:** Writing – review & editing, Supervision. **Peter Gilbert:** Writing – review & editing, Investigation. **Dmitri Mauquoy:** Writing – review & editing, Funding acquisition, Conceptualization. **Anthony J. Newton:** Writing – review & editing, Methodology, Funding acquisition, Conceptualization. **Richard J. Payne:** Writing – review & editing, Supervision, Methodology, Investigation, Funding acquisition, Conceptualization. **Justyna Serafin:** Writing – review & editing, Investigation. **Roxane Andersen:** Writing – review & editing, Writing – original draft, Validation, Supervision, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Data availability

Data will be made available on request.

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## Appendix A. Tree carbon measurements

At the Bad a' Cheò sites, the manually felled trees were subdivided into ten sections, five above (subsections F – J; Appendix Table 1) and five below (Subsections A – E) the crown with a chainsaw. 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 and 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 corresponding 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. We find that the proportion of C varies from the base to the top: in the lowest section of stem, which comprises the oldest wood, dry tree biomass is 62 % (LP) and 55 % (SS) while in the top section it is 44 % (LP) and 44 % (SS).

While belowground tree biomass is typically not part of the commercial yield of wood, belowground carbon stored in tree biomass was also estimated through a combination of root ball sampling and soil sampling for fine roots. 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, as is often the case (Bert and Danjon, 2006), but have been included as part of the total belowground biomass via the analysis of the upper layers of the peat.

**Appendix Table 1**
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 | C content (%) | Fresh:dry ratio | C content (%) |
| J (top) | 0.44 ± 0.12 | 47.90 ± 0.82 | 0.44 ± 0.02 | 48.37 ± 0.63 |
| I | 0.48 ± 0.11 | 48.08 ± 0.81 | 0.44 ± 0.02 | 48.29 ± 0.36 |
| H | 0.53 ± 0.11 | 47.46 ± 2.16 | 0.47 ± 0.05 | 48.32 ± 0.49 |
| G | 0.52 ± 0.07 | 47.25 ± 2.88 | 0.47 ± 0.04 | 48.37 ± 0.69 |
| F | 0.59 ± 0.09 | 48.17 ± 1.27 | 0.52 ± 0.05 | 48.58 ± 0.40 |
| E | 0.58 ± 0.09 | 48.65 ± 0.79 | 0.48 ± 0.03 | 48.21 ± 0.83 |
| D | 0.59 ± 0.08 | 48.33 ± 0.72 | 0.51 ± 0.03 | 48.42 ± 0.84 |
| C | 0.60 ± 0.07 | 48.95 ± 0.78 | 0.52 ± 0.04 | 48.12 ± 0.56 |
| B | 0.61 ± 0.07 | 48.48 ± 0.92 | 0.54 ± 0.04 | 48.38 ± 0.97 |
| A (base) | 0.62 ± 0.04 | 48.98 ± 0.51 | 0.44 ± 0.02 | 48.37 ± 0.63 |
| Roots | 0.42 ± 0.05 | 47.06 ± 1.64 | 0.55 ± 0.06 | 48.57 ± 0.60 |

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