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Changes in carbon-stock and soil properties following afforestation in SW Iceland

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Author's Statement

I hereby declare that I have collected and processed research data and compiled this thesis myself with the help of instructors. The thesis has not been partially or fully submitted to a higher degree before.

Hvanneyri, 19 September 2019

Joel Charles Owona

Abstract

Afforestation does not only establish new forests on treeless lands, but also changes many other aspects of the ecosystem, including the fauna, ground vegetation and soil properties. One of the most important ecosystem changes is the influence on the ecosystem carbon (C) stocks in different aboveground and belowground C pools. If afforestation is to be used as a method to sequester atmospheric carbon dioxide (CO_2), to mitigate climate warming, it is important not only to consider changes in aboveground tree biomass, but also in the other four major ecosystem pools (dead wood, ground vegetation, litter layer and soil organic carbon (SOC)). The true CO_2 mitigating potential of afforestation is the net change in all those five pools, but especially the last two pools have often been neglected in prior studies.

Here I present a study of three afforestation sites in SW Iceland named Heiðmörk, Nesjavellir and Ölfusvatn forests. Heiðmörk is the largest site, planted with different coniferous tree species since ca. 1950 and also contains large naturally regenerated (self-seeded) areas of the native downy birch (*Betula pubesecens*). The Nesjavellir and Ölfusvatn forests are younger, since ca. 1997, and the former has planted stands of both birch or conifers and also some naturally regenerated areas self-seeded from local birch forest remnant, while the latter only has planted stands of either birch or conifers.

To examine the influences of afforestation: i) between sites, ii) between different forest types and iii) with respect to increasing forest age, the present study compared different ecosystem properties of adjacent treeless control sites with afforested areas. The ecosystem properties included: i) ground vegetation cover, composition and biomass, ii) soil physical properties (bulk density stoniness, soil and litter depths as well as soil and litter dry mass), iii) soil chemical properties (pH, SOC and N concentration in different soil layers, C/N ratio in both soils and litter) and iv) ecosystem C stocks (soils, litter, fine roots, ground vegetation and standing trees biomass). Another aim of the present study was to test if ecosystem C-stocks could be validated using minimum number of measurement plots in individual forests.

On average across all sites, forest types and forest ages, the soils of the forest sites had 12% larger SOC stocks compared to the treeless sites in 2017. Significant differences in the SOC stocks appeared mainly in the upper top soils (0-10 cm) depth. Litter C, necromass and

thickness were also found to be significantly higher in the afforested sites, while ground vegetation was significantly reduced, but these properties also differed between forest types and with age of the forest. Soil bulk density, pH and C/N ratio were found to remain similar across all sites and species. Soils under conifer tree species were not found to become acidic contrary to what was hypothesized.

On average, pure coniferous plots contained somewhat higher SOC stock (10,991 g C m^{-2}) than birch plots (10,340 g C m^{-2}) and the difference was even more pronounced for the litter layer, where the pure conifer stands had on average 92% larger litter C stocks than the native birch stands. Ground vegetation, on the other hand, was significantly reduced under conifers (-77%) while it remained under downy birch forests (+23%), but its C stock was far the smallest of the three and had only minor effect on the ecosystem C-balance. The annual rates of litter C accumulation were 22.0 and 4.9 g C m^{-2} yr^{-1} in the pure conifer and the native birch, respectively, and the observed average annual sequestration rates of SOC were 84.2 and 64.2 g C m^{-2} yr^{-1}, respectively. The changes in ground vegetation C-stock amounted to –3.5 and 1.3 g C m^{-2} yr^{-1}, respectively.

My results indicate that "general C-sequestration rate constants" should be avoided when forest owners are estimating the mitigation potential of their afforested lands. At least their sites should be classed into coniferous stands and birch stands and their C-sequestration should be estimated separately. Another finding is that the C-sequestration in the litter and topsoil layers may be of the same magnitude as the aboveground forest biomass during the first 20-40 years after afforestation. The present work provides valuable equations that enable forest owners to estimate the different C-pools in planted and self-seeded forest stands for different forest types in SW Iceland.

Keywords: ecosystem carbon stock; carbon sequestration; forest types; belowground

Útdráttur

[*Breytingar á kolefnisforða og öðrum jarvegsþáttum við nýskógrækt á SV-landi*]

Nýskógrækt klæðir ekki bara skóglaust land trjám, heldur hefur hún margvísleg önnur áhrif á vistkerfið og breytir dýralífi, gróðurfari og jarðvegsþáttum. Ein af mikilvægustu breytingunum sem gjarnan verða er uppsöfnun kolefnis (C) í mismunandi forða ofanjarðar og neðan. Ef nota á nýskógrækt sem mótvægisaðgerð á móti loftslagshlýnun með því að auka bindingu koldíoxíðs (CO_2) úr andrúmslofti, þá er mikilvægt að ekki bara taka tillit til breytinga á kolefnisforða í viði ofanjarðar, heldur einnig í hinum fjórum megin C-forðunum sem skógarvistkerfið inniheldur (botngróðri, dauðum viði, feyru og lífrænu kolefni í jarðvegi (SOC). Kolefnisbinding nýskógrækar er sú nettó-breyting sem verður í öllum þessum C-forðum yfir ákveðið tímabil, en sérstaklega síðast töldu tveir C-forðarnir hafa oft ekki verið teknir með í fyrri rannsóknum.

