Soil carbon balance of afforested peatlands in the maritime temperate climatic zone

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A. Jonay JOVANI-SANCHO¹,²,³, Thomas CUMMINS⁴ and Kenneth A. BYRNE¹*

¹: Department of Biological Sciences, School of Natural Sciences, University of Limerick, Limerick, Ireland
²: Current affiliation: School of Biosciences, University of Nottingham, Nottingham, NG7 2RD, UK
³: Current affiliation: UK Centre for Ecology & Hydrology, Environment Centre Wales, Bangor, Gwynedd, LL57 2UW, United Kingdom

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Abstract

Drainage and conversion of natural peatlands into forestry increases soil CO₂ emissions through decomposition of peat and modifies the quantity and quality of litter inputs and therefore the soil carbon balance. In organic soils, CO₂ net emissions and removals, are reported using carbon emission factors (EF). The choice of specific default Tier 1 EF values from the IPCC 2013 Wetlands supplement depends on land use categories and climate zones. However, Tier 1 EF for afforested peatlands in the temperate maritime climate zone are based on data from eight sites, mainly located in the hemiboreal zone, and the uncertainty associated with these default values is a concern. In addition, moving from Tier 1 to higher-Tier carbon reporting values is highly desirable when large areas are affected by land use changes. In this study, we estimated site-specific soil carbon balance for the development of Tier 2 soil CO₂-C EFs for afforested peatlands. Soil heterotrophic respiration and aboveground litterfall were measured during two years at eight afforested peatland sites in Ireland. In addition, fine roots turnover rate and site-specific fine root biomass were used to quantify belowground litter inputs. Our findings found that draining of peatlands, and planting them with either Sitka spruce or lodgepole pine, resulted in soils being net carbon sources. The soil carbon balance at multi-year sites varied between 63 ± 92 and 309 ± 67 g C m⁻² year⁻¹. Mean CO₂-C EF for afforested peatlands was 1.68 ± 0.33 t CO₂-C ha⁻¹ year⁻¹. The improved CO₂-C EFs presented here for afforested peatlands are proposed as a basis to update national CO₂-C emissions from this land-use class in Ireland. Furthermore, new data from these sites will significantly contribute to the development of more reliable IPCC default Tier 1 CO₂-C EFs for afforested peatlands in the maritime temperate climate zone.

Key words: heterotrophic respiration, Histosol, blanket peat, fine roots, forest litter, Sitka spruce, lodgepole pine, emission factors.
1. Introduction

Soil plays a major role in the global carbon (C) balance as it is the largest terrestrial pool of organic C worldwide with 1,500–2,400 Pg C in the upper 100–200 cm of the soil (Batjes, 2014). Of this, approximately 547 Pg C is stored in northern peatlands and organic soils (Yu et al., 2010) covering about 3% of the Earth’s land surface (Parish et al., 2008). In undrained or rewetted conditions, a high water table maintains anaerobic soil conditions, which restrict organic matter decomposition, while continuing plant-litter inputs maintain an accumulating, negative soil C balance. When peatlands are drained for other land uses such as peat extraction, agriculture or forestry, the soil becomes aerobic and the peat progressively decomposes thereby emitting large amounts of carbon dioxide (CO₂) through respiration by decomposer organisms (Parish et al., 2008). In addition, drainage alters the physical and chemical properties of peat and, while land-use changes modify the quantity and quality of litter inputs and therefore the soil C balance in peatlands (Hargreaves et al., 2003, Laiho and Pearson, 2016, Laiho and Finér, 1996).

From an approximate extent of 50 million ha of natural peatlands that have been drained worldwide (Joosten, 2010), around 15 million ha have been drained for forestry in northern biomes (Paavilainen and Päivänen, 1995). Moreover, it has been estimated that approximately 440,000 ha of peatlands have been drained for afforestation in Ireland (Renou-Wilson and Byrne, 2015) and 439,000 ha in the United Kingdom (Evans et al., 2017). Between 46 and 51% of total afforestation in Ireland during the period 1990–2005 was on organic soils (Black et al., 2008). Drainage for forestry increases soil CO₂ emissions through decomposition of peat, while forest growth accumulates biomass and litter C. Therefore these afforested peatlands can act as sinks or sources of CO₂ depending on the balance between C input and loss. Understanding the soil C balance of these extensive ecosystems has great importance in the global C cycle as CO₂ is the anthropogenic greenhouse gas (GHG) contributing most to climate change (IPCC, 2007). Although the net CH₄ exchange should also be considered in the soil C balance, net CH₄ fluxes in drained forest ecosystems are low (Yamulki et al., 2013, Coles and Yavitt, 2002, Zerva and Mencuccini, 2005) and they have their own emission factor (EF) reported as CH₄_organic (IPCC, 2014, Evans et al., 2017).

Approximately 26% of total CO₂ emitted (around 555 Pg C) during the industrial era (1870–2015) is derived from land use change activities (including deforestation, reforestation and afforestation) with the other main contributors being fossil fuel combustion and cement production.
(Le Quéré et al., 2015, Hartmann et al., 2013). Moreover, the atmospheric concentration of CO₂ has significantly increased from 317 ppm in 1958 to 412 ppm in 2018 (Keeling et al., 2001). To help tackle the negative effects of increased atmospheric GHG concentrations on the global climate, signatory countries of the Kyoto Protocol (UNFCCC, 1997), and the Paris Agreement (COP21), need to report national GHG inventories biennially under the United Nations Framework Convention on Climate Change. This includes monitoring and reporting soil C stocks (SCS), rates of change, and their associated GHG emissions.

In organic soils, CO₂ net emissions and removals, are reported using C EFs (in t CO₂-C ha⁻¹ year⁻¹) (IPCC, 2003). The Intergovernmental Panel on Climate Change (IPCC, 2014) established the need to disaggregate CO₂-C EFs among off-site losses (e.g. leaching of dissolved organic C; CO₂-C DOC), anthropogenic peat fires (Lfire-CO₂-C) and on-site emissions (CO₂-Con-site) — the EF considered in this study. In drained peatland forests, CO₂-Con-site EF represents the net decomposition loss, which can be estimated as the difference between soil CO₂ emissions originating from the decomposition of peat plus litter (heterotrophic respiration; RH), and CO₂ removals from litter accumulation (above and belowground inputs) (Ojanen et al., 2012, IPCC, 2014, Minkkinen et al., 2018, Evans et al., 2017).

