Forest Ecology and Management 482 (2021) 118861

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Carbon and water balance of an afforested shallow drained peatland in Iceland

Brynhildur Bjarnadottir [a,*], Guler Aslan Sungur [b,c], Bjarni D. Sigurdsson [d], Bjarki T. Kjartansson [e], Hlynur Oskarsson [d], Edda S. Oddsdottir [e], Gunnhildur E. Gunnarsdottir [f], Andrew Black [g]

[a] University of Akureyri, IS-600 Akureyri, Iceland
[b] Munzur University, Tunceli 62000, Turkey
[c] Iowa State University Ames, IA, USA
[d] Agricultural University of Iceland, Hvanneyri, IS-311 Borgarnes, Iceland
[e] Icelandic Forest Research, Mogilsa, IS-116 Reykjavik, Iceland
[f] Soil Conservation Service of Iceland, IS-851 Hella, Iceland
[g] University of British Columbia, Vancouver, Canada

ARTICLE INFO

Keywords:
Carbon cycle
Land-use change
Populus trichocarpa
Mitigation
Drained wetland

ABSTRACT

Drainage of peatlands increases the depth of the oxic peat layer and can turn them into a carbon (C) source to the atmosphere. Afforestation of drained peatlands could help to reverse this process since the trees may enhance C sequestration. We followed the C and water dynamics of an afforested drained peatland in S-Iceland during a 2 year period, during which the Black Cottonwood (*Populus balsamifera* ssp. *trichocarpa*) plantation was 23–25 year old. Net ecosystem exchange (NEE) of carbon dioxide (CO_2) was measured with the eddy covariance method and C pools of trees and ground vegetation were measured using the stock change method. Lateral losses of dissolved and particulated organic C (DOC, POC) were estimated from weekly water-runoff samples. Unexpectedly, the afforested drained peatland was a strong sink of carbon during the two years, with an average NEE value of 714 g C m^{-2} yr^{-1}. Only 0.5% of the total NEE was lost through lateral DOC and POC transport, leaving 710 g C m^{-2} yr^{-1} as the total net ecosystem production (NEP). Ca. 91% of the observed NEP could be explained by the annual biomass increment of the Black Cottonwood trees and 1.3% by the ground vegetation. This means that the remaining 7.5% of the total NEP most likely accumulated in peat soil and litter, contributing to the soil C stocks. The dormant-season CO_2 emissions were unexpectedly low, which was explained by a high groundwater level at this drained site outside the ca. 5 months of the active growing season. On average, 66% of the annual measured precipitation was estimated to have evaporated back to the atmosphere. This left 416 mm for potential runoff, which was somewhat lower value than the measured runoff (662 mm). These results indicate that during the age span of ca. 20–25 years, afforestation was a valid method to reverse the expected negative C-balance of this drained grassland pasture in Iceland. Although the site is currently a soil C sink, simulation studies with process models are needed to test whether such sites could remain C sinks when managed for forestry over several tree-stand rotations.

1. Introduction

Globally, peatlands are one of the most important carbon (C) stores comprising at least 550 Gt of C, which equals to 30% of all global soil organic carbon (SOC) or 75% of all atmospheric C (Gorham, 1991). This number is substantial given that peatlands cover only approximately 3% of the worlds land area. In the context of climate change, peatlands are hence ecosystems of significant importance (FAO, 2012; IPCC, 2014). In addition to their carbon sequestration properties, peatlands also provide various other ecosystem services, including regulating services of water quality and quantity, and support services in terms of biodiversity, soil formation and nutrient cycling (Gorham, 1991; Millennium Ecosystem Assessment, 2005; Parish et al., 2008).

In N-Europe, where human population densities are relatively high, up to 90% of peatlands have been cleared, drained or degraded which has led to a massive increase in net emissions of greenhouse gases

* Corresponding author.
E-mail address: brynhildurb@unak.is (B. Bjarnadottir).

https://doi.org/10.1016/j.foreco.2020.118861
Received 25 September 2020; Received in revised form 7 December 2020; Accepted 8 December 2020
Available online 23 December 2020

(GHG), even comparable to global industrial emissions in those countries (Parish et al., 2008). Small changes in wetland hydrology can lead to large changes in GHG emissions due to its influence on peatland biogeochemistry. Thus, proper understanding of the GHG response of northern peatlands to hydrological alterations requires detailed and process-based knowledge, gathered from both descriptive studies of the GHG exchange as well as manipulation experiments.

In their undisturbed natural stage, peatlands normally act as sinks of atmospheric CO_2 and sources of CH_4, but with drainage this is reversed (Gorham, 1991; Lund et al., 2010; Waddinton and Price, 2013; Bubier et al., 2005). Drainage lowers the soil water table, which increases aeration of the peat and significantly enhances organic matter oxidation with subsequent increased CO_2 emissions to the atmosphere (Schipper & McLeod, 2002; Parish et al., 2008). At the same time, drained wetlands usually have much reduced or non-existent CH_4 release (Couwenberg et al., 2011), while emissions of the third GHG, N_2O, can in some instances become a significant part of the overall GHG emissions (He et al., 2016; Ernfors et al., 2008). Studies on changes in C-stocks in agricultural drained peat soils have shown that in general they decrease under efficient drainage (e.g. Oleszczuk et al., 2008; Maljanen et al., 2010; Tiemeyer et al., 2016). Such net losses of soil C have also been reported from various drained pasture sites in South and West Iceland (Gunnarsdottir, 2017).

It has been estimated that around 15 million hectares of peatlands and wetlands have been drained for forestry in the temperate and boreal regions and most of this drainage occurred between the 1960s and late 1990s (Paavilainen and Päivänen, 1995). For temperate and boreal soils drained for forestry, most studies report that the forest ecosystem is a net-sequester of C some decades after the afforestation took place (e.g. Hargreaves et al., 2003; Lohila et al., 2011; Turetsky et al., 2011; Minkkinen et al., 2018), while others have found, or simulated (e.g. He et al., 2016), that such forests can act as sources of C to the atmosphere (e.g. Lohila et al., 2007; Simola et al., 2012). The land-use category 'forestry drained peatlands' has a net CO_2 emission factor (EF) in IPCC's Wetland Supplement (IPCC, 2014; Barthelmes et al., 2015), but as stated by Jauhiainen et al. (2019) there is still a large uncertainty linked to those average EFs. They further stated that 95% confidence interval for the IPCC's Tier 1 CO_2-C EF for boreal nutrient-poor soils ranges from -0.23 removal to 0.73 t CO_2-C ha^{-1} yr^{-1} emissions.

The net-climate effect of land-use change is also affected by changes in the fluxes of the two other natural GHGs, methane (CH_4) and nitrous oxide (N_2O), and therefore the two other GHGs are preferably included in the GHG budget (usually as CO_2 eq.; Lohila et al. 2011). However, studies on the whole GHG flux balance of drained forested areas are still rare and Maljanen et al. (2010) emphasised the lack of knowledge on the CO_2 flux budget of drained peatland forests at higher latitudes where the two other GHGs were measured. Those CO_2-flux studies that do exist from the NE-Atlantic region include e.g. Lohila et al. (2011) and Minkkinen et al. (2018) who reported a natural Scots pine forest (*Pinus sylvestris*) on a drained boreal bog in Finland to be a relatively strong carbon sink (-1.4 to -2.7 t CO_2-C ha^{-1} yr^{-1}) during a 4-year period. Hommeltenberg et al. (2014) also reported that a drained 44-year-old Norway spruce (*Picea abies*) forest in Germany was a stronger sink than a natural bog forest (-1.3 to -3.0 t CO_2-C ha^{-1} yr^{-1}). Both studies fall outside the 95% confidence interval of the IPCC's EF as estimated by Jauhiainen et al. (2019).

Long-term changes in C-balance can be derived from stock-change studies (Jauhiainen et al., 2019). Hargreaves et al. (2003) reported on the sink strength of a chronosequence of Sitka spruce (*Picea sitchensis*) stands in Scotland (1, 2, 3, 4, 8, 9 and 26-year-old) and compared to undrained peatland. The sink strength was around -25 g C m^{-2} yr^{-1} (0.25 t CO_2-C ha^{-1} yr^{-1}). The drained sites in Scotland were found to be sources during the first four years following afforestation (2 to 4 t C ha^{-1} yr^{-1}), but after ground vegetation had colonized the disturbed surfaces (4–9 years later), the sites became sinks of approx. 3 t C ha^{-1} yr^{-1} and at 26 years the sink strength was ca. 5 t C ha^{-1} yr^{-1}. Again, much higher than the IPCC's EF. The authors concluded that such afforested drained peatlands in Scotland were C sinks for 90–190 years after drainage and afforestation. Similarly, Minkkinen and Laine (1998) found that the C stores had increased on average by 5.9 ± 14.4 kg C m^{-2} ca. 60 years after drainage of 273 peatland sites in Finland. On the other hand, He et al. (2016) reported a Norway spruce forest in Sweden to be a net source of CO_2 and N_2O when a full rotation was simulated. This, again, highlights the large uncertainty in the current regional and national drainage emission factors as stated by Jauhiainen et al. (2019) and therefore the need for more studies on these effects.

A limitation of most of the above flux studies is their focus on only the net ecosystem exchange (NEE). Such studies report measurements of vertical CO_2 fluxes to and from the drained ecosystem, but they do not include leaching losses or lateral transport of C from the ecosystem as dissolved organic carbon (DOC) and particulate organic carbon (POC) in the draining ditch water. When such lateral fluxes are included the C balance generally is defined as Net Ecosystem Production (NEP) (Chapin III et al., 2002). In general, DOC is a relatively small component of undisturbed peatland CO_2 fluxes, however since NEE of such sites is generally close to zero, it still could potentially have a large impact on the peatland NEP (Hyvönen et al., 2007). Dillon and Molot (1997) calculated that the aquatic export from temperate and boreal peatlands can range between 1 and 50 g C m^{-2} yr^{-1}, representing around 10% of the total C release. Studies from Finnish drained peatlands show leaching losses ranging between 10 and 15 g C m^{-2} yr^{-1} (Sarkkola et al., 2009; Rantakari et al., 2010). Minkkinen et al. (2018) reported that this was ca. 4–7% of estimated NEE of a boreal drained bog forest, which equals only ca. 1% of Ecosystem Respiration (R_{eco}) or Gross Primary Production (GPP) measured at the same site. This lateral flux has, however, rarely been included in NEE studies on drained afforested peatlands and estimates of the relative size of this flux are therefore still highly uncertain (Jauhiainen et al., 2019).

Studies on the effect of afforestation on ecosystem water balances have generally found it to decrease water run-off or ground-water levels (Karlsen et al., 2016; FAO, 2008), while no effects (Stoy et al., 2013) or even increased run-off has also been reported (Williams et al., 2012). Micro-meteorological measurements within boreal forest ecosystems in Finland at similar latitudes as in this study, with leaf area index (LAI) ranging between 0.2 and 4.6, have shown that the annual evapotranspiration is often in the range of 40–60% of the annual precipitation and that this fraction increases as the forest canopy (LAI) increases (Launiainen et al., 2019).