Ég kynni hér rannsóknir frá þremur skógræktarsvæðum á SV-landi: Heiðmörk, Nesjavöllum og Ölfusvatni. Heiðmörk er stærsta svæðið og þar hefur verið gróðursettur barrskógur af ýmsum tegundum síðan um 1950, en þar hefur einnig umtalsverður sjálfsáður birkiskógur (*Betula pubesecens*) vaxið upp af skógarleifum sem þar voru til staðar. Skógræktarsvæðin á Nesjavöllum og Ölfusvatni eru yngri, eða frá því um 1997, en á því fyrra vaxa bæði gróðursettir barr- og birkiskógar, auk sjálfsáinna birkiskóga af skógarleifum, á meðan á Ölfusvatni eru bara gróðursettir birki- og barrskógar.

Til að meta áhrif nýskógræktar: i) á mismunandi stöðum, ii) í mismunandi skógargerðum, og iii) með aldri ræktaðra skóga, á ýmsar mældar vistkerfisbreytur, þá voru einnig gerðar mælingar á nálægum skóglausum svæðum til samanburðar. Vistkerfisbreyturnar voru: i) yfirborðsþekja, samsetning og lífmassi botngróðurs, ii) eðliseiginleikar jarðvegs (rúmþyngd, hlutfall grófjarðar, jarðvegsdýpi og þykkt feyrulags, auk þurrvigtar jarðvegs í efstu 30 cm og feyru), iii) efnaeiginleikar jarðvegs (sýrustig, C/N hlutfall og magn SOC og N í mismunandi lögum jarðvegs og í feyru) og iv) C-forðar vistkerfisins (jarðvegur, feyra, fínrætur, botngróður og lífmassi trjáa). Annað markmið með rannsókninni var að prófa hvort hægt væri að staðfesta kolefnisbindingu í mismunandi kolefnisforðum með lágmarksfjölda mæliflata í stökum skógum.

Að meðaltali yfir öll svæði, skógargerðir og aldur reita þá höfðu skógarnir um 12% meiri kolefnisforða í jarðvegi árið 2017 miðað við skóglausu samanburðarsvæðin og aukningin var einkum í efsta lagi steinefnajarðvegsins (0-10 cm). Magn feyru, C-forði hennar og þykkt jukust einnig að jafnaði í kjölfar nýskógræktarinnar en það dró úr magni og C-forða botngróðurs, en munur var á þessu milli skógargerða og einnig með aldri skóga. Rúmþyngd, sýrustig og C/N hlutfall breyttist að jafnaði lítið eftir nýskógræktina. Jarðvegur undir barrskógum súrnaði ekki marktækt þegar allir skógar voru bornir saman, þvert á það sem búist var við.

Að jafnaði var meiri kolefnisforði í efstu 30 cm jarðvegs undir barrskógum (10.991 g C m^{-2}) en undir birkiskógum (10.340 g C m^{-2}) og munurinn var hlutfallslega enn meiri í feyrulaginu, þar sem barrskógarnir höfðu 92% meiri C-forða en birkiskógar. Botngróður minnkaði hinsvegar undir barrskógum (-77%) en breyttist ekki marktækt undir birkiskógum (+23%). Kolefnisforði botngróðurs var langminnstur af þessum þremur forðum og hafði ekki teljandi áhrif á kolefnisjöfnuð vistkerfisins.

Að jafnaði þá bættust 22,0 og 4,9 g C m^{-2} við feyrulag í barrskógum og í birkiskógum allra svæðanna á hverju ári og samsvarandi árleg kolefnisbinding í efstu 30 cm jarðvegs barr- og birkiskóga var 84,2 and 64,2 g C m^{-2}, að jafnaði. Breytingin á C-forða botngróðurs eftir nýskógræktina var að jafnaði aðeins –3,5 og +1,3 g C m^{-2} í barr- og birkiskógum.

Rannsóknir mínar sýna að mjög varasamt er að nota almenna bindistuðla þegar skógareigendur meta kolefnisbindingu sem verður í skógum þeirra. Að minnsta kosti þarf þá að flokka ræktaða skóga í mismundandi skógargerðir áður en slíkum stuðlum er beitt. Önnur athyglisverð niðurstaða er að á fyrstu áratugunum (20-40 árum) eftir að nýskógrækt hefst getur kolefnisbinding í jarðvegi og feyru verið af sömu stærðargráðu og sem verður í viðarvexti skóganna. Verkefnið leggur einnig skógareigendum á SV-landi til mikilvægar spájöfnur fyrir breytingar á mismunandi kolefnisforðum í skógum þeirra.