Byrne and Farrell (2005) reported that soil respiration in afforested blanket peat in Ireland varies between 1.0 and 2.7 t C ha⁻¹ year⁻¹. The same authors concluded that it was likely that the soda-lime method used to measure soil respiration may have underestimated soil CO₂ fluxes because this method is not as reliable as infrared-absorption methods. Reported soil respiration values for afforested peatlands in oceanic climates and forestry-drained peatlands in boreal regions vary between 4.5 and 11.1 t C ha⁻¹ year⁻¹ (Ojanen et al., 2010, Mäkiranta et al., 2008, Yamulki et al., 2013, Jovani-Sancho et al., 2018, Minkkinen et al., 2018). Soil RH is commonly measured in partitioning studies of soil respiration using trenching techniques and portable soil respiration chambers (Mäkiranta et al., 2008, Saiz et al., 2006, Ferréa et al., 2012, Byrne and Kiely, 2006, Jovani-Sancho et al., 2018). Although more accurate methods with less soil disturbance effects exist to separate RH from total soil respiration, such as C isotope techniques, these methods are expensive and difficult to implement in the field or over large areas (Kuzyakov, 2006). In addition, trenching experiments and chamber methods are recommended for the estimation of the soil C balance over large areas (Ojanen et al., 2012). Nevertheless, it is important to acknowledge that trenching of roots generates some artifacts that may lead to overestimation or underestimation of...
These are: changes in soil moisture conditions, decomposition of roots, elimination of root exudates, and soil priming effects (Comstedt et al., 2011, Díaz-Pinés et al., 2010, Savage et al., 2018, Heinemeyer et al., 2012). Litterfall production can be estimated by direct measurements of tree litterfall using litterfall collectors (McShane et al., 1983, Saarsalmi et al., 2007) and indirect estimation of annual root inputs by applying root turnover rates to measured root biomass (Laiho et al., 2008). Furthermore, litter from ground vegetation (shrubs and mosses) should also be considered when appropriate (Laiho et al., 2008, Laiho et al., 2011, Ojanen et al., 2014).

The choice of specific default EF values to estimate net emissions—Tier 1 in the IPCC three-Tier scale of methodological approaches of increasing analytical complexity—depends on land use categories and climate zones (IPCC, 2014). In temperate drained afforested organic soils default EFs range between 2.0 and 3.3 t CO$_2$-C ha$^{-1}$ year$^{-1}$ (IPCC, 2014). However, these EFs are based on data from eight sites only and the uncertainty associated with these default values is a concern. Moreover, seven out of the eight study sites were located in the hemiboreal zone (Figure 1). In addition, moving from Tier 1 to region-specific and higher-Tier GHG reporting values is highly desirable when large areas are affected by land use changes (Wilson et al., 2015, IPCC, 2006) such as drainage and afforestation of natural peatlands. The same authors also recommend developing strong Tier 2 values for different land use categories and climate regions. The EF for afforested peatland currently used in Ireland’s GHG National Inventory Report (NIR) for the period 1990–2019 is 0.59 t CO$_2$-C ha$^{-1}$ year$^{-1}$ (Duffy et al., 2021). This EF is much lower than the Tier 1 default value range of 2.0 and 3.3 t CO$_2$-C ha$^{-1}$ year$^{-1}$ for the temperate zone (IPCC, 2014).

Duffy et al. (2021) attributed their lower EF to an interpretation of the soda-lime-derived soil respiration rates reported Byrne and Farrell (2005), though more reliable measures, and documented calculation, are now needed. Although the United Kingdom uses a model-based Tier 3 method to report CO$_2$ emissions from drained forest land, Evans et al. (2017) developed a Tier 2 EF of 2.0 t CO$_2$-C ha$^{-1}$ year$^{-1}$ for this land use and climatic zone. However, the same authors concluded that more studies were needed to reduce the large uncertainty of this EF because of the limited number of studies used to develop the EFs; the estimation of $R_{hi}$ fluxes was based on total soil respiration measurement (e.g. UK Tier 2 EF assumes that $R_{hi}$ represents 50% of total soil respiration); and the incomplete quantification of litter inputs into the soil (for both IPCC Tier 1 and UK Tier 2 values) (Evans et al., 2017). Similarly, net CO$_2$ emissions and removals for all C pools in the forest land category (including afforested peatlands) in Ireland are reported using a
Tier 3 model (i.e. CFS-CBM—Carbon Budget Model of the Canadian Forest Sector). However, this process-based model used in the NIR uses a static Tier 2 value to report soil CO₂ emissions from afforested peatlands (Duffy et al., 2021). Although a more dynamic EFs based on site characteristics, forest age, temperature and ground water level would allow to simulate soil CO₂ emissions on peatland forests more accurately, a static Tier 2 EF could be used to cross validate CO₂ predictions from Tier 3 models (Evans et al., 2017).

To understand whether soils under this land use are sinks or sources of C, and also to scale up results to regional and countrywide level, studies in which all the components of the soil C balance are quantified are necessary. Although several studies have been conducted to assess the soil C balance in forestry-drained peatlands in boreal and hemiboreal climates (Ojanen et al., 2013, Ojanen et al., 2014, Minkkinen et al., 2018, Meyer et al., 2013, Bechtold et al., 2018) and in afforested organo-mineral soils in temperate climates (Zerva et al., 2005, Friggens et al., 2020) the soil C balance of afforested peatlands in oceanic climates have received little attention. While undrained blanket peatlands in temperate maritime conditions are soil C sinks (excluding DOC losses and CH₄ fluxes) of around –0.56 ± 0.19 t C ha⁻¹ year⁻¹ (McVeigh et al., 2014, Koehler et al., 2011) the soil C balance of afforested peatlands is still uncertain. To the best of our knowledge, no work has been conducted in afforested peatlands in this climatic zone using the soil chamber method and accounting for estimates of all the litter inputs. The aim of this study was to estimate site-specific soil C balance for the development of Tier 2 countrywide soil CO₂-C EFs for afforested peatlands in temperate maritime climate conditions. Improved CO₂-C EFs presented here for afforested organic soils in Ireland are proposed as a basis to update national CO₂-C emissions from this land-use class. It was hypothesised that in these climatic conditions, soil CO₂ emissions from decomposition of peat and litter in drained afforested peatlands cannot be compensated by the C incorporation from litter inputs.

2. Methods

2.1. Study sites

Field studies were conducted at eight drained and afforested peatland sites located on the Mullaghareirk Mountains, in southern Ireland (Figure 1). The study sites were located in a maritime temperate climate zone — Cfb of the Köppen-Geiger climate classification by Kottek et al. (2006), characterized by abundant annual rainfall (1326–1716 mm year⁻¹) and mild mean...
annual air temperatures (8.9–10.9°C) (Table 1, Rockchapel, rainfall, and Mount Russell, temperature, weather stations for the years 2010–2020 and 1993–2020, respectively, Met Éireann, Irish Meteorological Service). The high precipitation, with nearly 200 days with 1 mm or more rainfall, in addition to the low annual potential evapotranspiration (about 500 mm year\(^{-1}\)) conditions prevailing in the area leads to persistently wet soils (Collins and Cummins 1996). Seven of the sites had plantations of Sitka spruce (\textit{Picea sitchensis} (Bong.) Carr.), between 18 and 44 years old (Table 2). The other site was a 23-years-old lodgepole pine (\textit{Pinus contorta} Dougl.) plantation (P23). All sites except S18 were first rotation plantations. All sites were established on poorly drained Dystric Histosols (IUSS Working Group WRB, 2015). All sites had closed canopy and the older sites were mature and ready for harvesting. In all sites, ground vegetation consisted of thin and patchy mosses over about half the ground area (\textit{Hylocomium splendens}, \textit{Pleurozium schreberi} and \textit{Polytrichum} sp.) and forest lichens. Ground vegetation was most abundant in the oldest sites (S39, S43 and S44) where thinning lanes (approximately 4 m wide) as a consequence of systematic thinnings and selective thinnings (in S43 only) had created gaps in the canopy enabling light the reach the forest floor. For a detailed description of the study sites see Jovani-Sancho et al. (2018).