Within Iceland, peatlands are prominent landscape features and cover approximately 9% of the land area (Arnalds et al., 2016). Since the early 1940's approximately half of the Icelandic peatlands have been fully drained, mostly for agricultural purposes (croplands, hay-fields and livestock grazing). About 34.000 km of ditches have been excavated and the total area of fully drained wetlands in Iceland is over 4200 km^2 or 4% of the total land area (Oskarsson, 1998; Wöll et al., 2014; Arnalds et al., 2016). A substantial fraction of these drained wetlands has fallen out of use and does not serve as an agricultural area today (Arnalds et al., 2016). Given the overall size of the drained peatlands in Iceland and the country's relatively small population size and industries, this land-use category is currently estimated to be the single greatest source of greenhouse gases (GHG) to the atmosphere (Wöll et al., 2014). There is therefore an urgent need for practical options to limit or mitigate those GHG emissions from drained peatlands. Two main options being considered by Icelandic authorities are rewetting (restoring) drained wetlands or afforestation of drained wetlands, where rewetting is not feasible or possible (Iceland's Climate Action Plan for 2018–2030, 2018). To date, neither land-use change option has been much practiced in Iceland. Restored peatlands wetlands cover 0.1 km^2 and afforested drained wetlands cover 38 km^2 (National Inventory Report of Iceland, 2019). Lack of published data on the outcome of these two mitigating methods have prevented Icelandic authorities from acting so far.

The aim of this study was to estimate the annual C and water balances of a 23–25 year old deciduous forest plantation on a drained peatland in S-Iceland, using the eddy-covariance technique and hydrological, meteorological and inventory measurements. We hypothesised that the drained forest would be a net CO_2 source, due to relatively high decomposition fluxes from the drained peatland soils and because of the expected high amounts of C that would leave the ecosystem as DOC and POC through drainage ditches in the relatively wet climate.

2. Material and methods

2.1. Site description

The study was conducted in a planted Black Cottonwood *Populus balsamifera* L. ssp. *trichocarpa* ((Torr. & A. Gray ex Hook.) Brayshaw) forest stand on a thick (>5 m) shallow drained organic soil in Sandlækjarmýri in S-Iceland (64°2′3N, 20°22′2W, 73 m a.s.l), hereafter referred to as the WetWood site (Fig. 1). It is situated within a ca. 8 km^2 wetland complex, located on a flat plain north of the river ûjórsá. Prior to draining the site was fen peatland dominated by sedges (*Carex nigra*) without any *Sphagnum* moss in the field layer. The section of the area which the WetWood site is located on was originally drained in 1959 by excavating ca. 3-m-deep open ditches into the peatland, which drain to SW towards the ûjórsá river (Fig. 1). The distance between the ditches was 400 m, resulting in poor drainage of the area as a whole. To improve this, the draining of the site was improved by adding few perpendicular (SE-NW) drainage pipes at ca. 0.5 m depth and some additional open ditches in the same direction in 1970; one of which has its whole catchment area (10.6 ha) within the study site (Fig. 1). This was not enough to fully drain the site. Following drainage, the study area was used for sheep grazing during summers by the local farmer (ûrándur Ingvarsson, farmer, pers. comm.). In 1989 the WetWood area became part of an 85-ha experimental afforestation project (IS: Iðnviðarverkefnið) for developing sustainable wood-chip production for the Icelandic ferrosilicon industry (Mikaelsson, 2011), and it is today the largest continuous Black Cottonwood plantation in Iceland. In 1990, 31 years after initial drainage, the WetWood site's grassland was ploughed, with a simple plough that removed a 20 cm deep and 0.7 m wide plough string and left it upside down at the side of the furrow. This was done at ca. 1.8-m intervals for site preparation and consequently the site was manually planted in 1991–1993 with containerized Black Cottonwood plants propagated from cuttings; mainly using the 'Iðunn' clone, but also smaller areas with the clones 'Halla' and 'Salka' (Ævarsson, 2007). No further forest management operations have since then taken place within the study area.

Micro-topographically the site is flat. The climate of the study site is classified as most parts of Icelands climate, maritime cool temperate to sub-arctic (Olafsson et al., 2007; Einarsson, 1984). Winters are relatively mild and summers rather cold. During 2005–2015, the mean annual air temperature and the mean temperatures of the warmest (Jul) and the coldest (Jan) months were 4.3 °C, 11.9 °C and −1.1 °C, respectively, at the closest official weather station and the mean annual precipitation during the same period was 1242 mm, rather evenly distributed throughout the year (Icelandic Met. Office, 2020). The prevailing wind directions are southwest and northeast. The top 15 cm of the drained peatland soil (Histic Andosol), including both furrows and plough strings, had 36.1% organic matter content, the bulk density was 0.37 g cm^{-3}, C:N ratio was 16.1 and pH was 6.2, which is quite typical for Icelandic wetlands in southern and western Iceland (Óskarsson et al., 2004; Arnalds et al., 2016). The forest floor has extensive moss cover of mainly *Hylocomium splendens*. Outside the scarified furrows and forest openings, the total vegetation cover was ca. 85%, with vascular plant cover of 48%, where the *Filipendula ulmaria, Deschampsia cespitosa, Agrostis capillaris* and *Equisetum pratense* were the most common species, in that order. Forest openings and access roads are fully covered by mostly *D. cespitosa, A. capillaris, E. pratense* and *Carex nigra*, in that order. The site is similar in its vegetation characteristics to other undrained and drained sites in the same area, where the soil C-balance has been determined by stock-change approach (Gunnarsdottir, 2017), which

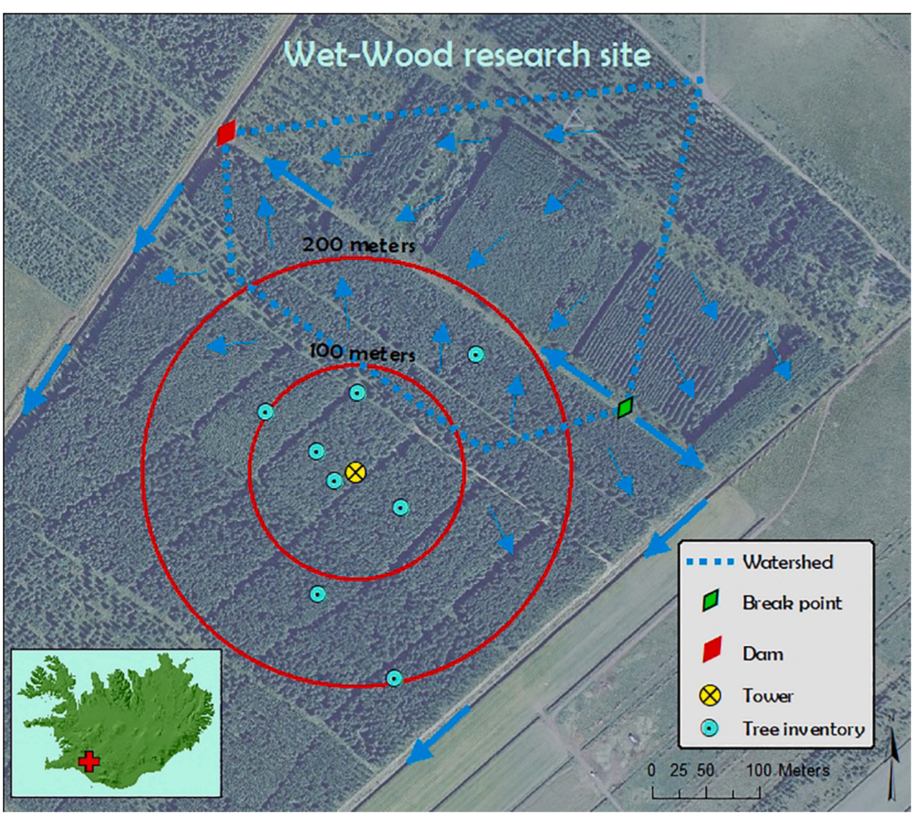


Fig. 1. An overview of the WetWood study site in S-Iceland. Yellow dot indicates the position of the EC flux tower and red circles represent 100 and 200 m radius from the EC tower. Blue dots show the eight tree inventory plots. Blue arrows show the direction of the draining water in the draining ditches and within the drainage area to the perpendicular draining ditch which contained the weir (red dot) and crossed a natural water divider (green dot), indicated by the direction of the water flow in it. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

showed a net C loss when converted to pastures.

2.2. Tree biomass and forest floor

Tree inventories of standing biomass were conducted for years 2014–2017. Measurements were done on eight 100–200 m^2 circular permanent plots within 200 m distance from the EC tower (Fig. 1). Standing wood volume and tree biomass were calculated with volume and biomass equations for Black Cottonwood in Iceland from Snorrason and Einarsson (2006). The current annual increment of aboveground and belowground woody biomass and leaf biomass were also calculated from the tree inventory data (see Supplement Tab. S1).

Harvest measurements on ground vegetation were done in late August 2016, the time of maximum standing biomass. Ground vegetation was collected from ten randomly chosen 20 × 50 cm plots, divided between three habitat types within the forest: (scarified surfaces (n = 3), undisturbed surfaces (n = 4) and forest openings (n = 3)). All living plants < 0.5 m in height were harvested and standing biomass and annual aboveground litter production was calculated from its biomass ratios for different functional types (see Supplement for further info).

2.3. Flux tower measurements

Eddy-covariance (EC) measurement of CO_2, latent heat, sensible heat fluxes and friction velocity (u_*) at the WetWood experimental site, started in October 2014 and continued until October 2016. The EC system was an open-path CO_2/water vapour infrared gas analyzer (LI-7500, LI-COR Inc., Lincoln NE, USA) and a 3-D sonic anemometer (Solent R3, Gill Instruments Ltd., Lymington Hampshire, UK). The sensors were mounted at 12-m height on a mast located near the center of the plantation (Fig. 1). The EC system measured fluxes at a sampling rate of 20 Hz. Fluxes were block-averaged over 30-min periods with the EcoFlux software (In Situ Flux Systems AB, Ockelbo, Sweden). Given that the site was micro-topographically flat, the EC footprint was concentrated to the homogenous peatland forest with at least 200 m radius (Fig. 1) and on average the mean hight of the canopy was 2-m below the height of the sensor during the measurement time (Kljun et al., 2004).

Basic meteorological data, such as air temperature, soil temperature, precipitation, net and global radiation, and soil heat flux were collected at the site with Campbell CR10X data logger (Campbell Scientific Ltd.) (sums or averages over 30-min intervals). Soil heat flux was measured using 2 soil heat flux plates (HFP01 SC Hukseflux thermal sensors) installed at 5 cm depth and corrected for heat storage change in the 0–5 cm layer. Technical problems occurred several times during the study period, which led to data loss, mainly of weather data. In such cases, data from the Árnes synoptic station, 6 km away, were used. A detailed description of the meteorological instrumentation can be found in Bjarnadottir et al. (2007).

2.4. EC data processing

Because of a technical failure in the LI-7500 IRGA in measuring the water vapour mole fraction during the measurement period, the latent heat flux (*LE*) was calculated as the residual in the energy balance of the plantation (Amiro, 2009):

$$LE = R_n - G - H \quad (1)$$

where R_n is the net radiation (W m^{-2}), G is the soil heat flux (W m^{-2}) and H is the sensible heat flux (W m^{-2}). Using EC-measured H (W m^{-2}) and the calculated values of LE (W m^{-2}), 30-min values of measured CO_2 flux (F_{CO2}) were corrected using Eq. (45b) in Webb et al. (1980). This is known as the Webb-Pearman and Leauning (WPL) correction which corrects for the density effects due to heat and water vapour transfer (Monteith and Unsworth, 2013). During suppressed turbulence or reduced mixing at night (low u_*), part of the CO_2 produced by ecosystem respiration may accumulate near the surface and be advected away from the tower below the measurement height. This may result in EC-measured F_{CO2} underestimating the actual exchange of CO_2 between the plantation and the atmosphere. Therefore, we checked for a threshold value of u_* to ensure adequate mixing using nighttime F_{CO2} values. However, our data did not give a particular threshold value for u_* which is likely due to the relatively high wind speed during the year at the site (average over the year 5.45 m s^{-1}).