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1. Introduction

1.1. European forests and their C sequestration

The latest Global Forest Resources Assessment (FRA) report indicates that, on average, global forest cover decreased from 31.6% to 30.6% of the total land area between 1990 and 2015 (FAO, 2018). However, forest cover is not decreasing everywhere and the FRA report shows that e.g. in Europe, the forest cover increased during this period and it is expected to continue increasing. The increase is believed to be due to large-scale afforestation programmes as well as natural reversal of low-productive agricultural land back to forests (FAO, 2018). For example, in eastern Europe, the transformation from socialist to market economy has led to the abandonment of large formerly cultivated areas resulting in their reforestation (MacDonald et al., 2000; Vilén et al., 2016). This was further supported by the State of Europe's Forests Report, which shows that forested land is more than one third of the continent's surface area and it continues to increase steadily amounting to 215 million ha which accounts for 33% of the total land area of Europe (Forest Europe, 2015). European Environmental Agency (2016) ranked northern Europe as the most forested region (53%), while South-East Europe is the least forested region (23%). The Forest Europe (2015) report further indicates that the total growing stock of European forests amounts to 35 million m^3 with an average density of 163 m^3 ha^{-1}, which is larger than the world average (133 m^3 ha^{-1}).

The increased re-growth of forests in Europe during the last 50 years has, on average, added ca. 1.75 petagram (Pg) C yr^{-1} to the standing biomass carbon (C) stock and an increase in the net primary production (NPP) of 1.67 Pg C yr^{-1} in 57-year period (Ciais et al., 2019). Roberto et al. (2017) also confirm such a large forest biomass sink when they observe an average NPP increment of 0.64 Pg C yr^{-1} from 2000 to 2012 on the continent. i.e. ca. 40% of the 50 years NPP value that Ciais et al. (2019) reported in only 12 years. Using the Ciais et al. (2019) standing biomass values it may be estimated that between 1950 and 2007, ca. 2.3 Pg C has accumulated in forests ecosystem in Europe, which would offset 10% of the cumulated EU emission for the same period. This clearly shows how important forest C sequestration can be in national greenhouse gas (GHG) budgets.

Generally speaking, the accumulation of biomass stocks in European forests is a result of woody NPP exceeding losses by timber harvesting and natural disturbances, such as fire, pests and wind throw. The current harvest is only 50% of the woody NPP for conifers in Europe and only 34% for broadleaved forests, but in the 1950s, the harvested fraction of woody NPP was 1.5 times higher than today (Roberto et al., 2017). This is indicating a reduced pressure in exploiting forest resources and thus, increment in tree standing biomass and slower turnover rates, which leads to increased carbon sequestration.

The forests and forest cover of Europe are not evenly distributed over the continent (Fig. 1). Sweden has the largest areas of forested land (~ 28 million ha), followed by Finland (~ 22.2 million ha; which has the highest relative forest cover), Spain (~ 18.4 million ha) and the least forested country is Sam Marino, followed by Malta. Iceland, the Netherlands, and Moldova, which are all in the same category of forest cover ranging from about 0.8 -5.0 million hectares (Fig. 1). This shows that Iceland is not the least forested country in Europe, which is both because of afforestation and natural regeneration of native downy birch (*Betula pubescens*) woodlands in recent years (Sigurðsson and Magnússon, 2019).