2.2. Soil carbon balance

Site-specific soil C balance (\(\Delta C_{\text{soil}}\)) was calculated as the difference between the (negative) C inputs and (positive) C outputs of the soil (Ojanen et al., 2012), net accumulation of soil C being a negative balance (i.e. atmospheric view). Soil C inputs were dependent on plant production and consisted of the C incorporation from aboveground litterfall, belowground root litter, and moss-layer litter production. Sitka spruce and lodgepole pine were the only vascular plants growing in the study sites (except site P23 which had some scattered plants of \textit{Vaccinium myrtillus}), therefore, aboveground vascular-plant litterfall consisted of tree litterfall only. Given the low abundance of ground vegetation including mosses (see site description), soil C inputs from moss turnover were not included in the study. By contrast, C outputs consisted of CO\(_2\) emissions produced as a consequence of the combined aerobic decomposition of litter and peat. Therefore, and after omitting C outputs as CH\(_4\) fluxes and as off-site C losses in water, and C inputs from moss litter, the soil C balance was calculated as:

\[
\Delta C_{\text{soil}} = R_H - \text{Litter}_{AB} - \text{Litter}_{BG}
\]  

(1)
where $\Delta C_{soil}$ is the soil C balance, $Litter_{AG}$ is the aboveground tree litterfall and $Litter_{BG}$ is the belowground fine root litter. All components in Eq. 1 were expressed in g C m$^{-2}$ year$^{-1}$. In addition, at multi-year sites, single calculated mean values of each component were used in each site balance. Soil C EFs were reported in t CO$_2$-C ha$^{-1}$ year$^{-1}$.

2.3. Soil heterotrophic respiration

Between January and February 2014, seven measurement points were identified within each of the eight study sites. A stratification design based on the areal proportions of the different microtopographies of each site was used to position the measurement points. Three measurement points were established on the flat ground, two in the furrows and two on the tree-planting lines on peat ribbons or between mounds, depending of the study site. Each measurement point consisted of PVC root-cutting collar (PVC cylinder) with 15.4 cm internal diameter and 32 cm long driven vertically 30 cm into the soil. Collars produced minimum disturbance to the litter and peat layers while at the same time cut all roots, prevented root ingrowth, and provided a soil-surface contact for the soil respiration chamber. Mosses and tree seedlings growing inside the collars were removed at the beginning of the measurements and when necessary throughout the study period. Removable aluminium nets of 1 mm mesh size were laid over the collar openings to prevent soil disturbances by animals and to prevent the further accumulation of fresh litter within each collar between measurements.

Jovani-Sancho et al. (2017) demonstrated that, in these ecosystems, a 30 cm long trenching collar is sufficient to exclude root and mycorrhizal respiration by cutting all roots inside the collar. Soil CO$_2$ efflux at the measurement points, representing $R_H$, was measured weekly or fortnightly from June 2014 until mid-February 2016 using a portable infrared gas analyser attached to the collar, forming a closed chamber of 15.4 cm diameter and 14.8 cm height (2755 cm$^3$) (EGM-4 and modified SRC-1; PP Systems Ltd., UK). Concurrently with the $R_H$ measurements, soil temperature (T) was measured at 10 cm soil depth and water table depth (WTD) was measured inside perforated plastic pipes inserted into the peat. In addition, at each site, soil T was recorded continuously at 10 cm soil depth. While continuous soil T was averaged for hourly and daily values, WTD gaps between consecutive measurements were filled by linear interpolation. For a full description of the soil respiration chamber and the measurements method see Jovani-Sancho et al. (2018).
Jovani-Sancho et al. (2018) developed site-specific RH models by multiple nonlinear regression analysis using mean hourly RH, soil T and WTD values for each site (mean of seven subsites) and sampling event. After comparing several models, the same authors reported that Eq. (2) and Eq. (3) were the best models to simulate hourly RH effluxes at site S38 and at all the other sites, respectively,

\[ y = a_i (exp)^{b_i T} \times (exp) \left[ -0.5 \left( \frac{WTD - WTD_i}{d_i} \right)^2 \right] \]  

(2)

\[ y = a_i (exp)^{b_i T} + c_i (exp) \left[ -0.5 \left( \frac{WTD - WTD_i}{d_i} \right)^2 \right] \]  

(3)

where \( y \) is the measured soil CO₂ efflux rate, \( T \) is the measured soil temperature at 10 cm depth, \( a_i \) and \( b_i \) are fitted parameters greater than 0 obtained by nonlinear regression analysis, \( WTD \) is the measured water table level, \( c_i, d_i \) and \( WTD_i \) are specific-fitted parameters determined using least squares nonlinear regression (Table S1). Daily CO₂ emissions were calculated by summing hourly simulated values over a 24 hours period. Thereafter, annual RH effluxes were estimated by summing daily simulated values over two 12-months periods (Year 1: June 1, 2014 to May 31, 2015; Year 2: June 1, 2015 to May 31, 2016). For a full description of the RH modelling analysis see Jovani-Sancho et al. (2018).

2.4. Aboveground litterfall

At each site, seven circular litterfall traps (0.08 m²), installed next to each soil respiration subsite, were used to collect Litter_AG. Litterfall traps consisted of PVC containers (27 cm high) perforated at the bottom to allow free drainage of water. To prevent accumulation of water between the forest floor and the bottom of the litterfall traps, these traps were installed about 10 cm over the forest floor by placing blocks of woods below them. Litterfall traps were installed during April 2014 and all of them were emptied on the 31st of May 2014. Thereafter, Litter_AG was collected monthly from the end of June 2014 to the end of May 2016. Within 24 hours, collected litterfall samples, placed inside foil trays, were first air-dried inside a poly tunnel for 48 hours. Then, samples were oven-dried for 72 hours at 65°C to assess dry Litter_AG. Results were scaled up to site level and monthly and annual Litter_AG were calculated for each site. Based on (Reidy and Bolger, 2013), the C content of aboveground litterfall was assumed to be 47.57%.
2.5. Root litter inputs

In July 2015, seven soil peat cores, including the litter (recognisable leaves, twigs and small branches; L) and fermentation (a mix of partially decomposed organic matter, roots and fungi; F) layers were taken within each study site. Samples were taken following the same stratification design used for the R_H measurements. Therefore, three cores were taken in the flat area, two in the furrow microtopography and two in the tree-planting lines. Continuous volumetric soil cores (internal dimensions of 7.0 × 7.5 × 80 cm) were collected, using a modification of the volumetric peat sampler proposed by Jeglum et al. (1991). Prior to sampling, the sampler was gently pushed into the soil. Then, using a sharp knife, soil and roots were pre-cut through the top 20 cm. Thereafter, the sampler was completely pushed into the soil. Using a lever and a tripod, the sampler was extracted from the soil, and the soil core (containing the roots) was then divided into ten segments at the following depths: 0.0–2.5, 2.5–5.0, 5.0–7.5, 7.5–10.0, 10.0–12.5, 12.5–15.0, 15–20, 20–30, 30–50 and 50–80 cm. Each soil segment was sealed in a plastic bag and kept at −18°C until processed. Living roots were manually sorted and fine roots < 2 mm were washed to remove soil. Roots were then oven-dried for 72 hours at 65°C to calculate fine root dry biomass. Finally, results were scaled up to site level.