Net ecosystem exchange (NEE) was calculated as F_{CO2} + the rate CO_2 storage change in the air column below the EC sensors. The storage term was calculated from the concentration data measured at the 12-m height. In most cases the storage term was found to be negligible, also because of the high wind speed (*t*-test; p > 0.5).

Further screening and quality checking of the data was done by applying a filtering procedure to the flux data to remove spikes and outliers (Thomas et al., 2011; Liu et al., 2012; Aslan-Sungur et al., 2016). This was done according to the following equation:

$$IX_{ij} - X_j I > 3\sigma_j - > spike \quad (2)$$

where $X_{i,j}$ is the half-hourly value of quantity X_j at time i, X_j is the mean value during a selected time window, and σ_j is the standard deviation of quantity, X_j. The time window in this study was three weeks centred at observation.

The observation of wintertime CO_2 uptake has led researchers to question the reliability of the LI-7500 IRGA in cold air conditions (Grelle and Burba, 2007; Burba et al., 2006). The problem is due to heat generated by the open-path analyser in cold conditions leading to a sensible heat flux inside the open-path array which affects the CO_2 density (Burba et al., 2008; Bonneville et al., 2008). Burba et al. (2006) pointed out that temperatures inside the open-path array were correlated with wind speed such that at higher wind speeds, heat produced by the instrument was more effectively removed from the open-path array so the difference between air temperature and surface temperature of the instrument was reduced. They found that for all temperatures, winds exceeding 6–8 m s^{-1} reduced the surface temperature of the detector housing of the LI-7500 to less than 1°C above ambient. Brown et al. (2010) detected a threshold related to windspeed and negative NEE values during the non-growing season in a lodgdepole pine stand in B.C. in Canada. They discarded all values below the threshold value (4 m s^{-1}) resulting in 56% of the non-growing season data remaining. Therefore, we plotted the fraction of negative NEE values during non-growing season against wind speed in order to see if we could detect a threshold in the wind speed. No such threshold was detected in the data, again likely because of the high wind speeds at the site (data not shown). After the total cleaning procedure, 27% of the data were discarded.

NEE was partitioned between ecosystem respiration (R_e) and gross primary production (GPP). Nighttime R_e was assumed to equal nighttime NEE and daytime R_e was estimated using the relationship between nighttime NEE and soil temperature. GPP was then calculated using:

$$GPP = R_e - NEE \quad (3)$$

Gap filling of missing data and further partitioning of fluxes, was done using the gap-filling and partitioning package developed by Max Planck Institute for Biogeochemistry, using R-studio software (REddyProc package; https://www.bgc-jena.mpg.de/bgi/index.php/Services/REddyProcWeb). The gap filling tool uses look-up tables to look for similar conditions in radiation, temperature and/or VPD between periods of time where data are available to gap-fill missing data. The tool also considers the covariation of fluxes with meteorological variables and the temporal autocorrelation of the fluxes (Reichstein et al., 2005; Lasslop et al., 2010; Wutzler et al., 2018).

Evapotranspiration (ET) was estimated from calculated values of *LE* (using Eq. (1)), with the following equation:

$$ET = LE_{av}/L_c \tag{4}$$

where LE_{av} is the average value of LE (W m^{-2}) over 24 h and L_c is the factor (28.94 W m^{-2}/mm day^{-1}) for converting daily average LE *to* ET in mm day^{-1}.

2.5. Ground water level

Ground water level was monitored every 30 min by two automatic HOBO U20 Water Level Loggers (Onset Inc., Bourne, MA), one installed at the bottom of a 2-m-deep well installed close to the EC tower and the other 50 cm above the surface for atmospheric pressure correction.

2.6. Runoff water discharge

A plywood weir was installed in Oct 2014 in the perpendicular ditch which has its whole catchment area (10.6 ha) within the study site (Fig. 1), creating a small dam. A perforated 1.5 m long and 15 cm wide tube was fixed to the ditch bottom with two 3 m long metallic posts, 1 m into the dam, with a pair of HOBO U20 Water Level Loggers to monitor its water level every 30 min. Runoff was calculated from this data with a rating curve which was constructed for the ditch channel (see Supplement, Eq. S2).

2.7. Runoff water chemistry and NEP

From Oct. 2014 until Nov. 2015 weekly water samples (5 L) were taken from the weir and filtered with a vacuum pump through a 0.45 µm pore size glass filter to divide between particulate organic carbon (POC) and dissolved organic carbon (DOC). The concentrations of POC and DOC were related to the measured runoff at the time of sampling and those relationships, together with the 30 min runoff data (Eq. S2) were used to calculate both seasonal and annual flux of POC and DOC through the ditch channels (see Supplement for further information about the methods used). When TOC has been subtracted from NEE the remaining annual carbon balance was termed as NEP (Chapin III et al., 2002).

3. Results

3.1. Meteorological conditions

The mean annual air temperature during the two study years was 3.7 and 4.6 °C, respectively (Fig. 2). The daily mean temperature ranged from −12 °C in December to 17 °C in August, both years. The total precipitation during these two years was 1237 mm and 1396 mm (Fig. 2), with the second year being both warmer and wetter than the former year. Most of the precipitation fell during the autumn months and the summers were the driest periods. Daily mean soil water table (WT) level was reflective of the precipitation and fluctuated between +5 and –85 cm in both years. It showed a clear annual trend reaching its greatest depth at the end of September in both years and rose again after rainfalls in October. As for the former year, the WT level fluctuated around the soil surface from January-April indicating that the area was very wet during these months. However, during the second year, the WT level was deeper during most of the winter as a result of somewhat lower precipitation.

3.2. Stand characteristics and production

The average stand density, stand basal area, DBH and dominant height were 1431 trees ha^{-1}, 26.6 m^2 ha^{-1}, 15.3 cm and 11.5 m, respectively, in autumn 2014 (see supplement Tab. S1). Standing wood volume and mean annual increment in 2015 were 139 m^3 ha^{-1} and 6.1 m^3 ha^{-1} yr^{-1}. No mortality occurred in the forest stand during the study period and measured stand BA increased by 2.0 and 3.0 m^2 ha^{-1} in 2015 and 2016, respectively. The total aboveground and belowground wood

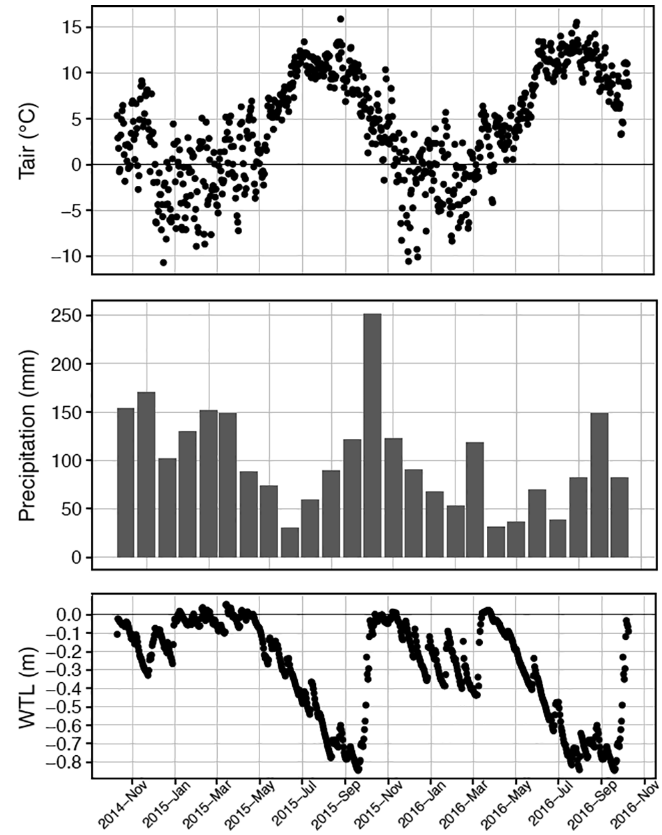


Fig. 2. Weather data: Mean daily temperature (°C), precipitation (mm/month) and average water table level (m) at the WetWood research site during the two year measurement period (Oct 2014-Oct 2016).

increment was 407 and 595 g C m^{-2} yr^{-1}, where the stem biomass increment accounted for 54.4% of the total tree woody increment, branches for 18.4%, stumps for 5.4%, coarse roots for 16.0% and fine roots for 5.8% (Table 1). Leaf production was 144 and 149 g C m^{-2} yr^{-1} for 2015 and 2016, respectively. An average annual total tree woody and leaf biomass production for the two years when the eddy covariance measurements took place (2015 and 2016) was 647 g C m^{-2} yr^{-1} (see Supplement Tab. S1).

Table 1
Annual C balance of the WetWood site. Carbon fluxes (NEE, R_{eco}, GPP, TOC, POC, DOC) and net annual change in biomass in the peatland ecosystem during the two measurement years (g C m^{-2} yr^{-1}). Leaves and ground vegetation contribute to the annual soil C balance.

	1st year*	2nd year**	Average
NEE	**−614**	**−814**	**−714**
GPP	−1183	−1387	−1285
R_{eco}	569	573	571
TOC	**3.86**	**–**	**–**
POC	1.02	–	–
DOC	2.84	–	–
Tree biomass	**−551**	**−744**	**−648**
Stem	−221	−324	−273
Branches	−75	−110	−93
	−144	−149	−147
Leaves	–22	–32	–27
Stump	−65	−94	–80
Coarse roots	−24	−35	−30
Fine roots***			
Ground vegetation		**−9**	

* 1st year is 10 Oct 2014 to 09 Oct 2015.
** 2nd year is 10 Oct 2015 to 09 Oct 2016.
*** Not including the potential turnover of fine roots within the year.

The annual C production of the ground vegetation was measured 3.6, 45.4 and 20.9 g DM m^{-2} in scarified areas, undisturbed areas and in openings in the forest after scaling with 25%, 65% and 10% areal distribution within the study area, respectively (data not shown). This equals to only 9 g C m^{-2} annual aboveground production in ground vegetation, due to the high contribution of perennial moss cover (see Supplement).

3.3. Net ecosystem CO_2 exchange (NEE)

The EC flux measurements indicated that the site was a CO_2 sink during the two years (Fig. 3 and Table 1). NEE for the first year was −2253 g CO_2 m^{-2} yr^{-1} (614 g C m^{-2} yr^{-1}) and for the second year is was −2987 g CO_2 m^{-2} yr^{-1} (814 g C m^{-2} yr^{-1}). The average value for the two years was 2620 g CO_2 m^{-2} yr^{-1}, corresponding to 714 g C m^{-2} yr^{-1} or 714 t C ha^{-1} yr^{-1}. The site showed a typical seasonal CO_2 flux pattern. It acted as a source of CO_2 during the non-growing season, from October until early April, with ecosystem respiration (R_{eco}) being the main driving factor. R_{eco} did not differ much between the two years (569 g C m^{-2} yr^{-1} and 573 g C m^{-2} yr^{-1}, respectively or on average 571 g C m^{-2} yr^{-1}) ($p > 0.5$) (Fig. 7 and Table 1). The system shifted from being a CO_2 source to a sink in mid-April and showed net CO_2 uptake until the end of September, both years. GPP was the main driving factor during the growing season and showed high values, indicating a high CO_2 assimilation by the ecosystem. The GPP was 1183 g C m^{-2} yr^{-1} and 1387 g C m^{-2} yr^{-1}, in the first and second years, respectively, with an average of 1285 g C m^{-2} yr^{-1} (Fig. 7 and Table 1). GPP was significantly higher ($p < 0.0001$) in the later year than in the former year and was the main driving factor of NEE during both years (Table 1). The onset and the end of the active growing season at the site was defined as the time when daily NEE remained negative (spring) or positive (autumn) for at least 6 consecutive days. During spring NEE started to fluctuate around 0 as early as in the beginning of April but did not show constant uptake values until the end of April (Fig. 3). However, this cannot have been a result of a tree canopy photosynthesis, since the leaf flush of the black cottonwood was observed to take place in middle of May in both years (data not shown). There was almost no difference in the length of the active growing season between the two years, or only a 4-day difference, i.e. 161 and 165 days, respectively. On average, 83% and 72% of the annual GPP and R_{eco} took place during the growing season.