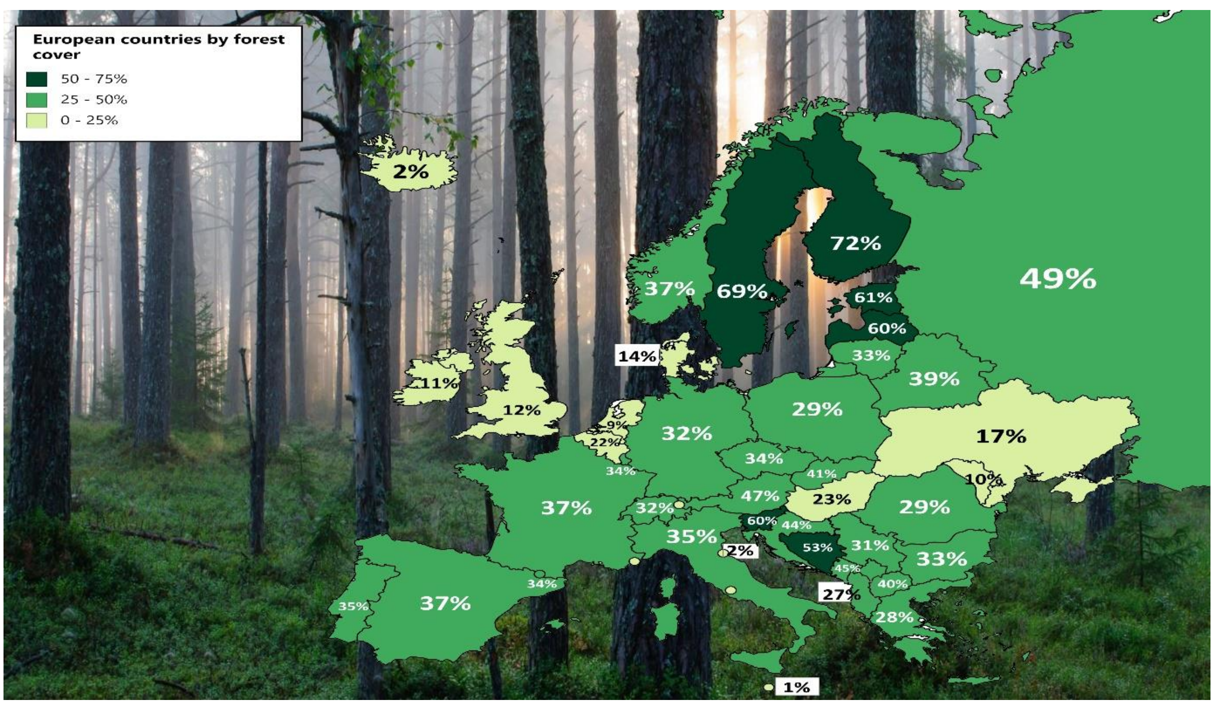


Figure 1: European forest cover by countries (Source: CIA-The World Factsheet, 2019).

1.2. Historical changes of forest and woodland cover in Iceland

Before the realisation of the need for afforestation in Iceland, forest coverage had declined from roughly 25% of the total surface land area after human settlement in the 9th century AD to <1% in the early 20th century AD, due to wood utilization, grazing, volcanic eruptions and harsh climate (Arnalds, 2015; Bjarnadóttir et al., 2007). Historically, an organised forestry started in the country in the year 1899 with the planting of the so called "pine stand" at Thingvellir and by the establishment of the Icelandic Forest Service (IFS) in 1907 (Eysteinsson, 2013). However, it was only from 1950s that afforestation activities by the IFS and forestry associations really started to increase and reached >1 million seedlings per year. By 2009 the afforestation had increased to about 4-6 million seedlings per annum (Fig. 2). However, after a collapse in the Icelandic economy in 2008, afforestation decreased to 3-4 million seedlings per annum after 2009, where it has remained until now.

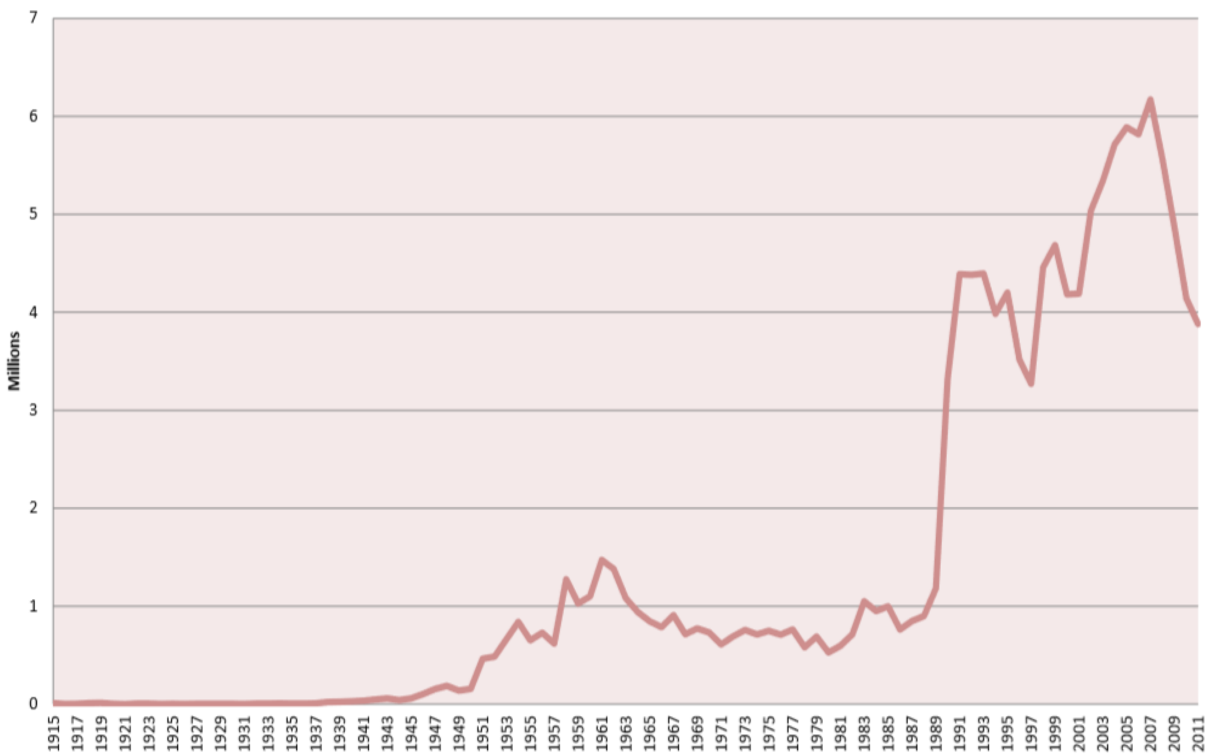


Figure 2. Total number of trees planted in Icelandic forests from 1919-2011 (Source: Eysteinsson, 2013)