Belowground litter inputs were estimated from the measured root biomass and a mean root turnover rate of 0.33 (range of 0.20–0.65). This turnover rate was derived from a Sitka spruce chronosequence (5 sites of ages varying between 12 and 41 years old) growing on organo-mineral soils within the same study area as the present study (Lane, 2016). In this study, a split tube sampler of 4.8 cm diameter was used to collect 20 cm long soil cores to measure fine root biomass (< 2mm). In addition, and based on the method proposed by Hirano et al. (2009), Lane (2016) calculated fine root production by inserting nylon meshes (10 × 20 cm) of 1 mm² aperture vertically into the soil. In each study site, 49 meshes were inserted across the different microtopographies (21 in the flat, 14 in the furrow and 14 in the ribbon locations). After digging out the meshes (between 12 and 18 months after inserting them), all roots that had grown through the mesh were trimmed to 1 cm (on both side of the mesh) and all fine roots ≤ 2 mm were collected to quantify fine root production. The root turnover rate was calculated as the ratio between annual fine root production in one year and total root biomass. Although the soil types differ, this turnover ratio constitutes the best available estimate for Sitka spruce under the same climatic conditions. The same root turnover rate was used to estimate root litter inputs at the
lodgepole pine site. Finally, total belowground C inputs were calculated for each site by multiplying the root litter production by a C content of 46.58% (Olajuyigbe et al., 2012). Site S39 was clearfelled in June 2015 before the collection of the peat cores. Therefore, an average for fine root biomass (i.e. average of the other six Sitka spruce sites) was used in the soil C balance in this site was

2.6. Statistics and uncertainty

It was assumed that all component fluxes were independent variables. Therefore, the 95% confidence intervals and the uncertainty of the soil C balance were calculated as the mean ± 1.96 × SE, and following the law of error propagation, respectively (Renou-Wilson et al., 2014),

\[ SE_{\Delta C_{Soil}} = \sqrt{SE_{R_H}^2 + SE_{Litter,AB}^2 + SE_{Litter, BG}^2} \]  

Statistical analyses were performed using GENSTAT 19 (VSN International, UK). The impacts of the microtopographies and site, soil T and WTD on R\(_H\) effluxes were investigated using mixed linear models, with repeated measurements, using the residual maximum likelihood (REML) method. In the model, the measuring points were used as subjects, the site was used as the random effect and the date were used as the time points. Site and microtopography (flat, ribbon and furrow) were the fixed effects in the model and soil T and WTD were added as covariates. The effects of the interactions between site and microtopography on the R\(_H\) effluxes were also investigated. Prior to the REML analysis, variables that had either negative or zero values (i.e. R\(_H\) and WTD) were converted into positive scales only. Thereafter, a visual inspection of the quantile-quantile plots (Q-Q plots) and Skewness, Kurtosis and Shapiro-Wilk tests were performed to assess the normality assumption of the data. Finally, not normally distributed data was transformed using a Box-Cox transformation. All the statistical tests were realized at the \( P = 0.05 \) significance level. All raw data were processed with EXCEL 2010; multiple non-linear regressions were conducted using SigmaPlot 12.0 (Systat Software Inc. USA). Figures were produced in R version 3.6.3 (R Core Team, 2013) using the R package ggplot2 (Wickham, 2009).
3. Results

3.1. Soil heterotrophic respiration

Heterotrophic respiration was closely related to soil temperature and followed a seasonal pattern with minima in late winter and increasing gradually into late summer (Figure 2). Similarly, lowest and highest CO₂ emissions were observed during the late winter and late summer months, respectively (Figure 3). Across the seven Sitka spruce sites, the absolute minimum and maximum hourly \( R_H \) effluxes measured at individual microtopographic sub-sites were 0 and 1.60 g CO₂ m\(^{-2}\) h\(^{-1}\), respectively. At the lodgepole pine site, the range of hourly \( R_H \) fluxes was 0.02 to 0.74 g CO₂ m\(^{-2}\) h\(^{-1}\). Site mean hourly \( R_H \) (± SE, n = 7) varied between 0.18 ± 0.01 and 0.30 ± 0.02 g CO₂ m\(^{-2}\) h\(^{-1}\) and the REML test showed that \( R_H \) in this site was significantly lower than in all the other sites (\( P < 0.001 \); Table 3 and Figure S1). In addition, among the Sitka spruce sites, mean hourly \( R_H \) was lowest at S43 (\( P < 0.001 \)) and highest in S27 but it was not significantly different than hourly \( R_H \) rates in sites S18, S27 and S39 (Figure S1). Although S18 and S28 did not show any difference in \( R_H \) across the three microtopographies, this effect was statically different in other sites (\( P < 0.001 \)). Heterotrophic respiration was highest in the furrow microtopography in sites S24 and S43 (Figure S2). However, in sites S27, S39, S44 and P23, \( R_H \) effluxes from the furrow microtopography were not different from those measured in the ribbon. Lowest mean \( R_H \) efflux were measured in the flat microtopography in site P23. Simulated annual \( R_H \) across all sites ranged between 377 and 679 g C m\(^{-2}\) year\(^{-1}\) (Table 4). Mean annual \( R_H \) at multi-year sites ranged between 416 ± 14 g C m\(^{-2}\) year\(^{-1}\), at site S43, and 607 ± 53 g C m\(^{-2}\) year\(^{-1}\), at site S18. Sites S27 and S39 were clearfelled in June 2015 and therefore annual \( R_H \) was calculated for a single year only, and was 627 and 580 g C m\(^{-2}\) year\(^{-1}\), respectively.

3.2. Aboveground litterfall

Across the seven Sitka spruce sites, mean monthly litterfall varied between 5 and 139 g m\(^{-2}\) month\(^{-1}\) (Figure 4 and Table 4). These were recorded in April 2015 at S43 and June 2015 at S24 sites, respectively. At the lodgepole pine site, minimum and maximum monthly litterfall were 3 g m\(^{-2}\) month\(^{-1}\) (in April 2015) and 226 g m\(^{-2}\) month\(^{-1}\) (in November 2015), respectively. Although no clear seasonal pattern of litterfall was observed at any of the sites, litterfall inputs were, on average, 37.8% greater in Year 2 than in Year 1. In addition, it was not possible to collect litterfall samples in February and July 2015 at any of the sites, or in December 2014 at the S24 site.
Therefore, litterfall from those months was included in the consecutive month of sampling and the two-month collection amount was split in half. Mean annual litterfall at the Sitka spruce sites varied between $160 \pm 31$ g C m$^{-2}$ year$^{-1}$ (at site S43) and $320 \pm 38$ g C m$^{-2}$ year$^{-1}$ (S28). Mean annual litterfall at the lodgepole pine site was $245 \pm 16$ g C m$^{-2}$ year$^{-1}$.

### 3.3. Root biomass

Total dry fine root biomass at the Sitka spruce sites varied between $600.0 \pm 73.4$ g m$^{-2}$ at the youngest site, S18, and $1181.0 \pm 223.1$ g m$^{-2}$ in site S27. Fine root biomass at the lodgepole pine site was $496.2 \pm 55.4$ g m$^{-2}$. All study sites had similar vertical fine root distributions, with over 98%, on average, in the top 20 cm of soil (Figure 5). All Sitka spruce sites had 100% of the fine roots located in the top 30 cm of the soil profile. The lodgepole pine site had less than 1% of the fine roots biomass between the 30–50 cm depth-interval. Root turnover varied between $92 \pm 11$ g C m$^{-2}$ year$^{-1}$ (at site S18) and $182 \pm 34$ g C m$^{-2}$ year$^{-1}$ (at site S27). Similarly, root turnover at the pine site was $76 \pm 9$ g C m$^{-2}$ year$^{-1}$ (Table 4).