3.4. Water runoff and evaporation

Here, 'runoff' or total discharge (D) is defined as all water that is leached from the catchment area and left it through the drainage ditch. The ditch water runoff during Oct 2014 – Oct 2015 showed a substantial seasonal variability and ranged between 0.03 and 23.1 L sec^{-1} and between 0.12 and 1.5 million litres day^{-1} (Fig. 4). The runoff was at minimum during summer but increased during autumn and winter with increasing precipitation and soil water levels. Alternating freeze–thaw periods occurred repeatedly through the winter in Iceland's oceanic climate, which led to the observed runoff tops. The annual runoff from Dec 2014 to Nov 2015 was 70.1 million litres, which equals 662 mm leaving the whole catchment area, or 48% of the measured precipitation at the site (Fig. 7). The estimated annual evaporation during the two-year period was 821 and 803 mm for each year (Fig. 5) or 66% and 57% of measured precipitation (Fig. 5). Of the total evaporation for the former year, 58% of it took place during the growing season while this proportion was 51% for the second year. In Fig. 7 the water balance for one year (the former year) is shown. Measured runoff turned out to be 662 mm yr^{-1}, while estimated runoff based on the difference in precipitation and evaporation, assuming no net storage, gave a value of 416 mm yr^{-1}.

3.5. Lateral carbon fluxes (DOC and POC)

The DOC concentration in the ditch runoff water was stable and low (<5 mg l^{-1}) during winter but rose abruptly to 15–20 mg l^{-1} in May and June when the site's soil water level dropped. It receded again to 5–8 mg l^{-1} in July and maintained that concentration during the late summer and into the autumn. POC, on the other hand, had the lowest concentrations during winter, ca. 1 mg l^{-1}, but gradually rose during summer and into the autumn, reaching 5–6 mg l^{-1} during July-September (see Supplement Fig. S1). Electric conductivity of drainage water ranged between 107 and 223 µS cm^{-1}, with an average of 157 µS cm^{-1}, and was generally highest in the summer (see Supplement Fig. S1). The pH of the drainage water was highest during the first two weeks after the water table was raised in the dam, whereas it followed a rather regular seasonal curve thereafter (see Supplement Fig. S1). No such "initial wetting signal" was observed for the other chemical parameters.

The DOC concentration did not vary significantly with discharge nor pH, but electric conductivity (*EC;*) decreased linearly with increasing runoff ($P < 0.0001$, $r^2 = 0.35$):

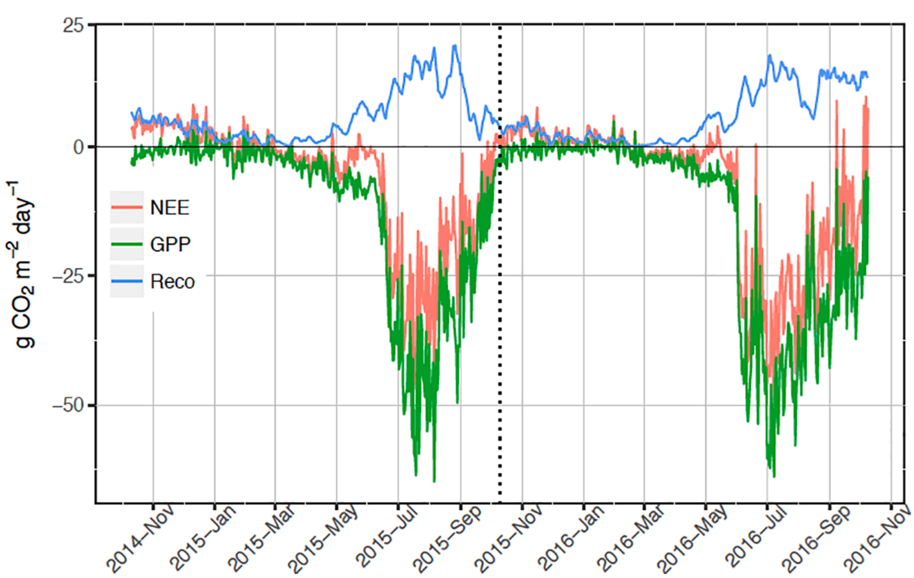


Fig. 3. Gap filled and partioned daily NEE, GPP and R_{eco} at the WetWood research site during the two year measurement period.

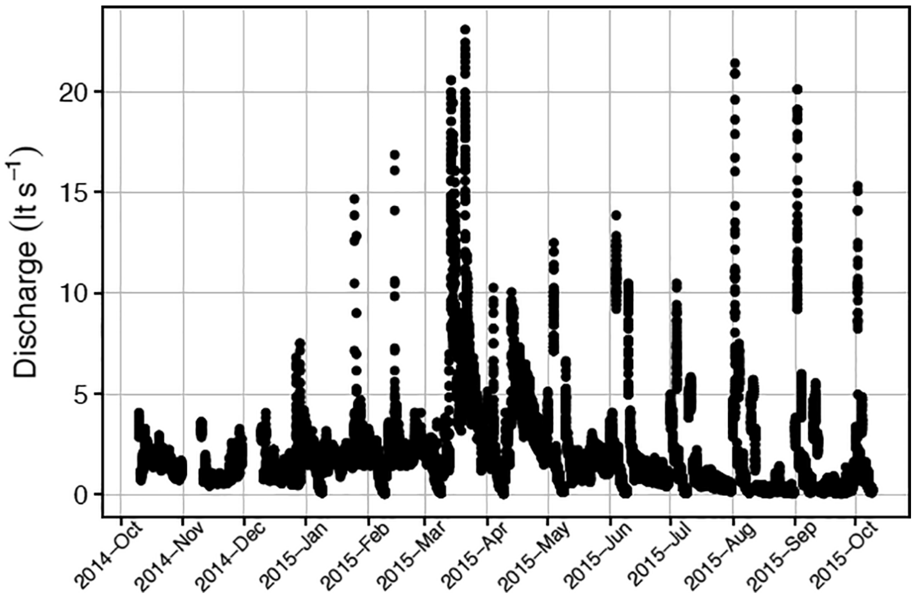


Fig. 4. Water discharge every 30 min during Oct 2014 to Oct 2015 through the weir in a draining ditch at the WetWood site that collected water from a 10.6 ha catchment area.

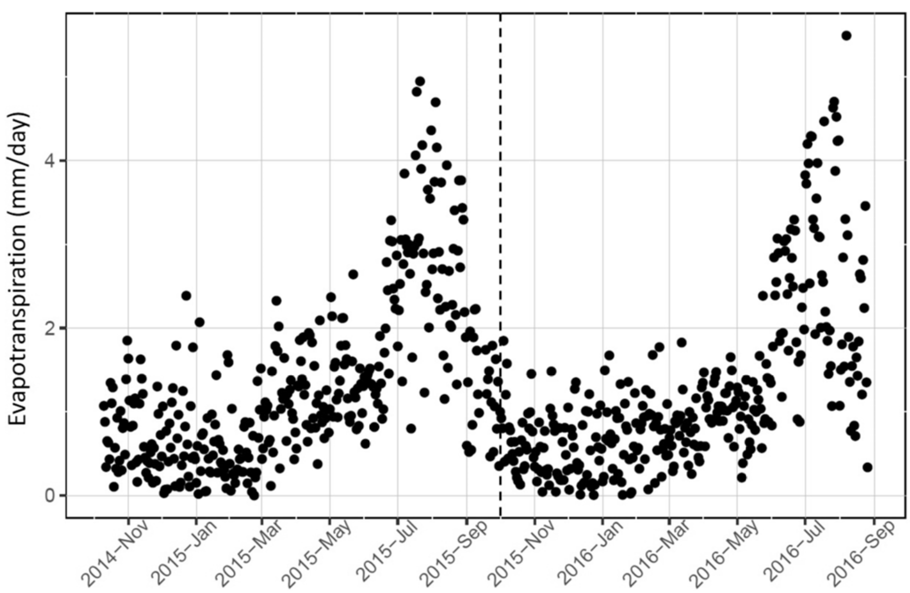


Fig. 5. Daily evapotranspiration (mm/day) during the two-year study period (Oct 2014-Oct 2016) at the WetWood site.

$$EC = 176.8 - 10.818 \times D \tag{5}$$

where D is the discharge in l 30-min^{-1}. The POC concentration (mg l^{-1}) also decreased in a curvilinear way with increasing runoff (D) until it reached a minimum concentration at D of 4000 l 30-min^{-1} ($r^2 = 0.64$, p < 0.0001):

$$POC = 1.38 \times 10^{-07} \times D^2 - 0.0016 \times D + 4.8685 \tag{6}$$

At the peak discharge episodes, with D $>$ 4000 l 30-min^{-1}; which summed up to ca. 250.000 l day^{-1}) the POC relationship with D became positively linear ($r^2 = 0.41$, p < 0.0001):

$$POC = 4.8685 + 0.0001 \times D \tag{7}$$

Daily DOC transport was calculated based on the weekly measured DOC concentrations and the D passing through the weir for each measurement interval, and an average DOC concentration from each two measurement was used for each interval (running average of two samplings). This was justified since DOC did not change significantly with D, pH or EC. Daily POC transport was, on the other hand, estimated using the 30-min D data and estimated POC concentrations at given D with the relationships shown in Eq. (6) and Eq. (7) The results are shown in Fig. 6, where total organic carbon (TOC) transport is the sum of DOC and POC.

Daily DOC transport ranged between 63 and 4130 g C day^{-1} with lowest transport values during late summer, but highest values in the autumn and late winter to early summer (Fig. 6). The total annual DOC transport from the catchment amounted to 316 kg C yr^{-1}. Daily POC values ranged between 24 and 2216 g C day^{-1}, with lowest transport values from Jun to Sep, but a sharp increase at leaf fall in Oct-Nov and also during winter thaws and late winter and spring thawing (Fig. 6). The total annual POC transport from the catchment amounted to 94 kg C yr^{-1}, or on average 26% of the TOC transport.

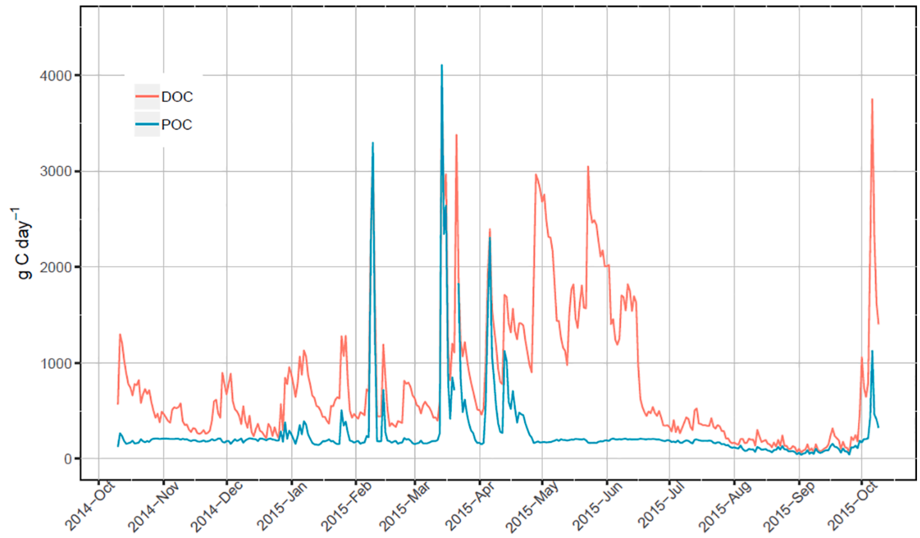


Fig. 6. Daily transport of DOC and POC (g C day^{-1}) during a one year period (Oct 2014-Oct 2015) through the weir in a draining ditch at the WetWood site that collected water from a 10.6 ha catchment area.