The main reason for the increased afforestation in the 2000s in Iceland was the emergence of government forest-related legislation that had the goal to increase forest and woodland cover in Iceland to at least 5% of the lowland land area (3.3% of total land area) during the next 40-year period (Haraldsson et al., 2007). This would be a 275% increase in the total forest and woodland coverage of the 1990s, when it was only 1.2% (Bjarnadóttir et al., 2007). Half way down the road (ca. 15-20 years later) this goal has only been partly met. New forest map by Icelandic Forest Service (IFS, 2017) showed that in the 2010-2014 period, forests and woodland cover had reached 1.906 km^2 (Fig. 3), i.e. 1.9% of total land area of Iceland.

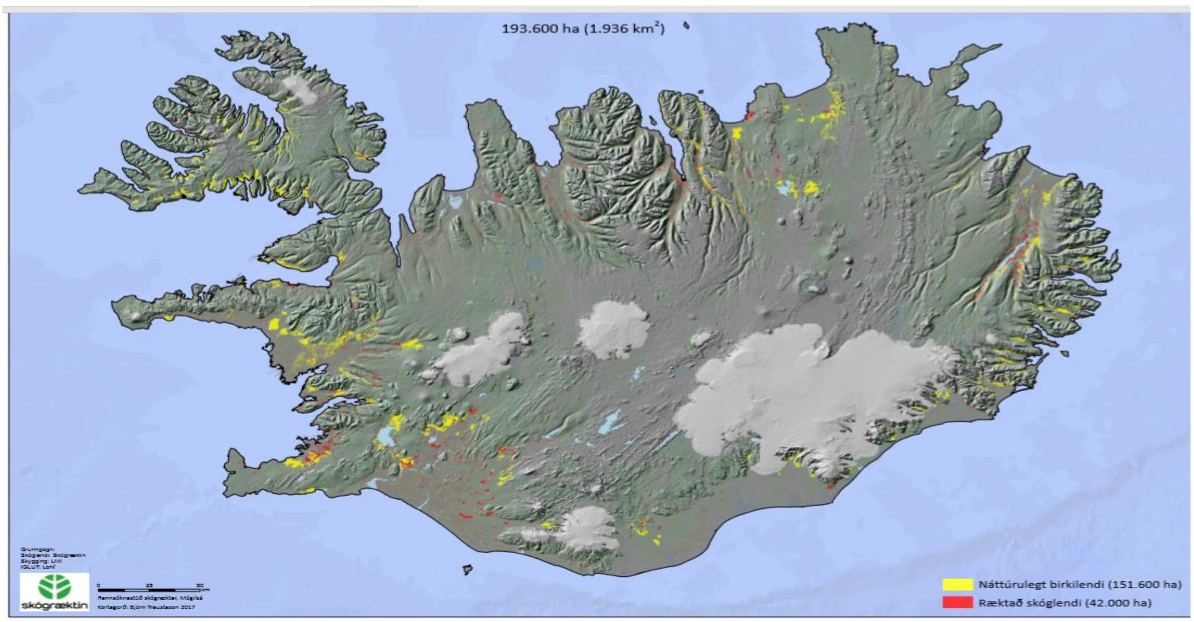


Figure 3. Map of forest coverage in Iceland between 2010-2014. Yellow areas are downy birch forests and woodlands and red areas are plantations, mainly by exotic tree species. (Source: Icelandic Forest Service, 2017).

Today, Icelandic forests are divided into natural downy birch woodlands (Fig. 4-a) and planted forests (Fig. 4-b) which cover approximately 1.506 km^2 and 400 km^2, respectively (IFS, 2016). The western region of Iceland had the largest relative area of the natural birch woodlands compared to planted forest (85% vs 15%). In the eastern and southern regions this ratio is ca. 60% vs. 40% plantations. Sigurdsson et al. (2005) gives six major tree species used in afforestation in Iceland; the native downy birch (*21%* of annual planting), and the exotic Russian larch (*Larix sukaczewii*, *22*%), lodge pole pine (*Pinus contorta*, *15%*), Sitka spruce (*Picea sitchensis 14%*) and Norway spruce (*Picea abies*, 6%). Besides the native birch, which can reach

14 m in Iceland, all the commonly used exotic tree species have reached at least 22 m in height and show mean annual increments ranging from 5 to 20 m^3 ha^{-1} yr^{-1} (IFS 2016).