### 3.4. Soil carbon balance

Across all sites, the single-year soil C balance varied between $-86 \pm 28$ and $422 \pm 12.6$ g C m$^{-2}$ year$^{-1}$ (Figure 6 and Table 4). In addition, the soil C balance at multi-year Sitka spruce sites (not S27 and S39, where only one year was used) varied between $63 \pm 92$ and $309 \pm 67$ g C m$^{-2}$ year$^{-1}$. The multi-year soil C balance at the lodgepole pine site was $107 \pm 40$ g C m$^{-2}$ year$^{-1}$ (Figure 7). Thereafter, a mean soil C balance for afforested peatland in maritime temperate conditions was derived for Sitka spruce plantations. This soil C balance is the equivalent as the soil C EF. Therefore, the calculated EF for this land use category and tree species was $1.77 \pm 0.34$ t CO$_2$-C ha$^{-1}$ year$^{-1}$. The mean C EF for afforested peatlands, independent of tree species, was $1.68 \pm 0.33$ t CO$_2$-C ha$^{-1}$ year$^{-1}$ (Table 5).

### 4. Discussion

This study represents one of the few soil C balance studies in afforested peatlands in maritime temperate conditions. The combination of soil respiration measurements using chamber methods with the quantification of C inputs into the soil constitutes a useful and practical approach to assess whether these soils are acting as C sources or sinks. The main challenge of this method is the necessity to estimate all the C components accurately (Ojanen et al., 2012). Although soil C
inputs from ground vegetation were not measured at the field sites, the potential implications of such inputs are included here,

4.1. Soil carbon balance

Draining of blanket peatlands, and planting them with either Sitka spruce or lodgepole pine, resulted in soils being net C sources. That is, these peatlands emitted more C to the atmosphere than they accumulated in the soil. Only one site had inconclusive results, due to the large uncertainty associated with this site. The 95% confidence interval of the multi-year soil C balance in S24 varied between –117 and 242 g C m$^{-2}$ year$^{-1}$. The annual soil C balance varied across sites and also between the two studied years. Maximum air and soil temperature were higher during Year 1 than Year 2. In addition, Year 1 was dryer than Year 2 (i.e. less rainfall and deeper WTD). These differences were greater during the months of maximum biological activity (June to September), with air and soil temperature being 1.7 and 1.4°C greater in Year 1 than in Year 2. Rainfall and WTD, for the same period, were 42.1 and 20.8% lower and deeper, respectively, in Year 1 than in Year 2. This temporal variation in the climatic conditions between years (Year 1 being drier and warmer) resulted in higher $R_H$ fluxes and less aboveground litterfall in Year 1 than in Year 2, and therefore the soil lost three times more C to the atmosphere in Year 1 than in Year 2 (assuming the same fine root turnover in both years). A similar effect of the temperature, rainfall and WTD conditions on soil CO$_2$ emissions was found by Wilson et al. (2016) when developing C EF for peatlands managed for extracted in the same climatic zone. In addition, (Jovani-Sancho et al., 2018) showed that RH fluxes increased with deeper WTD (greater aerobic conditions in the peat) and that this CO$_2$ efflux was maximum when WTD was –66 cm. This would support the present results.

Although no other soil C balance estimates for afforested peatland in the same climate zone exist in the literature, the soil C balance estimates reported in the present study contrast with some studies from boreal peatland forests. Bjarnadottir et al. (2021), using eddy covariance techniques, found that a Black Cottonwood ($Populus balsamifera$ ssp. $trichocarpa$) plantation on a shallow-drained peatland was a soil C sink of –55 g C m$^{-2}$ year$^{-1}$. Similarly, Minkkinen et al. (2018), also using eddy covariance methods, reported that a boreal drained peatland forest acted as a soil C sink of –60 g C m$^{-2}$ year$^{-1}$. However, another research in the same study site in Finland concluded that, if the same chamber method than the one used in the present study was used instead of the eddy covariance method, the soil C balance would vary between –16 ± 44 and 106 ±
44 g C m⁻² year⁻¹ (Ojanen et al., 2012), illustrating the uncertainties in the chamber method. Differences between the boreal and the Irish sites are likely due to differences in mean annual temperature (4.3 and 5.1 vs 10.0°C, respectively), land-use history, drainage intensity and soil properties. Furthermore, differences in soil Rₜₚ fluxes (475 vs 521 g C m⁻² year⁻¹; in the Finnish and the Irish sites—average of the eight study sites, respectively) could also help explaining the different soil C balances between the boreal and the temperate sites. Another study in a Norway spruce-dominated (Picea abies) afforested peatland in south-western Sweden reported that the soil net C efflux varied between 630 and 810 g C m⁻² year⁻¹ depending on the method used (eddy covariance techniques vs. chamber and trenching methods) (Meyer et al., 2013). The same authors attributed this large net soil C efflux to the very high Rₜₚ emissions measured in this nutrient-rich and former agricultural land (i.e. 1300 g C m⁻² year⁻¹) and to the accumulated uncertainties associated to the chamber method. Similarly, another chronosequence study in a forestry-drained peatland with natural regeneration of pure downy birch (Betula pubescens) found that these sites acted as continuous soil C sources (Uri et al., 2017). This study, based in chamber and trenching methods, reported that the soil C balance varied between 161 and 293 g C m⁻² year⁻¹.

Another reason for the rather contradictory results, between the previously compared studies, in the reported soil C balances could be attributed to the presence of mosses (and other herbaceous vegetation) and the incorporation of this plant litter into the soil C balance. Uri et al. (2017) reported that in their study site, the herbaceous litter varied between 5 to 19 g C m⁻² year⁻¹. In addition, Laiho et al. (2011) found that, in drained peatland forests, mosses may incorporate significant amounts of C into the soil. Furthermore, Badorek et al. (2011) reported that ground vegetation represent 20–30% of the total gross primary production in a drained peatland forest and Minkkinen et al. (2018) found that, in the same study site, moss litter input represented over 20% of the total litter production (i.e. 90 g C m⁻² year⁻¹). Notwithstanding, the presence of vascular plants at the Irish sites was insignificant and only sites S39, S43 and S44 had mosses in ditches and open-canopy areas (i.e. thinning lanes). By contrast, the Finnish site, with lower tree density and better forest floor light conditions, had 90–100% of the forest floor covered by mosses (Badorek et al., 2011, Minkkinen et al., 2018). Although it is not possible to quantify the moss litter input it is likely that the belowground litterfall inputs may have been underestimated in the present soil C balance. Further studies should aim to quantity moss litter inputs (and other vascular
presents if present) in afforested peatlands to help reducing the uncertainty of the soil C balance and develop more accurate estimates.