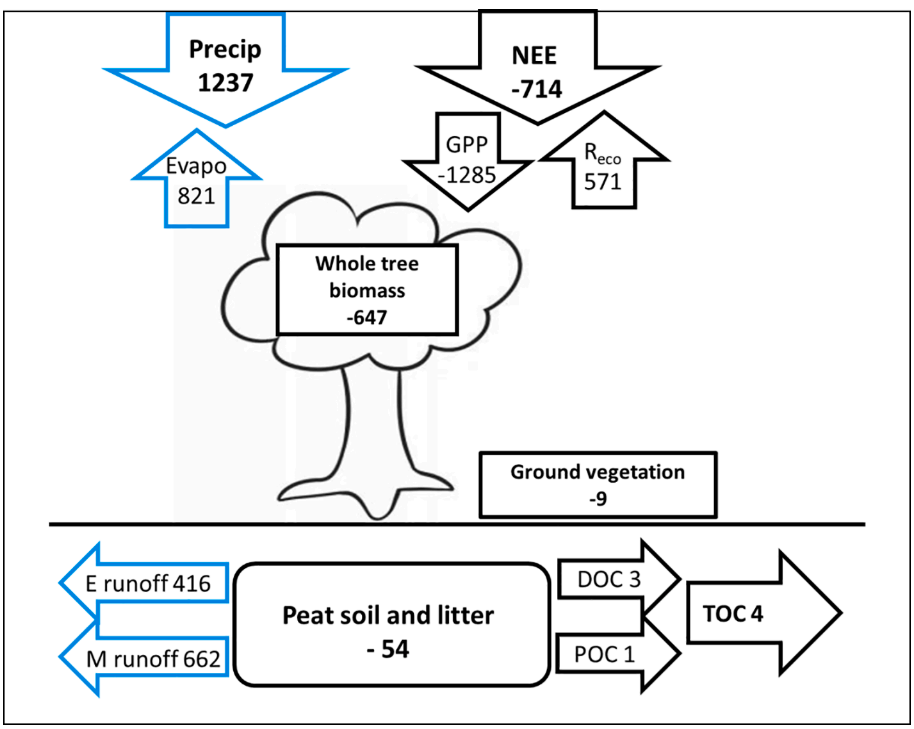


Fig. 7. Average annual C (black boxes) and water (blue boxes) balances at the WetWood research site. Measured C fluxes (g C m^{-2} yr^{-1}) and C stocks of trees and ground vegetation (g C m^{-2} yr^{-1}) are shown on the right side (black boxes), while the water balance is shown on the left side (blue boxes). *E* runoff represents the estimated runoff (mm yr^{-1}) while *M* runoff represents the measured runoff (mm yr^{-1}). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

3.6. Carbon and water balances

The mean annual measured carbon balance of the ecosystem is shown in Fig. 7, right side. The observed NEE:GPP ratio was 56%. On average, the ecosystem assimilated 1285 g C m^{-2} yr^{-1} and lost 571 g C m^{-2} yr^{-1} through R_{eco}, leaving 714 g C m^{-2} yr^{-1} as NEE, of which 4 g C m^{-2} yr^{-1} were lost through lateral aquatic transport. The lateral C loss was relatively small, or only 0.5% of the NEE. The net ecosystem production (NEP) was therefore 710 g C m^{-2} year^{-1}, of which 91% could be explained by the annual biomass increment of the trees (whereof aboveground biomass was 72% and belowground was 19%, respectively) and 1.3% could be explained by the aboveground annual production of ground vegetation. The difference between NEP and biomass sequestration therefore amounted to 54 g C m^{-2}, which most likely accumulated in fine roots, leaf and root litter and soil stocks.

The water balance for the former year is shown in Fig. 7, left side. On average, 66% of the annual measured precipitation was estimated to have evaporated back to the atmosphere, according to the flux estimates. This left 416 mm for potential runoff, which was somewhat lower value than the measured runoff (662 mm).

4. Discussion

4.1. Carbon balance of the drained afforestation area

Other studies in Iceland have reported that drainage of peatlands for pastures generally leads to losses in soil C-stocks of 0.7–8,25 t C ha yr^{-1} (Guðmundsson and Oskarsson, 2014; Olafsdottir, 2015; Gunnarsdottir, 2017), which is assumed to represent a change in the ecosystem soil C-balance because of the relatively small C-stocks in plants in such systems (e.g. Snorrason et al., 2002; Hyvönen et al., 2007). In this paper we have reported that the drained afforested peatland at the WetWood site in Iceland was a strong CO_2 sink during the two measurement years, equalling to 710 g C m^{-2} yr^{-1} NEP after deducting the lateral ditch water TOC loss from the NEE. This C-sequestration value is relatively high compared to other studies. Minkkinen et al. (2018) and Lohila et al. (2011) reported an average annual sink value of 230 g C m^{-2} yr^{-1} in a 60–180 year old natural Scots Pine forest on a peatland that had been drained in 1970 in Finland. In that study 73% of the total NEE was sequestered to the tree stand and 60 g C m^{-2} yr^{-1} accumulated in soil, which compared well with the ca. 55 g C m^{-2} yr^{-1} residual C-flux balance in the present study (Fig. 7). The higher ratio stored in woody biomass in the present study was not unexpected, because of its 2–3 times higher mean annual stemwood increment (MAI) compared to the older pine stand in Finland (Minkkinen et al., 2018).

Other studies have found both C sinks or sources in drained forest soils. Ojanen et al. (2013) reported an average of 63 g C m^{-2} yr^{-1} source and 14 g C m^{-2} yr^{-1} sink across 68 fertile and poor drained forest sites in Finland, respectively. Hommeltenberg et al. (2014) found that a 44 year old Norway spruce forest on a drained bog at a subalpine site in Germany was a CO_2 sink of 130–300 g C m^{-2} yr^{-1}. The C-sink values in the present study were more similar to the findings of Hargreaves et al. (2003) who reported a range in C-sink from 200 to 500 g C m^{-2} yr^{-1} for an afforested Sitka spruce drained peatland site of similar age (26 year old) as the Black Cottonwood measured at the WetWood site.

A strong CO_2-sink at a drained site can either stem from an unusually high C uptake rate (GPP) or an unusually low rate of C emission (R_{eco}), or both, and it is of interest to try to elucidate which process was governing the present findings. In the present study GPP was somewhat higher than in most comparable studies (cf. Ojanen et al., 2012). The tree species Black Cottonwood was introduced in Iceland around 1940 and has proven to be one of the country's fastest growing tree species with a mean annual increments (MAI) up to 10 m^3 ha^{-1} yr^{-1} in a ca. 30 year stands in southern Iceland (Bogason et al., 2018). The high GPP value found in the present study can be explained by relatively high MAI, which is among the highest measured in Iceland. Unselectively early start of GPP in the present study can be explained by moss photosynthesis before spring leaf flush and by bark photosynthesis of poplar trees during off season period (Aschan et al., 2001; Pfanz & Aschan, 2001). Young *Populus* sp. twigs can offset up to 80% of dark respiration values with such bark photosynthesis, which means that respiration during off season period can be cancelled out to some extent. This seems to be the case in our study as there was a clear sign of CO_2 uptake already in early April even though the trees did not flush until middle of May. Furthermore, the composition of the ground vegetation (high cover of mosses) may explain some of the off-season uptake (Schipperges & Gehrke, 1996).

The high observed sink value at the WetWood site was also partly due to unexpected low annual R_{eco} from the drained peatland site, even though the observed weather conditions were not extreme and did not deviate much from 30 year average climate conditions (Icelandic Met. Office, 2020). The observed R_{eco} was, however, similar as has been found in a young forest site on mineral soil in Iceland (Bjarnadottir et al., 2007). The water table level at the WetWood research site was relatively high, indicating a suppressed R_{eco} during off season periods. Many studies have shown a direct relationship between water table level and R_{eco} (e.g. Flanagan & Syed, 2011; Munir et al., 2017). In the mild oceanic climate that characterises Iceland, the annual variation in R_{eco} was mainly due to different winter conditions and the ecosystem in general was dominated by winter processes in the present study. The unusually high CO_2-sink values we observed in our study seemed therefore both to stem from having an unusually productive forest type growing on drained wetlands that had relatively low winter CO_2 emissions due to high off-season ground water levels; i.e. the site was in fact poorly drained.

The residual value (−54 g C m^{-2} yr^{-1}) for the annual litter accumulation (enhanced soil C sink in Fig. 7) is rather uncertain since it also accumulates all potential measurement- and scaling errors included in the measurements. It is also of similar size as the reported accuracy of eddy covariance estimates under good conditions (Baldocchi, 2003). Therefore, we can only claim that there seems not to be any detectable change in the C stock of the drained peat soils at this age of the forest and most of the observed accumulation was in woody biomass. Further studies are therefore needed to investigate the C stock-change of the peat soil during shorter and longer periods following the afforestation.

4.2. *Water balance of the drained afforestation area*

The measured annual runoff (662 mm) was 54% of the annual precipitation at a nearby climate station. This is a lower runoff ratio than has been measured in drained agricultural catchment in W-Iceland (83%; Thorsteinsson et al., 2019) or estimated from evapotranspiration fluxes from young, open Black Cottonwood plantation in S-Iceland (70%; Sigurdsson et al., 2004). However, bearing in mind that the WetWood study site had dense forest cover with BA of ca. 30 and LAI of >3, this relatively low runoff ratio is not surprising and corresponds well to similarly dense forest sites in Scandinavia (Jansson et al., 1999) and probably indicated higher intercepted rainfall evaporation.

The evapotranspiration flux from the forest site was higher than would be expected from the measured precipitation-runoff balance, or the runoff was higher than the precipitation-evapotranspiration balance. Still we think that the difference, which amounted to 246 mm, is not excessive in the wet climate of the site. One should e.g. consider that ca. half of the catchment area that drained into the ditch extended into an area with less dense and more heterogeneous forest cover than was within 200 m found around the flux tower (Fig. 1). There, the intercepted rainfall would be considerably lower and therefore this could easily explain why the realized runoff was higher, when keeping in mind the observed difference in runoff-ratio in open or closed forest stands (Sigurdsson et al., 2004).

It has recently been pointed out that even if some flux balance studies have reported DOC and POC concentrations in draining ditch water (cf. Evans et al., 2016), very few such studies include total water discharge that is needed for converting those concentrations to C-flux estimates for a specific forest area/catchment area (Jauhiainen et al., 2019). The present study is therefore an important contribution in that sense. The total DOC + POC transport of ca. 3 + 1 g m^{-2} yr^{-1} was less than we had hypothesised and hence the importance of those lateral C-fluxes in the drained site C-balance was far less important than anticipated. In the present study the lateral loss of C through leaching and runoff was less than1% of the total NEE. When compared to the average DOC fluxes of ca. 10–20 g m^{-2} yr^{-1} DOC from near-natural wetlands in areas with similar precipitation within the Boreal zone (Evans et al., 2016), the low values we found are not unrealistic, and the low TOC:NEE ratio is also a function of the relatively high measured NEE of this forest. In Iceland, no other studies have reported DOC and POC annual fluxes from drained wetlands; but Bjarnadóttir (2012) found that during the growing season, drainage ditches from agricultural area in western Iceland transported on average 28.3 and 9.3 mg C m^{-2} day^{-1} for DOC and POC, respectively. It is noteworthy that the ratio between the DOC:POC is the same as found in the annual fluxes in the present study, but the actual amounts for the same time period in the present study were 9.0 and 1.6 mg C m^{-2} day^{-1}, when we convert the catchment fluxes in Fig. 6 to m^2-values. The relatively higher observed values in the Bjarnadóttir (2012) study could have been caused by water from undrained wetlands that also had some discharge into ditches.