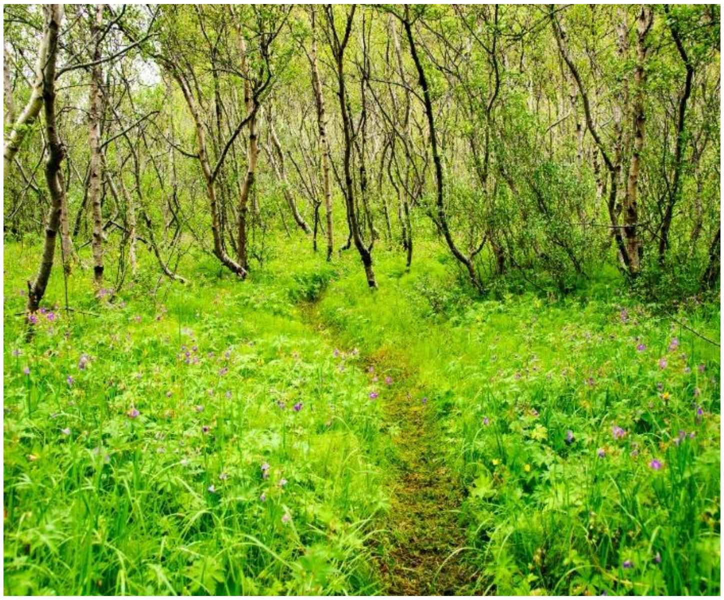


Figure 4-a. A natural birch forest in Asbyrgi in North Iceland (Photo: Kawhi 2017)

Figure 4-b. Planted lodge pole pine forest in Heiðmörk (Photo: Vanhavifta 2017*)*

Afforestation which involves conversion of treeless lands to forest plantations has been found to change ground vegetation cover, soil properties and most importantly, sequester large amounts of carbon (C) in the forests ecosystem (Berthrong et al., 2009). The recent development in afforestation projects in Iceland might give therefore the country a unique opportunity to sequester a substantial amount of C and thereby mitigate the effects of climate change. Some forests professionals believe that young forests are optimum for sequestering C because they grow faster than old forests (Harmon, 2001). He observed that younger and healthier trees do remove C from the atmosphere faster than the old decadent forests with high rates of mortality and decomposition.

1.3. The global carbon cycle and the importance of forests soils

The global carbon cycle (Fig. 5) shows how the greenhouse gas CO_2 flows through a series of interconnected reservoirs (pools) within the Earth system (Ciais et al., 2013). These reservoirs are; atmosphere (~828 Pg C), terrestrial biosphere which contains C in vegetation living biomass (450 to 650 Pg C) and in dead organic matter in litter and soils (1500 to 2400 Pg C). The last

IPCC report (Ciais et al., 2013) further noted that there is an additional amount of old soil carbon in wetlands soils (300 – 700 Pg C) and permafrost (~1700 Pg C). In the ocean, carbon is available predominantly as Dissolved Inorganic Carbon (DIC, ~38,000 Pg C). For fossil fuel reserves, the sediment storage is a sum of 150 Pg C and 1600 Pg C of deep-sea $CaCO_3$.

In the terrestrial ecosystem, the C-cycle involves interactions of several ecosystem processes, of which photosynthesis and respiration are keyfactors. Soil Organic Carbon (SOC) in forest soils results from fixation of CO_2 from the atmosphere into plant biomass by photosynthesis, which with time is deposited into the soil through litter deposits and dying roots, i.e. the direct source of SOC is from growth and death of plants, as well as indirectly from the transfer of C-enriched compounds from roots to microbes (Ontl and Schulte, 2012). Soil microbes and their metabolic activity can influence release of C back to the atmosphere through microbial respiration and decomposition (Bardgett et al., 2008). However, global warming is believed to accelerate rates of heterotrophic microbial activity (decomposition of soil organic matter), thereby increasing the efflux of CO_2 to the atmosphere and export of dissolved organic carbon by hydrological leaching (Davidson and Janssens, 2006). This implies that, when C inputs and outputs are in balanced status, there is no net change in SOC levels. However, when inputs from photosynthesis exceed C losses, SOC levels increase with time. It is a well-known fact that the terrestrial C cycle is dominated by the balance between photosynthesis and respiration (Gougoulias et al., 2014).

IPCC (2000) noted that on average all forests biomes (tropical, temperate and boreal), store relatively more C in soil than in standing tree biomass (i.e. 31% in tree biomass and 69% in soil). The report (IPCC, 2000), further explains that the forest biomes differ in this respect; in tropical forests, the relative amount of C stored in tree biomass is highest, or ~ 50% and 50% are stored in soil), while this ratio generally decreases with latitude. In Europe, an analysis was made to assess the relative share of different forest C pools (i.e. above- and below-ground biomass) based on data from countries that reported all the pools (Fig. 6). It can be observed that there the C-stocks in soil dominate and are about twice as large as the standing tree biomass pool (Forest Europe and FAO, 2011). The other C pools are smaller relative to the standing tree biomass C and the SOC. Kindermann et al. (2008) estimated the proportion of European Forest C-stock to be; SOC (398 Pg C), standing tree biomass (234 Pg C), litter (62 Pg C), dead wood (41 Pg C) and fine roots (23 Pg C).