It is clear that the selection of different methods to calculate the soil C balance has important implications when assessing if a peatland forest site is acting as a C sink or a source. While eddy covariance techniques produce more accurate results than chamber methods, its applicability to larger areas is more complex and requires expensive equipment (Ojanen et al., 2012). By contrast, chamber measurements can be conducted over large areas easily and portable CO$_2$ analysers, which have become widely available, facilitates the collection or large datasets over short periods of time. However, the determination of $R_H$ fluxes using trenching collars is affected by several factors such as addition of extra organic matter from severed roots, exclusion of root exudates and fresh litterfall that stimulates decomposition of recalcitrant organic substrates and changing of soil moisture conditions by interrupting the lateral water flow in the soil and increasing the risk of water pooling inside the trenching collars and reducing soil CO$_2$ emissions (as observed in some collars installed in the furrow microtopography during the winter months—see Figure 3, Site S27) (Kuzyakov et al., 2007, Moore et al., 2020). Although some of these artifacts may have underestimated and / or overestimated soil CO$_2$ emissions simultaneously through the length of this study, it is hypothesized that their effect on the overall $R_H$ flux was minimized by delaying the first measurements between 4 and 6 months after trenching, because no root regrowth was observed inside the trenched area at the end of the experiments and because the increase of water content within the collar may have been not significant as reported by Mäkiranta et al. (2008) in a similar trenching experiment in Finland—although soil moisture content was not measure in the Irish study. Another limitation of chamber method is that the proposed $R_H$ models require continuous measurements of soil $T$ and WTD to simulate annual CO$_2$ fluxes accurately. While soil temperature is usually recorded at weather stations and it may be estimated accurately over large areas using remote sensing techniques (Xu et al., 2020), continuous WTD data on peatland forests is scarce and the use of remote sensing techniques to simulate WTD time series accurately needs further refinement (Burdun et al., 2020, Bechtold et al., 2018). Therefore, if continuous WTD is not available or if the uncertainty of simulating WTD gaps between consecutive measurements by linear interpolation is a concern, it is suggested that temperature exponential models are used, instead of combined model of soil temperature and WTD, to scale up $R_H$ effluxes at a regional level (Jovani-Sancho et al., 2018).
4.2. Litterfall inputs and uncertainty

In this study, fine root biomass was greatest in the top 10 cm of soil, decreasing exponentially with soil depth (Figure 5). This was also reported by Steele et al. (1997) in a mature black spruce (Picea mariana (Mill.)) stand in Canada. Across all spruce sites, 100% of all measured fine root biomass was found in the upper 30 cm of the soil profile, with a tiny proportion of pine roots in the 30–50 cm zone. Thus, it is possible to conclude that the trenching collars completely terminated the $R_A$ efflux.

Mean dry root biomass for the Sitka spruce and the lodgepole pine sites were 865.2 and 496.2 g m$^{-2}$ respectively. This is much higher that fine root biomass range reported for a Sitka spruce chronosequence in organo-mineral soils by Lane (2016) which varied between 180 ± 120 and 490 ± 80 g m$^{-2}$. Other values found in the literature for dry root biomass are 748 and 448 g m$^{-2}$ for Sitka spruce and lodgepole pine stands respectively, growing on blanket peat in Ireland (39 and 27 years old respectively) (Byrne and Farrell, 2005), 772 g m$^{-2}$ (using a C concentration factor of 46.58% (Olajuyigbe et al., 2012)) in a 30 year old Sitka spruce stand growing on a peaty gley soil (Zerva et al., 2005), and 264 g m$^{-2}$ in a 31 year old Sitka spruce stand growing on a mineral gley soil (Saiz et al., 2006). Previously reported fine root (< 1 mm) biomass estimates for Norway spruce (Picea abies (L.) Karst.) range 100–1090 g m$^{-2}$ (Santantonio et al., 1977). The measured fine root values in our sites are in the higher end of previously reported values for each species. We believe that the large amount of fine roots found in our sites is an adaptation of the tree species that maximises the uptake of nutrients and oxygen from these waterlogged nutrient-poor soils. The low oxygen at depth, and low fertility of these soils led to shallow root systems, growing almost exclusively in the F and L layers where most of the nutrients are located and recycled. This suggestion would be supported by the previously reported and new values for Sitka spruce, where fine root biomass increased in the order: blanket peat soils < peaty Gleys < mineral Gleys (Saiz et al., 2007, Zerva et al., 2005).

One of the main limitations of the present study was the lack of site-specific fine-root turnover rates. The fine root turnover rate of 0.33 (range of 0.20–0.65) reported by Lane (2016) is similar to the fine root turnover rate (roots < 2 mm diameter) reported for Pinus sylvestris in southern Finland (between 0.34 and 0.75) (Finér and Laine, 1998) but lower than fine roots turnover for spruce and pine forest in Finland (i.e. 0.81 and 0.87, respectively) reported by (Liski et al., 2006). The fine root turnover rate developed by Lane (2016) is, to the best of our
knowledge, the only value for Sitka spruce that exist in the literature. Moreover, this fine root turnover rate was developed in the same study area, under the same environmental conditions and using a similar chronosequence and experimental design approach (similar ages, species, root diameters and sampling strategy). Therefore, with the current existing data, using this fine root turnover rate is the best approximation to estimate belowground litter inputs in the present soil C balance study. Notwithstanding, previous studies in boreal peatlands determined that the selection or lower or higher fine root turnover rates (measured or reported in the literature) had a significant effect on the overall soil C balance, determining whether the site was a C sink or source (Ojanen et al., 2012, Ojanen et al., 2014). Literature values of fine root turnover rates for different species are very scarce but Ojanen et al. (2012) concluded that using the higher turnover rates resulted in the most accurate values when compared with eddy-covariance-estimated soil C balance. In addition, the sampling method used to measure fine roots is susceptible to errors (Byrne and Farrell, 2005) and it is possible that the values reported in this new study could have been overestimated due to misidentification of live roots. Further studies should aim to reduce the total fine root biomass uncertainty and also to develop turnover rates specifically for the most common tree species (i.e. Sitka spruce and lodgepole pine) growing on afforested peatlands. This will help reducing the overall uncertainty of the soil C balance estimate and improve the accuracy of the soil EF factors used to report CO₂ emissions at the national and international level from this land-use category.

4.3. Soil C emissions factors and implications for National Inventory

Although the IPCC (2014) Wetlands Supplement Guidelines includes default Tier 1 values for drained forest land in oceanic climates, they also encourage countries to move from Tier 1 to country-specific Tier 2 approaches. This means that countries should develop their own EFs for each land use category and climatic zones when appropriate. The mean CO₂-C EF reported in this study (1.68 t CO₂-C ha⁻¹ year⁻¹) is within the lower end of Tier 1 default values for cold wet temperate conditions (IPCC, 2014) and UK Tier 2 values (Evans et al., 2017) (Table 5). Nevertheless, this new EF is almost three time greater than the current CO₂-C EF used in the Ireland National Inventory Reporting system (0.59 t CO₂-C ha⁻¹ year⁻¹) (Duffy et al., 2021). The CO₂-C EF used by Duffy et al. (2021) was estimated from annual total soil respiration (which includes root respiration) values that were measured by the soda-lime method (Byrne and Farrell, 2005). It is known that the soda-lime and alkali traps methods may underestimate high soil CO₂ effluxes and their accuracy and performance remain uncertain (Rochette et al., 1992, Janssens et
al., 2000, Rochette and Hutchinson, 2005). Thus, it is very likely that the CO₂-C EF reported for Ireland have been underestimated. Similarly, (IPCC, 2014) and Evans et al. (2017) also estimated R₉ₐ fluxes from studies that measured total soil respiration (except Minkkinen et al. (2007) within the IPCC Tier 1 references that measured R₉ₐ fluxes). In addition, none of the developed Tier 1 and Tier 2 EF incorporated a complete litter cycling (i.e. below and above ground litter inputs measured simultaneously with the R₉ₐ fluxes). These C additions had to be estimated from other studies which may have produced large uncertainties and potential biases (Evans et al., 2017). Although the present study lacks site-specific root turnover rates, moss and fungal litter inputs it also quantifies most of the C inputs and outputs simultaneously allowing to the development of a good estimate of the site-specific soil C balance.