4.3. *GHG inventories and how to interpolate over time*

The results of this project raise several compelling questions. Will the

WetWood peatland remain a CO_2 sink, and has it been a strong CO_2 sink ever since the area was planted with Black Cottonwood in 1989? It is evident that many boreal and temperate peatland forest ecosystems, where drainage has been successful, act as contemporary C sinks (Ojanen et al., 2013; Meyer et al., 2013; Minkkinen et al., 2018) because the tree stand C sequestration exceeds the loss of C from soil. However, in peatlands used for forestry it is the soil C storage that is important in the long-term, given that the tree stock will eventually be harvested and the C in wood products will eventually be lost back to the atmosphere. Hommeltenberg et al. (2014) and He et al. (2016) both emphasised the need to look at the whole life cycle of a planted forest when reporting on long term balance of afforestation on drained wetland soils. We agree with their suggestion. The present study only considered the C-balance during two years of a relatively young Black Cottonwood plantation, without including both the establishment phase, possible thinnings and the end of the forest rotation, all of which could have relatively less strong sink values. The most relevant question is therefore: Will sites like WetWood remain C sinks in the long-term if they are managed for forestry? After harvesting, soil decomposition processes will go on with logging residues enhancing the decomposition of the underlying peat soil (Mäkiranta et al., 2012) creating a loss of soil C through soil respiration of a certain amount depending on soil quality (Minkkinen et al., 2007, 2018). On the other hand, after harvesting the water table level is likely to rise because of the removal of the transpiring tree stand, which will reduce the peat decomposition rate (Mäkiranta et al. 2010). This reduction is, however, probably quite small in well drained sites and therefore the site is likely to be a strong C source at least for the first five years after harvesting (Korkiakoski et al., 2019; Kolari et al., 2004), or until sequestration by growing vegetation again reaches the point of outweighing respiratory C losses. There is therefore an urgent need for further studies and simulations on the total climate effect of long-term management of drained peatlands.

When reporting on the total climate effect of mitigation options it is important to report on all three types of greenhouse gases. In this study, we have only reported on the CO_2 flux in the ecosystem but left out both CH_4 and N_2O. Comparable studies show that CH_4 fluxes from forested drained peatlands are normally low and negative (i.e. CH_4 is taken up by the soil) and the flux shows little temporal and seasonal variation (e.g. Lohila et al., 2011; Meyer et al., 2013; Ojanen et al., 2013; He et al., 2016; Minkkinen et al., 2018). Reported numbers range from −0.06 to $-2.0\ g\ m^{-2}\ yr^{-1}$, making most studies state that the contribution of CH_4 to the GHG budget is negligible. However, it should be noted that given the high water table level of the site during the non-growing season (Fig. 2), it is likely that the site is a net CH_4 emitter during these months. These fluxes are, on the other hand, likely to be quite low given the low soil temperature during the non-growing season. Measurements of N_2O emissions in comparable ecosystems typically show a net source of N_2O ranging from 0.1 to 2.2 $g\ N_2O\ m^{-2}\ yr^{-1}$ (Lohila et al., 2011; Meyer et al., 2013; He et al., 2016). No comparable studies on afforested drained peatlands exist in Iceland. An Icelandic study conducted in a drained peatland used for grazing in W-Iceland showed that on annual bases the CH_4 emission contributed 8% of the total C efflux (Olafsdottir, 2015).

5. Conclusions

The present study gives insight into the current carbon and water balance of a 55–57 year old poorly drained peatland site in S-Iceland that has been afforested over the last 22–25 years. The study site was a strong sink of carbon during the two years, with an average NEE value of 714 $g\ C\ m^{-2}\ yr^{-1}$. Only 0.5% of the total NEE was lost through lateral TOC transport, leaving 710 $g\ C\ m^{-2}\ yr^{-1}$ as the total NEP. Around 91% of the observed NEP could be explained by the annual biomass increment of the Black Cottonwood trees and 1.3% by the ground vegetation, leaving 7,5% that most likely accumulated in leaf, fine-root litter and soil C stocks. A limitation to the ecosystem C balance (NEP) is that it only covers CO_2 and TOC but not CH_4. On average, 66% of the annual measured precipitation was estimated to have evaporated back to the atmosphere. This left 416 mm for potential runoff, which was somewhat lower value than the measured runoff (662 mm).

We therefore conclude that at a certain age span, afforestation seems to be a valid method to reverse the expected negative C-balance of poorly drained pastures in Iceland that have been abandoned, but where conditions or landowners will not allow wetland restoration.

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.

Acknowledgement

This research was supported by the Energy Research fund of Landsvirkjun, the National Power Company of Iceland, with an additional support from the Iceland State Electricity. It also contributes to the Nordic CAR-ES project (Centre of Advanced Research on Environmental Services from Nordic Forest Ecosystems) and to the SNS 120 program (Nordic Forest Research on Anthropogenic greenhouse gas emissions from organic forest soils: improved inventories and implications for sustainable management).

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.foreco.2020.118861.

References

Amiro, B., 2009. Measuring boreal forest evapotranspiration using the energy balance residual. J. Hydrol. 366, 112–118. https://doi.org/10.1016/j.jhydrol.2008.12.021.

Arnalds, O., Gudmundsson, J., Oskarsson, H., Brink, S.H., Gisladottir, F.O., 2016. Icelandic Inland Wetlands: characteristics and extent of draining. Wetlands 36 (4), 759–769. https://doi.org/10.5194/bg-7-2711-2010.

Aschan, G.G., Wittmann, C.C., Pfanz, H.H., 2001. Age-dependent bark photosynthesis of aspen twigs. Trees 15 (7), 431–437. https://doi.org/10.1007/s004680100120.

Aslan-Sungur, G., Lee, X., Evrendilek, F., Karakaya, N., 2016. Large interannual variability in net ecosystem carbon dioxide exchange of a disturbed temperate peatland. Sci. Total Environ. 554–555, 192–202. https://doi.org/10.1016/j.scitotenv.2016.02.153.

Baldocchi, D.D., 2003. Assessing the eddy covariance technique for evaluating carbon dioxide exchange rates of ecosystems: past, present and future. Global Change Biol. 9, 479–492. https://doi.org/10.1046/j.1365-2486.2003.00629.x.

Barthelmes, A., Couwenberg, J., Risager, M., Tegetmeyer, C., & Joosten, H., 2015. Peatlands and Climate in a Ramsar context: A Nordic-Baltic Perspective. https://doi.org/10.6027/TN2015-544.

Bjarnadottir, B., Sigurdsson, B.D., Lindroth, A., 2007. Estimate of annual carbon balance of a young Siberian Larch (*Larix Sibirica*) plantation in Iceland. Tellus B 59 (5), 891–899. https://doi.org/10.1111/j.1600-0889.2007.00300.x.

Bjarnadóttir, S.L., 2012. Short term study of carbon and nitrogen export from a drained Icelandic peatland and first effects of rewetting (M.Sc thesis). Technical University of Denmark, Copenhagen, p. 55 p..

Bogason, J.A., Jónsson, J.Á., Sigurðsson, B.D., 2018. Er framleiðsla iðnviðar með skammlotuskógrækt raunhæf hérlendis? [Is short rotation forestry possible in Iceland? Real case study]. Skógræktarritið 2018 (2), 34–46 (In Icelandic).

Bonneville, M.C., Strachan, I.B., Humphreys, E.R., Roulet, N.T., 2008. Net ecosystem CO2 exchange in a temperate cattail mars hin relation to biophysical properties. Agric. For. Meteorol. 148, 69–81. https://doi.org/10.1016/j.agrformet.2007.09.004.

Brown, M., Black, T.A., Nesic, Z., Foord, V.N., Spittlehouse, D.L., Fredeen, N.J., Burton, P.J., Trofymow, J.A., 2010. Impact of mountain pine beetle on the net ecosystem production of lodgepole pine stands in British Columbia. Agricult. Fores. Meterol. 150, 254–264. https://doi.org/10.1016/j.agrformet.2009.11.008.

Bubier, J.L., Moore, T., Savage, K., Crill, P., 2005. A comparison of methane flux in a boreal landscape between a dry and a wet year. Global Biogeochem. Cycles 19 (GB1023). https://doi.org/10.1029/2004GB002351.

Burba, G., Andersson, D.J., Xu, L., McDermitt, D.K., 2006. Correcting apparent off-season CO_2 uptake due to surface heating of an open path gas analyzer: progress report of an ongoing study. In: In 27th Conference on Agricultural and Forest Meteorology, P4–4. American Meteorological Society, San Diego, CA, p. 12 pp..

Burba, G., Mcdermitt, D., Grelle, A., Anderson, D., Xu, L., 2008. Addressing the influence of instrument surface heat exchange on the measurements of CO2 flux from open-