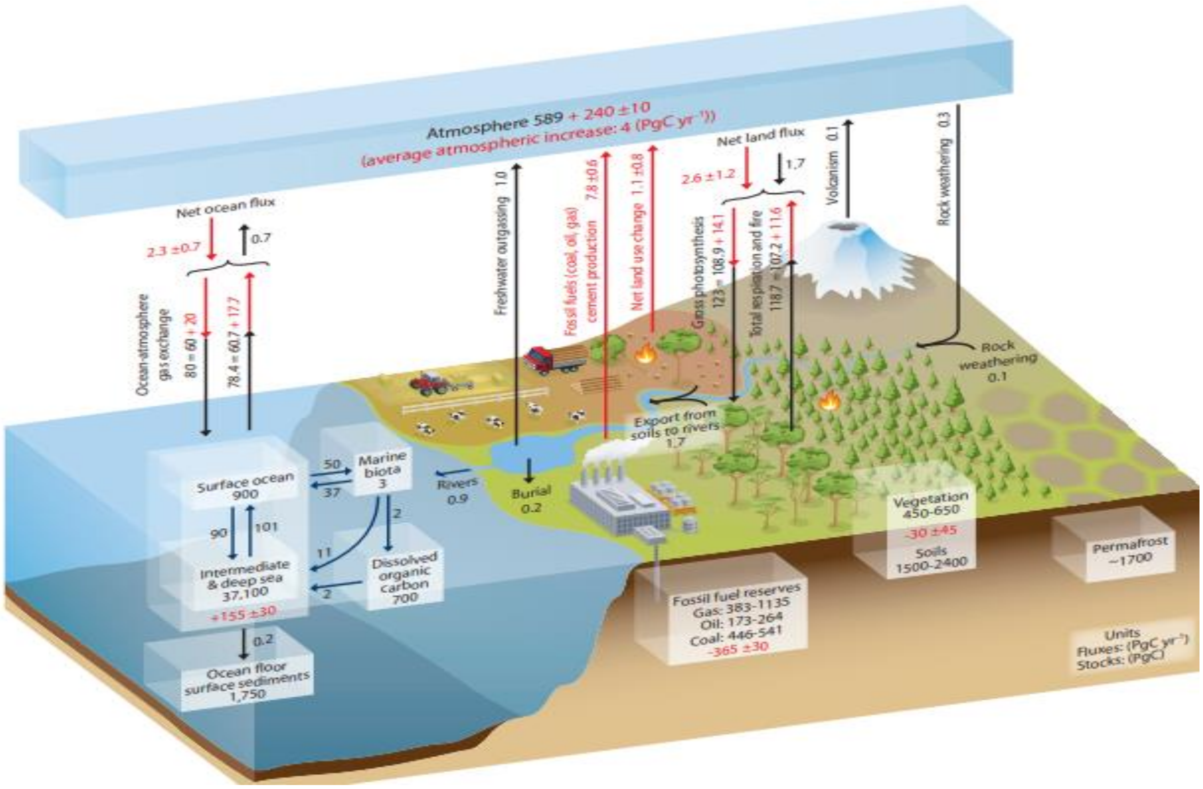


Figure 5. Simplified schematic of the global carbon cycle (Source: Ciais et al., 2013). Numbers represent reservoir mass, also called ‘carbon stocks’ in Pg C (1 Pg C = 10^{15} g C) and annual carbon exchange fluxes in Pg C yr^{-1}). Black numbers and arrows indicate reservoir mass and exchange fluxes estimated for the time period to the industrial Era, about 1750. Fossil fuel reserve are for GEA (2006) and are consistent with numbers used by IPCC WGIII for future scenarios. The sediment storage is a sum of 150 Pg C of the organic C in the mixed layer and 1600 Pg C of the deep-sea $CaCO_3$ sediments available to neutralise fossil fuel. Red arrows and numbers indicate annual ‘anthropogenic’ fluxes average over the 2000-2009-time period. These fluxes are the perturbation of the C-cycle during industrial Era post 1750. These fluxes (red arrows) are: Fossil fuel cement emissions of CO_2, Net land use change, and the average atmospheric increase in CO_2, also called ‘CO_2 growth rate’. The uptake of atmospheric CO_2 by the ocean and by terrestrial ecosystems, often called ‘C sinks’ are the red arrows part of Net land flux and Net ocean flux. Red numbers in the reservoirs denote cumulative changes of anthropogenic carbon over the industrial period (1750-2011). By convention, a positive cumulative change means that the reservoir has gained C since 1750. The cumulative change of anthropogenic C in the terrestrial reservoir, is the sum of C cumulatively lost through land use change and C cumulated since 1750 in other ecosystems. Note that the mass balance of the two ocean C-stocks surface ocean and intermediate and deep ocean includes a yearly accumulation of anthropogenic C. Uncertainties are reported as 90% confidence intervals. Emissions estimates and land and ocean sinks are in red. The change of gross terrestrial flux (red arrows of Gross photosynthesis and Total respiration and fires) have been estimated from the CMIP5 model results. The change in air-sea exchange fluxes (red arrows of ocean atmosphere gas exchange) have been estimated from difference in atmospheric partial pressure of CO_2 since 1750. Individual gross fluxes and their changes since the beginning of Industrial Era have typically uncertainties of more than 20%, while their difference (Net land flux and Net ocean flux in the figure) are determined from the independence measurements with much higher accuracy. Therefore, to achieve overall balance, the values of the more uncertain gross fluxes have been adjusted so that their difference matches the Net land flux and the Net ocean flux estimates. Fluxes from volcanic eruptions, rock weathering, (silicate and carbonates