The main issue regarding the use of default Tier 1 EFs is the large uncertainty associated with them (Wilson et al., 2015, Renou-Wilson et al., 2016). The development of country-specific Tier 2 EFs in this study has an uncertainty of approximately 38%. This apparently large uncertainty is due to the spatial and temporal variability encountered during the two-year duration of this study and the differences found between the study sites. However, this study adds CO₂-C EFs from eight new sites to the default Tier 1 values of the “Forests Land, drained” land use category for the temperate zone reported in the IPCC (2014) Wetlands Supplement Guidelines. The total area of stocked afforested peatlands in Ireland in 2017 was estimated in 260730 ha and the estimated C emissions from this land-use category using the Duffy et al. (2021) EF would be 153831 t CO₂-C. If the new EF reported in this study is used in the 2019 GHG National Inventory Report, we estimate that on-site CO₂-C emissions from afforested organic soils in Ireland would be between 1.8 and 3.9 times greater (i.e. 271159–604894 t CO₂-C) than the current reported emissions but also around 35 and 16% lower than CO₂-C emissions estimated with the IPCC (2014) default Tier 1 and UK Tier 2 values (Table 6).

4.4. Implications for sustainable land management

These plantation forests on drained blanket peatland are net soil C emission sources, in this maturing phase. With assumed greater emissions in establishment and post-harvest phases (Hargreaves et al., 2003), the soil-based assessment of C balance used under the UNFCCC for these managed ecosystems is consistently one of loss of C to the atmosphere as CO₂. Because of losses from decomposing peat due to oxygen entry following drainage, above- and below-ground C input is too small to give net soil C sequestration. This status demonstrated here suggests that
afforestation of blanket peat, with necessary installation of drains, is unsuited as a strategy to fix atmospheric C. When considering the C balance of these forests, soil C storage is crucial to maintaining and increasing any C storage potential. While the tree layer will sequester C, the merchantable fraction of stemwood will be removed and converted into wood products with the harvest residues remaining on site where they will decompose and contribute to CO₂ emissions. Root litter and moss litter (the latter not evaluated here) will contribute a proportion of humus to soil, and are contributions to C balance requiring further quantification. Design of measures to reduce emissions, such as rewetting, should not be limited to site level consideration but should also include potential impacts on other ecosystem services at landscape and regional scale.

5. Conclusion

This study confirms that drained afforested blanket peatland soils are net sources of CO₂ emissions from oxidation of soil C. In addition, the new data from eight sites will significantly contribute to the development of more reliable, and with lower uncertainty, IPCC Tier 1 CO₂-C default values for afforested peatlands in the maritime temperate climate zone. However, further studies are necessary to reduce the uncertainty associated with these findings such as belowground C allocation and interannual variation over several years. In addition studies should be extended to include regional differences in forested blanket peatland, different peat and forest types, as well as alternative management strategies such as rewetting. Furthermore, afforested blanket peatlands with low forest productivity (e.g. low yield classes) in where alternative management strategies, with the aim of maximizing litter input accumulation into the soil and decreasing peat oxidation, are not feasible should be prioritized as C storage and forest-based environmental services ecosystems rather than as commercial tree plantations.

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Table 1 Climatic data consisting of minimum (Min), maximum (Max) and mean air temperature, soil temperature, water table depth and rainfall measured during year 1 (01/06/2014–31/05/2015), year 2 (01/06/2015–31/05/2016) and long term average. Air temperature (June 1993 to September 2020) from Mount Russell weather station (40 km east of the study sites and rainfall (Jun 2010 to July 2020) from Rockchapel weather station (within the study site area) (Met Eireann). Measured soil temperature and water table depth represent the average for the eight study sites.

|                        | Year 1 | Year 2 | Long term average |
|------------------------|--------|--------|-------------------|
| **Air temperature (°C)** |        |        |                   |
| Min                    | -0.2   | 1.9    | 4.6               |
| Max                    | 20.9   | 17.6   | 16.1              |
| Average                | 10.1   | 10.0   | 10.0              |
| **Soil temperature (°C)** |        |        | n.a               |
| Min                    | 0.6    | 1.1    | n.a               |
| Max                    | 19.9   | 17.0   | n.a               |
| Average                | 9.4    | 9.3    | n.a               |
| **Water table depth (cm)** |       |        | n.a               |
| Min                    | -56.9  | -42.3  | n.a               |
| Max                    | -24.3  | -20.7  | n.a               |
| Average                | -37.6  | -32.2  | n.a               |
| **Rainfall (mm)**      |        |        |                   |
| Annual                 | 1295   | 1843   | 1532              |

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Table 2 General site features at the Sitka spruce (S) and lodgepole pine (P). Description include stand basal area (ba) and soil carbon stocks (SCS)

| Site code | Peat depth (cm) | Ground preparation | Slope (%) | Density (trees ha\(^{-1}\)) | Mean ba (m\(^2\) ha\(^{-1}\)) | Yield Class | Bulk density (g cm\(^{-3}\)) | SCS (t C ha\(^{-1}\)) |
|-----------|-----------------|---------------------|-----------|-----------------------------|-----------------------------|-------------|-----------------------------|---------------------|
| S18       | > 80            | Mounded             | 2.1       | 3,000                       | 43.5                        | 16          | 0.102                       | 468.6               |
| S24       | 65-75           | Mounded             | 4.4       | 2,767                       | 51.8                        | 14          | 0.110                       | 456.8               |
| S27       | > 80            | Double ploughed     | 4.6       | 2,469                       | 63.9                        | 16          | 0.131                       | 604.4               |
| S28       | 60-75           | Single ploughed     | 4.8       | 1,733                       | 93.4                        | 20          | 0.159                       | 579.6               |
| S39       | 50-60           | Double ploughed     | 4.1       | 2,345                       | n.d                         | 14          | 0.260                       | 787.1               |
| S43       | > 80            | Double ploughed     | 2.3       | 2,133                       | 55.8                        | 10          | 0.093                       | 509.0               |
| S44       | > 80            | Single ploughed     | 2.2       | 1,733                       | 72.6                        | 10          | 0.114                       | 615.5               |
| P23       | > 80            | Mounded             | 1.8       | 2,500                       | 52.5                        | 12          | 0.098                       | 499.3               |

S – Sitka spruce; P – lodgepole pine. Subscript number under the site code correspond to age of the plantation (in 2015). SCS measured for the top 80 cm of soil (plus the litter layer) except in S24 and S39 that SCS represents the top 75 and 65 cm of soil (plus the litter layer), respectively.
Table 3 Statistics results from the mixed linear models with repeated measurements, describing the
treatment effects of the fixed effects site and microtopography and the covariates soil temperature
at 10 cm (soil T) and water table depth (WTD) on heterotrophic respiration ($R_H$). Prior to the
analysis, $R_H$ and WTD were Box-Cox transformed to fulfil the normality assumption.

| Fixed term                  | Fixed effect | Wald statistic | n.d.f. | F statistic | d.d.f | F pr  |
|-----------------------------|--------------|----------------|--------|-------------|-------|-------|
| Site                        |              | 130.0          | 7      | 18.6        | 647.6 | <0.001|
| Microtopography             |              | 17.0           | 2      | 8.5         | 638.9 | <0.001|
| Site × Microtopography      |              | 93.4           | 14     | 6.7         | 653.4 | <0.001|
| Soil T                      |              | 1061.7         | 1      | 1061.1      | 1359.0| <0.001|
| WTD                         |              | 187.6          | 1      | 187.6       | 1699.3| <0.001|