path analyzers. Glob. Change Biol. 14, 1854–1876. https://doi.org/10.1111/j.1365-2486.2008.01606.x.
Chapin III, S.F., Matson, P.A. & Mooney, H.A., 2002. Principles of Terrestrial Ecosystem Ecology. Springer, New York. https://doi.org/10.1007/978-1-4419-9504-9.
Couwenberg, J., Thiele, A., Tannenberger, F., Augustin, J., Bärisch, S., Dubovi, D., Liashchynskaya, N., Michaelis, D., Minke, M., Skuratovich, A., Joosten, H., 2011. Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. Hydrobiologia 674, 67–89. https://doi.org/10.1007/s10750-011-0729-x.
Dillon, P.J., Molot, L.A., 1997. Effect of landscape form on export of dissolved organic carbon, iron, and phosphorus from forested stream catchments. Water Resour. Res. 33 (11), 2591–2600. https://doi.org/10.1029/97WR01921.
Einarsson, M.A., 1984. Climate of Iceland. In: Loon, Van (Ed.), World survey of climatology, vol 15. Elsevier, Amsterdam, pp. 673–697.
Evans, C.D., Renou-Wilson, F., Strack, M., 2016. The role of waterborne carbon in the greenhouse gas balance of drained and re-wetted peatlands. Aquat Sci 78, 573–590. https://doi.org/10.1007/s00027-015-0447-y.
Ernfors, M., von Arnold, K., Stendahl, J., Olsson, M., Klemedtsson, L., 2008. Nitrous oxide emissions from drained organic forest soils - an up-scaling based on C: N ratios. Biogeochemistry 89, 29–41. https://doi.org/10.1007/s10533-008-9190-y.
FAO, 2008. Forests and water. In FAO Foresty Paper 155, p. 92, Rome.
FAO, 2012. Peatlands - Guidance for climate change mitigation by conservation, rehabilitation and sustainable use, Rome, Food and Agriculture Organization of the United Nations. Eds: Joosten H., Tapio-Biström M. L., Tol S., available at: http://www.fao.org/docrep/015/an762e/an762e.pdf.
Flanagan, L.B., Syed, K.H., 2011. Stimulation of both photosynthesis and respiration in response to warmer and drier conditions in a boreal peatland ecosystem. Glob. Change Biol. 17, 2271–2287. https://doi.org/10.1111/j.1365-2486.2010.02378.x.
Gorham, E., 1991. Northern peatlands: role in the carbon cycle and probable responses to climatic warming. Ecol. Appl. 1 (2), 182–195. https://doi.org/10.2307/1941811.
Grelle, A., Burba, G., 2007. Fine-wire thermometer to connect CO2 fluxes by open-path analyzers for artificial density fluctuations. Agric. For. Meteorol. 147, 48–57. https://doi.org/10.1016/j.agrformet.2007.06.007.
Guðmundsson, J. & Oskarsson, H., 2014. Carbon dioxide emission from drained organic soils in West-Iceland. Proceedings from: Soil carbon sequestration for climate food security and ecosystem services, international conference Reykjavík, Iceland, May 26.-29. 2013. Agricultural University of Iceland & European Commission Joint Research Centre. SCS2013, pp. 155-159.
Gunnarsdottir, G.E.G., 2017. A novel approach to estimate carbon loss from drained peatlands in Iceland. Master thesis. Faculty of Life and Environmental Sciences 46.
Hargreaves, K.J., Milne, R., Cannell, M.G.R., 2003. Carbon balance of afforested peatland in Scotland. Forestry 76, 299–317. https://doi.org/10.1093/forestry/76.3.299.
He, H., Jansson, P.E., Svensson, M., Björklund, J., Tarvainen, L., Klemedtsson, L., Kasimir, Å., 2016. Forests on drained agricultural peatland are potentially large sources of greenhouse gases – insights from a full rotation period simulation. Biogeosciences 13, 2305–2318. https://doi.org/10.5194/bg-13-2305-2016.
Hommeltenberg, J., Schmid, H.P., Drösler, M., Werle, P., 2014. Can a bog drained for forestry be a stronger carbon sink than a natural bog forest? Biogeosciences 11, 3477–3493. https://doi.org/10.5194/bg-11-3477-2014.
Hyvönen, R., Ågren, G.I., Linder, S., Persson, T., Cotrufo, M.F., Ekblad, A., Freeman, M., Grelle, A., Janssens, I.A., Jarvis, P.G., Kellomäki, S., Lindroth, A., Loustau, D., Lundmark, T., Norby, R.J., Oren, R., Pilegaard, K., Ryan, M.G., Sigurdsson, B.D., Strömgren, M., Van Oijen, M., Wallin, G., 2007. The likely impact of elevated [CO_2], nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: a literature review. New Phytol. 171, 275–296. https://doi.org/10.1111/j.1469-8137.2007.01967.x.
Icelandic Meteorological Office, 2020. Weather observations. https://en.vedur.is/weather/observations/areas/south/ (accessed 15. September 2020).
Icelands Climate Action Plan for 2018-2030 – Summary, 2018. Ministry for the Environment and Natural Resources. Reykjavík, Iceland.
IPCC, 2014. 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M., and Troxler, T. G. (eds). Published: IPCC, Switzerland. 353 p., https://www.ipcc-nggip.iges.or.jp/public/wetlands/.
Jansson, P.E., Beier, C., Borg, G.Ch., Guðmundsson, J., Roos, J. & Ahonen, J., 1999. Application of water and heat model to forested sites. In: Nitrogen Processes in Arable and Forest Soils in the Nordic Countries - Field-scale Modelling and Experiments. Jansson P.-E., Persson T., Kätterer, T. (eds), Nordic Council of Ministers, Copenhagen. Tema Nord 560: 53-62.
Jauhiainen, J., Alm, J., Bjarnadottir, B., Callesen, I., Christiansen, J.R., Clarke, N., Dalsgaard, L., He, H., Jordan, S., Kazanavičiūtė, V., Klemedtsson, L., Lauren, A., Lazdins, A., Lehtonen, A., Lohila, A., Lupikis, A., Mander, Ü., Minkkinen, K., Kasimir, Å., Olsson, M., Ojanen, P., Óskarsson, H., Sigurdsson, B.D., Søgaard, G., Soosaar, K., Vesterdal, L., Laiho, R., 2019. Reviews and syntheses: Greenhouse gas exchange data from drained organic forest soils – a review of current approaches and recommendations for future research. Biogeosciences 16 (23), 4687–4703. https://doi.org/10.5194/bg-16-4687-2019.
Karlsen, R.H., Grabs, T., Bishop, K., Buffam, I., Laudon, H., Seibert, J., 2016. Landscape controls on spatiotemporal discharge variability in a boreal catchment. Water Resour Res. 52, 6541–6556. https://doi.org/10.1002/2016WR019186.
Kljun, N., Calanca, P., Rotach, M.W., Schmid, H.P., 2004. A simple parameterisation for flux footprint predictions. Boundary-Layer Meteorol. 112, 503–523. https://doi.org/10.1023/B:BOUN.0000030653.71031.96.
Kolari, P., Pumpanen, J., Rannik, Ü., Ilvesniemi, H., Hari, P., Berninger, F., 2004. Carbon balance of different aged Scots pine forests in Southern Finland. Glob. Change Biol. 10, 1106–1119. https://doi.org/10.1111/j.1529-8817.2003.00797.x.
Korkiakoski, M., Tuovinen, J.P., Penttilä, T., Sarkkola, S., Ojanen, P., Minkkinen, K., Rainne, J., Laurila, T., Lohila, A., 2019. Greenhouse gas and energy fluxes in a boreal peatland forest after clear-cutting. Biogeosciences 16, 3703–3723. https://doi.org/10.5194/bg-16-3703-2019.
Lasslop, G., Reichstein, M., Papale, D., Richardson, A.D., Arneth, A., Barr, A., Stoy, P., Wohlfahrt, G., 2010. Separation of net ecosystem exchange into assimilation and respiration using a light response curve approach: critical issues and global evaluation. Glob. Chang Biol. 16, 187–208. https://doi.org/10.1111/j.1365-2486.2009.02041.x.
Launiainen, S., Guan, M., Salmivaara, A., Kieloaho, A.-J., 2019. Modeling boreal forest evapotranspiration and water balance at stand and catchment scales: a spatial approach. Hydrol. Earth Syst. Sci. 23, 3457–3480. https://doi.org/10.5194/hess-23-3457-2019.
Liu, H.Z., Feng, J.W., Jarvi, L., Vesala, T., 2012. Four-year (2006–2009) eddy covariance measurements of CO2 flux over an urban area in Beijing. Atmos. Chem. Phys. 12, 7881–7892. https://doi.org/10.5194/acp-12-7881-2012.
Lohila, A., Laurila, T., Aro, L., Aurela, M., Tuovinen, J.-P., Laine, J., Kolari, P., Minkkinen, K., 2007. Carbon dioxide exchange above a 30-year-old Scots pine plantation established on organic soil cropland. Boreal Environ. Res. 12, 141–157.
Lohila, A., Minkkinen, K., Aurela, M., Tuovinen, J.-P., Penttilä, T., Laurila, T., 2011. Greenhouse gas flux measurements in a forestry-drained peatland indicate a large carbon sink. Biogeosciences 8, 3203–3218. https://doi.org/10.5194/bg-8-3203-2011.
Lund, M., Lafleur, P.M., Roulet, N.T., Lindroth, A., Christensen, T.R., Aurela, M., Chojnicki, B., Flanagan, L.B., Humphreys, E.R., Laurila, T., Oechel, W.C., Olejnik, J., Rinne, J., Schubert, P., Nilsson, M.B., 2010. Variability in exchange of CO_2 across 12 northern peatland and tundra sites. Glob. Change Biol. 16, 2436–2448. https://doi.org/10.1111/j.1365-2486.2009.02104.x.
Mäkiranta, P., Riutta, T., Penttilä, T., Minkkinen, K., 2010. Dynamics of net ecosystem CO_2 exchange and heterotrophic soil respiration following clearfelling in a drained peatland forest. Agr. Forest Meteorol. 150, 1585–1596. https://doi.org/10.1016/j.agrformet.2010.08.010.
Mäkiranta, P., Laiho, R., Penttilä, T., Minkkinen, K., 2012. The impact of logging residue on soil GHG fluxes in a drained peatland forest. Soil Biol. Biochem. 48, 1–9. https://doi.org/10.1016/j.soilbio.2012.01.005.
Maljanen, M., Sigurdsson, B.D., Guðmundsson, J., Óskarsson, H., Huttunen, J.T., Martikainen, P.J., 2010. Greenhouse gas balances of managed peatlands in the Nordic countries – present knowledge and gaps. Biogeosciences 7, 2711–2738. https://doi.org/10.5194/bg-7-2711-2010.
Meyer, A., Tarvainen, L., Nousratpour, A., Björk, R.G., Ernfors, M., Grelle, A., Kasimir Klemedtsson, Å., Lindroth, A., Räntfors, M., Rütting, T., Wallin, G., Weslien, P., Klemedtsson, L., 2013. A fertile peatland forest does not constitute a major greenhouse gas sink. Biogeosciences 10, 7739–7758. https://doi.org/10.5194/bg-10-7739-2013.
Mikaelsson, L., 2011. Productivity and biomass partitioning in 20-year Black Cottonwood at variable spacing. Graduate Thesis, Agricultural University of Iceland. 56, pp.
Millennium Ecosystem Assessment, 2005. Ecosystems and human well-being: wetlands and water. Synthesis. Washington, DC: Island Press, 68 pp.
Minkkinen, K., Laine, J., 1998. Long-term effect of forest drainage on the peat carbon stores of pine mires in Finland. Can. J. For. Res. 28, 1267–1275. https://doi.org/10.1139/x98-104.
Minkkinen, K., Laine, J., Shurpali, N., Mäkiranta, P., Alm, J., Penttilä, T., 2007. Heterotrophic soil respiration in forestry-drained peatlands. Boreal Environ. Res. 12, 115–126.
Minkkinen, K., Ojanen, P., Penttilä, T., Aurela, M., Laurila, T., Tuovinen, J.P., Lohila, A., 2018. Persistent carbon sink at a boreal drained bog forest. Biogeosciences 15, 3603–3624. https://doi.org/10.5194/bg-15-3603-2018.
Monteith, J.L., Unsworth, M.H., 2013. Principles of Environmental Physics, 4th ed. Academic Press, Oxford, UK.
Munir, T.M., Khadka, B., Xu, B., Strack, M., 2017. Partitioning forest-floor respiration into source based emissions in a boreal forested bog: Responses to experimental drought. Forests 8, 75. https://doi.org/10.3390/f8030075.
National Inventory report, Emissions of Greenhouse Gases in Iceland from 1990 to 2017, 2019. The Environment Agency of Iceland. Reykjavík, Iceland.
Ojanen, P., Minkkinen, K., Lohila, A., Badorek, T., Penttilä, T., 2012. Chamber measured soil respiration: a useful tool for estimating the carbon balance of peatland forest soils. Forest Ecol Manage. 277, 132–140. https://doi.org/10.1016/j.foreco.2012.04.027.
Ojanen, P., Minkkinen, K., Penttilä, T., 2013. The current greenhouse gas impact of forestry-drained boreal peatlands. Forest Ecol Manage. 289, 201–208. https://doi.org/10.1016/j.foreco.2012.10.008.
Olafsdottir, R., 2015. Carbon budget of a drained peatland in Western Iceland and initial effects of rewetting. Master thesis, Faculty of Environmental Sciences, Agricultural University of Iceland. Hvanneyri, Iceland. 51 pp.
Olafsson, H., Furger, M., Brümmer, B., 2007. The weather and climate of Iceland. Meteorol. Z. 16, 5–8.
Oleszczuk, R., Regina, K., Szajdak, H., Höper, H., Maryganova, V., 2008. Impacts of agricultural utilization of peat soils on the greenhouse gas balance. In: Strack, M. (Ed.), Peatlands and Climate Change. Jyväskylä, Finland, Int. Peat Soc., pp. 70–97
Oskarsson, H., 1998. Framræsla votlendis á Vesturlandi [Drained peatlands in Vest-Iceland]. In: Íslensk votlendi - verndun og nýting. Editor: Jón S. Ólafsson. Háskólaútgáfan. Pages 121-130. [In Icelandic; English summary].
Óskarsson, H., Arnalds, Ó., Gudmundsson, J., Gudbergsson, G., 2004. Organic carbon in Icelandic Andosols: geographical variation and impact of erosion. Catena 56, 225–238. https://doi.org/10.1016/j.catena.2003.10.013.

Paavilainen, E., Päivänen, J., 1995. Peatland Forestry, ecology and principles, Ecolsstud, vol. 111. Springer, New York.

Parish, F., Sirin, A., Charman, D., Joosten, H., Minayeve, T., Silvius, M., Stringer, L. (Eds), 2008. Assessment on Peatlands, Biodiversity and Climate Change: Main Report. Global Environmental Centre, Kuala Lumpur and Wetlands International, Wageningen.

Pfanz, H., Aschan, G., 2001. The Existence of Bark and Stem Photosynthesis in Woody Plants and Its Significance for the Overall Carbon Gain. An Eco-Physiological and Ecological Approach. In: Esser K., Lüttge U., Kadereit J.W., Beyschlag W. (eds) Progress in Botany. Progress in Botany, vol 62. Springer, Berlin, Heidelberg. https://doi.org/10.1007/978-3-642-56849-7_19.

Rantakari, M., Mattsson, T., Kortelainen, P., Piirainen, S., Finer, L., Ahtiainen, M., 2010. Organic and inorganic carbon concentrations and fluxes from managed and unmanaged boreal first-order catchments. Sci. Total Environ. 408, 1649–1658. https://doi.org/10.1016/j.scitotenv.2009.12.025.

Reichstein, M., Falge, E., Baldocchi, D., Papale, D., Aubinet, M., Berbigier, P., Bernhofer, C., Buchmann, N., Gilmanov, T., Granier, A., Grünwald, T., Havránková, K., Ilvesniemi, H., Janous, D., Knohl, A., Laurila, T., Lohila, A., Loustau, D., Matteucci, G., Meyers, T., Miglietta, F., Ourcival, J.M., Pumpanen, J., Rambal, S., Rotenberg, E., Sanz, M., Tenhunen, J., Seufert, G., Vaccari, F., Vesala, T., Yakir, D., Valentini, R., 2005. On the separation of net ecosystem exchange into assimilation and ecosystem respiration: review and improved algorithm. Glob. Chang. Biol. 11 (9), 1424–1439. https://doi.org/10.1111/j.1365-2486.2005.001002.x.

Sarkkola, S., Koivusalo, H., Lauren, A., Kortelainen, P., Mattsson, T., Palviainen, M., Piirainen, S., Starr, M., Finer, L., 2009. Trends in hydrometeorological conditions and stream water organic carbon in boreal forested catchments. Sci. Total Environ. 408, 92–101. https://doi.org/10.1016/j.scitotenv.2009.09.008.

Schipper, L.A., McLeod, M., 2002. Subsidence rates and carbon loss in peat soils following conversion to pasture in the Waikato Region, New Zealand. Soil Use Manage. 18, 91–93. https://doi.org/10.1111/j.1475-2743.2002.tb00225.x.

Schipperges, B., Gehrke, C., 1996. Photosynthetic characteristics of Subarctic mosses and lichens. Ecol. Bull. 45, 121–126.

Sigurdsson, B.D., Bjarnadottir, B., Strachan, I. B., Pálmason, F., 2004. Tilraunaskógurinn í Gunnarsholti II. Vatnið í skóginum [Gunnarsholt Experimental Forest II. Annual water balance and water quality]. Skógræktarritið, 2004(1), 55-64 [In Icelandic; English summary].

Simola, H., Pitkänen, A., Turunen, J., 2012. Carbon loss in drained forestry peatlands in Finland, estimated by re-sampling peatlands surveyed in the 1980s. Eur. J. Soil Sci. 63, 798–807. https://doi.org/10.1111/j.1365-2389.2012.01499.x.

Snorrason, A., Sigurdsson, B.D., Gudbergsson, G., Svavarsdottir, K., Jonsson, Th.H., 2002. Carbon sequestration in forest plantations in Iceland. Icelandic Agric. Sci. 15, 81–93.

Snorrason, A., Einarsson, S., 2006. Single-tree biomass and stem volume functions for eleven tree species used in Icelandic forestry. Icelandic Agric. Sci. 19, 15–24.

Stoy, P.C., Mauder, M., Foken, T., Marcolla, B., Boegh, E., Ibrom, A., Arain, M.A., Arneth, A., Aurela, M., Bernhofer, C., Cescatti, A., Dellwik, E., Duce, P., Gianelle, D., van Gorsel, E., Kiely, G., Knohl, A., Margolis, H., McCaughley, H., Merbold, L., Motagnani, L., Papale, D., Reichstein, M., Saunders, M., Serrano-Ortiz, P., Sottocornola, M., Spano, D., Vaccari, F., Varlagin, A., 2013. A data-driven analysis of energy balance closure across FLUXNET research sites: The role of landscape scale heterogeneity. Agr. Forest Meteorol. 171, 137–152. https://doi.org/10.1016/j.agrformet.2012.11.004.

Thomas, M.V., Malhi, Y., Fenn, K.M., Fisher, J.B., Morecroft, M.D., Lloyd, C.R., Taylor, M.E., McNeil, D.D., 2011. Carbon dioxide fluxes over an ancient broadleaved deciduous woodland in southern England. Biogeosciences 8, 1595–1613. https://doi.org/10.5194/bg-8-1595-2011.

Thorsteinsson, B., Jóhannesson, G.H., Thorlacius, A., Gudmundsson, T., 2019. Precipitation, runoff and nutrient losses from cultivated Histosols in western Iceland. Icelandic Agricultural Sciences 32, 61–74. https://doi.org/10.16886/IAS.2019.06.

Tiemeyer, B., Albiac Borraz, E., Augustin, J., Bechtold, M., Beetz, S., Beyer, C., Drösler, M., Ebli, M., Eickenscheidt, T., Fiedler, S., Förster, C., Freibauer, A., Giebels, M., Glatzel, S., Heinichen, J., Hoffmann, M., Höper, H., Jurasinski, G., Leiber-Sauheitl, K., Peichl-Brak, M., Roßkopf, N., Sommer, M., Zeitz, J., 2016. High emissions of greenhouse gases from grasslands on peat and other organic soils. Glob. Change Biol 22, 4134–4149. https://doi.org/10.1111/gcb.13303.

Turetsky, M.R., Donahue, W.F., Benscoter, B.W., 2011. Experimental drying intensifies burning and carbon losses in a northern peatland. Nat. Commun. 2, 514. https://doi.org/10.1038/ncomms1523.

Waddinton, J.M., Price, J.S., 2013. Effect of peatland drainage, harvesting, and restoration on atmospheric water and carbon exchange. Phys. Geogr. 21 (5), 433–451. https://doi.org/10.1080/02723646.2000.10642719.

Webb, E.K., Pearman, G.I., Leuning, R., 1980. Correction of flux measurements for density effects due to heat and water vapour transfer. Quart. J. Roy. Meteorol. Soc. 106, 85–100. https://doi.org/10.1002/qj.49710644707.

Williams, C.A., Reichstein, M., Buchmann, N., Baldocchi, D., Beer, C., Schwalm, C., Wohlfahrt, G., Hasler, N., Bernhofer, C., Foken, T., Papale, D., Schymanski, S., Schaefer, K., 2012. Climate and vegetation controls on the surface water balance: Synthesis of evapotranspiration measured across a global network of flux towers. Water Resour. Res. 48, W06523. https://doi.org/10.1029/2011WR011586.

Wutzler, T., Lucas-Moffat, A., Migliavacca, M., Knauer, J., Sickel, K., Šigut, L., Menzer, O., Reichstein, M., 2018. Basic and extensible post-processing of eddy covariance flux data with ReddyProc. Biogeosciences 15, 5015–5030. https://doi.org/10.5194/bg-15-5015-2018.

Wöll, C., Hallsdóttir, B. S., Guðmundsson, J., Snorrason, A., ûórsson, j., Jónsson, P. V. K., Andrésson, K. & Einarsson, S., 2014. Emissions of greenhouse gases in Iceland from 1990 to 2012. National Inventory Report 2014.

Ævarsson, F., 2007. Frostþol og skyldleiki alaskaapsar (Populus balsamifera ssp. trichocarpa) á Íslandi. [Frost tolerance and clones of Black cottonwood (Populus balsamifera ssp. trichocarpa) in Iceland. Master thesis, Faculty of Environmental Biology, University of Iceland. Reykjvík, Iceland. 87 pp. [In Icelandic].

Update

Forest Ecology and Management

Volume 483, Issue , 1 March 2021, Page

DOI: https://doi.org/10.1016/j.foreco.2021.118934

Forest Ecology and Management 483 (2021) 118934

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

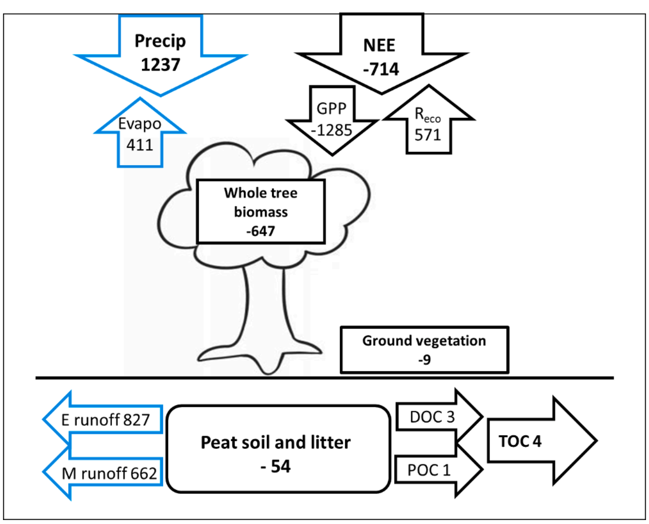
Corrigendum

Corrigendum to "Carbon and water balance of an afforested shallow drained peatland in Iceland" [For. Ecol. Manage. 482 (2021) 118861]

Brynhildur Bjarnadottir[a,*], Guler Aslan Sungur[b,c], Bjarni D. Sigurdsson[d], Bjarki T. Kjartansson[e], Hlynur Oskarsson[d], Edda S. Oddsdottir[e], Gunnhildur E. Gunnarsdottir[f], Andrew Black[g]

[a] University of Akureyri, IS-600 Akureyri, Iceland
[b] Munzur University, Tunceli 62000, Turkey
[c] Iowa State University, Ames, IA, USA
[d] Agricultural University of Iceland, Hvanneyri, IS-311 Borgarnes, Iceland
[e] Icelandic Forest Research, Mogilsa, IS-116 Reykjavik, Iceland
[f] Soil Conservation Service of Iceland, IS-851 Hella, Iceland
[g] University of British Columbia, Vancouver, Canada

The authors regret that the total evapotranspiration shown in Fig. 7 (821 mm) was the measured H_2O flux for the whole two year measurement period. The correct average value for the two years was 411 mm yr^{-1}, or 33% of the measured annual average precipitation. This left 827 mm yr^{-1} as the "estimated runoff" (see corrected figure). The sign of the average NEE values in the Abstract and Conclusions should be negative. All other information is correct, including the equations and interpretations. The authors would like to apologise for any inconvenience this has caused.



DOI of original article: https://doi.org/10.1016/j.foreco.2020.118861.
* Corresponding author.
E-mail address: brynhildurb@unak.is (B. Bjarnadottir).

https://doi.org/10.1016/j.foreco.2021.118934

Available online 16 January 2021