Significant differences ($p < 0.05$) are highlighted in italics; n.d.f, numerator degrees of freedom; d.d.f, denominator degrees of freedom; F pr, is the p-value associated with the F-statistic.
Table 4 Soil carbon balance (SCB) calculated as the difference between output in heterotrophic respiration ($R_H$), and sum of inputs in litters above ground (Litter$_{AG}$) and below ground (Litter$_{BG}$). Fine-root biomass pool is given for comparison with turnover. Positive values for the carbon components and also the carbon balance indicate net ecosystem loss of carbon to the atmosphere. All components are expressed in g C m$^{-2}$ year$^{-1}$

| Site | Year | $R_H$ | Litter$_{AG}$ | SE Litter$_{AG}$ | FRB | Litter$_{BG}^{**}$ | SE Litter$_{BG}^{**}$ | SCB | SE SCB |
|------|------|-------|---------------|----------------|-----|------------------|-------------------|-----|--------|
| S18  | Y1   | 679   | -165          | 6              | 279 | -92              | 11                | 422 | 13     |
| S18  | Y2   | 520   | -232          | 8              | 279 | -92              | 11                | 195 | 14     |
| S24  | Y1   | 615   | -235          | 14             | 511 | -169             | 21                | 212 | 26     |
| S24  | Y2   | 456   | -374          | 17             | 511 | -169             | 21                | -086| 28     |
| S27* | Y1   | 625   | -170          | 12             | 550 | -182             | 34                | 274 | 36     |
| S28  | Y1   | 604   | -282          | 15             | 282 | -93              | 14                | 229 | 21     |
| S28  | Y2   | 484   | -358          | 12             | 282 | -93              | 14                | 032 | 19     |
| S39* | Y1   | 584   | -209          | 16             | 403 | -133             | 16                | 241 | 22     |
| S43  | Y1   | 436   | -130          | 5              | 351 | -116             | 12                | 190 | 13     |
| S43  | Y2   | 381   | -192          | 11             | 351 | -116             | 12                | 074 | 17     |
| S44  | Y1   | 513   | -173          | 7              | 445 | -147             | 10                | 193 | 12     |
| S44  | Y2   | 377   | -240          | 11             | 445 | -147             | 10                | -010| 14     |
| P23  | Y1   | 469   | -229          | 9              | 231 | -76              | 9                 | 164 | 12     |
| P23  | Y2   | 407   | -260          | 11             | 231 | -76              | 9                 | 070 | 14     |

S = Sitka spruce; P = lodgepole pine. Subscript number under the site code correspond to age of the plantation (in 2015). *One year measurements only due to unplanned clearfell in June 2015. ** Fine root turnover of 0.33 from Lane (2016). SE – standard error of the mean. Due to the unplanned clearfell, fine root biomass in S39 was estimated as the mean value of the other six Sitka spruce sites.
Table 5 Soil carbon emission factors for “Forest land, drained”, in the maritime temperate climatic zone. Tier 1 values from the Wetlands supplement and Tier 2 values developed for Ireland (IE) and the United Kingdom (UK) with 95% confidence intervals (CI)

| IPCC Tier 1 Category | Source       | CO$_2$-C (t ha$^{-1}$ year$^{-1}$) | 95% CI (t ha$^{-1}$ year$^{-1}$) |
|----------------------|--------------|-----------------------------------|----------------------------------|
|                       | S18          | 3.09                              | 1.78                             | 4.39                             |
|                       | S24          | 0.63                              | -1.17                            | 2.43                             |
|                       | S27          | 2.74                              | 2.07                             | 3.41                             |
|                       | S28          | 1.30                              | 0.15                             | 2.46                             |
|                       | S39          | 2.41                              | 2.11                             | 2.72                             |
|                       | S43          | 1.32                              | 0.56                             | 2.08                             |
|                       | S44          | 0.92                              | -0.23                            | 2.06                             |
|                       | P23          | 1.07                              | 0.30                             | 1.85                             |
|                       | This paper   | 1.68                              | 1.04                             | 2.32                             |
|                       | IE           | 0.59                              | n.d.                             | n.d.                             |
|                       | NIR          |                                    |                                  |                                  |
|                       | IPCC 2014 Tier 1 | 2.60                      | 2.00                             | 3.30                             |
|                       | UK           | 2.00                              | 1.70                             | 2.30                             |
|                       | IPCC 2014 Tier 2 |                                      |                                  |                                  |

Ireland (IE) Tier 2 values from the National Inventory Reports (NIR) from Duffy et al. (2021), IPCC Tier 1 from IPCC (2014), UK Tier 2 values from Evans et al. (2017). n.d. not determined. S – Sitka spruce; P – lodgepole pine. 

Subscript number under the site code correspond to age of the plantation (in 2015).
Table 6 Annual CO₂ emissions from afforested peatlands in Ireland (IE), the United Kingdom (UK) and each UK administration calculated using Tier 1 emission factors from the IPCC (2014) Wetlands Supplement (2.60 t CO₂-C year⁻¹), UK Tier 2 values from Evans et al. (2017) Emissions Inventory for UK Peatlands (2.00 t CO₂-C year⁻¹), current IE Tier 2 values from Duffy et al. (2021) Ireland National Inventory Report (0.59 t CO₂-C year⁻¹) and the new emissions factor derived in this study (1.68 t CO₂-C year⁻¹). Area of afforested peatlands for IE and UK are taken from NFI (2018) and Evans et al. (2017), respectively.

| Country | Area (ha) | Soil C emissions (t CO₂-C year⁻¹) | IPCC Tier 1 | UK Tier 2 | NIR | This study |
|---------|-----------|----------------------------------|-------------|-----------|-----|-----------|
| IE      | 260730    | 677898                           | 521460      | 153831    | 438026 |
| England | 65492     | 170279                           | 130984      | 38640     | 110027 |
| Isle of Man | 118     | 307                               | 236         | 70        | 198   |
| N. Ireland | 31534   | 81988                            | 63068       | 18605     | 52977 |
| Scotland | 332746    | 865140                           | 665492      | 196320    | 559013 |
| Wales   | 9520      | 24752                            | 19040       | 5617      | 15994 |
| UK      | 439410    | 1142466                          | 878820      | 259252    | 738209 |

NIR: National Inventory Reports  
NFI: Ireland’s National Forest Inventory
Monthly aboveground litterfall (dry biomass in g m\(^{-2}\) month\(^{-1}\))

- S18
- S24
- S27
- S28
- S39
- S43
- S44
- P23

Month:
- June 14
- Aug 14
- Dec 14
- Feb 15
- Apr 15
- Jun 15
- Aug 15
- Oct 15
- Dec 15
- Feb 16
- Apr 16
- Jun 16

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Annual soil C fluxes (g C m$^{-2}$ year$^{-1}$)

Site

SCB  Litter$_{AG}$  Litter$_{BG}$  $R_H$

S18 Y1  S18 Y2  S24 Y1  S24 Y2  S27 Y1  S28 Y1  S28 Y2  S39 Y1  S43 Y1  S43 Y2  S44 Y1  S44 Y2  P23 Y1  P23 Y2

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