weathering reactions resulting into small uptake atmospheric CO_2), export of C from soils to rives, burial of C in fresh water lakes and reservoirs and transport of C by rivers to the ocean are all assumed to be pre-industrial fluxes, that is unchanged during the 1750-2011. The atmospheric inventories have been calculated using a conversion factor of 2.12 Pg C per PPM.

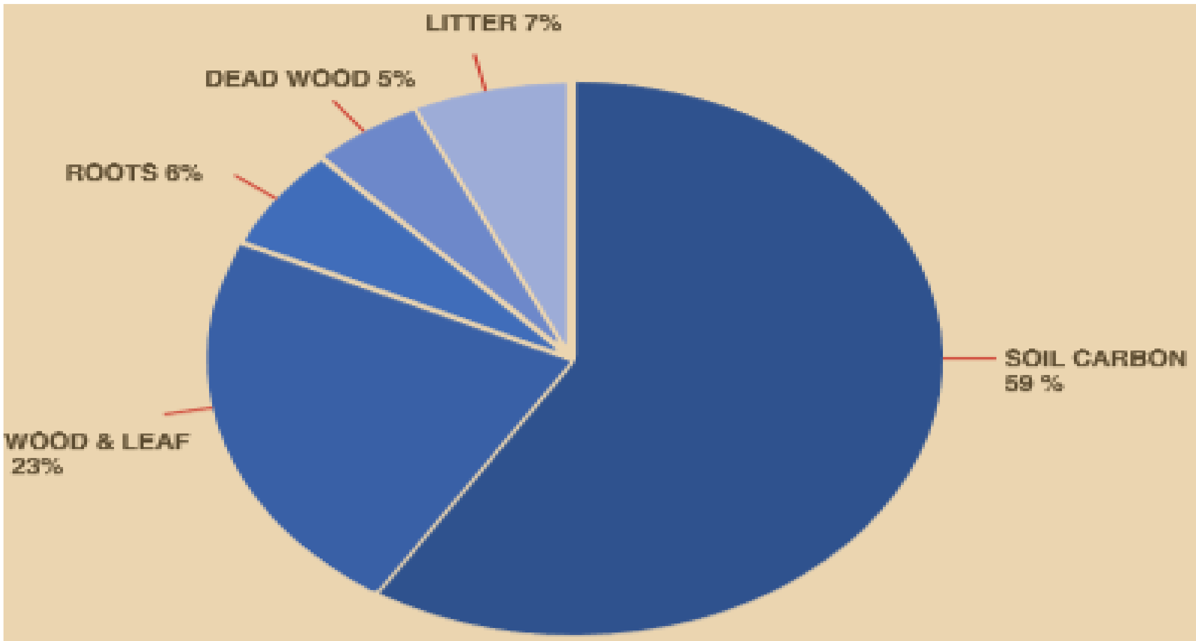


Figure 6. Portions of the five forest C pools in European countries in percent. Numbers are based on data from countries reported on their C pools. (Source: Forest Europe and FAO, 2011).

Generally, forests in the northern temperate zone are believed to assimilate more CO_2 than they release to the atmosphere. The reasons for this are being debated (Magnani el al., 2007; Ciase el al. 2008; Luyssaert et al., 2010), but recent studies suggest that the main drivers are forest management, nitrogen deposition and/or the combined effect of nitrogen deposition, increased atmospheric CO_2 concentration and climate warming (Erb et al., 2013; Pretzsch et al., 2014). Thus, European forests are very important from the climate change perspective and because of their increased C sink function, especial attention has been given to this ecosystem in search for the 'missing' carbon sink (i.e. natural C sequestration by the terrestrial ecosystems; De Vires et al., 2009).

1.4. The Icelandic carbon cycle

A description of processes related to carbon cycle in Iceland indicates a similar sources and sinks of C as that of the global C cycle (Fig. 7). The largest emission is from respiration by plants, animals and soil (66 million tonnes) each year, followed by drained wetlands (ca. 8.5 million tonnes). The latter is estimated based on the area of drained land and international emission factors for northern Europe. Volcanic activity generally emits about 2.2 million tonnes of CO_2 each year and emissions from industry, energy production, agriculture, waste and chemicals use in buildings and transport amounted to a total of 3.4 million tonnes. It is therefore estimated that about 80 million tonnes of CO_2 will be released from Iceland every year and about 18% of that emissions comes from direct or indirect humans' activities like fossil energy use and land use change (Fig. 7).

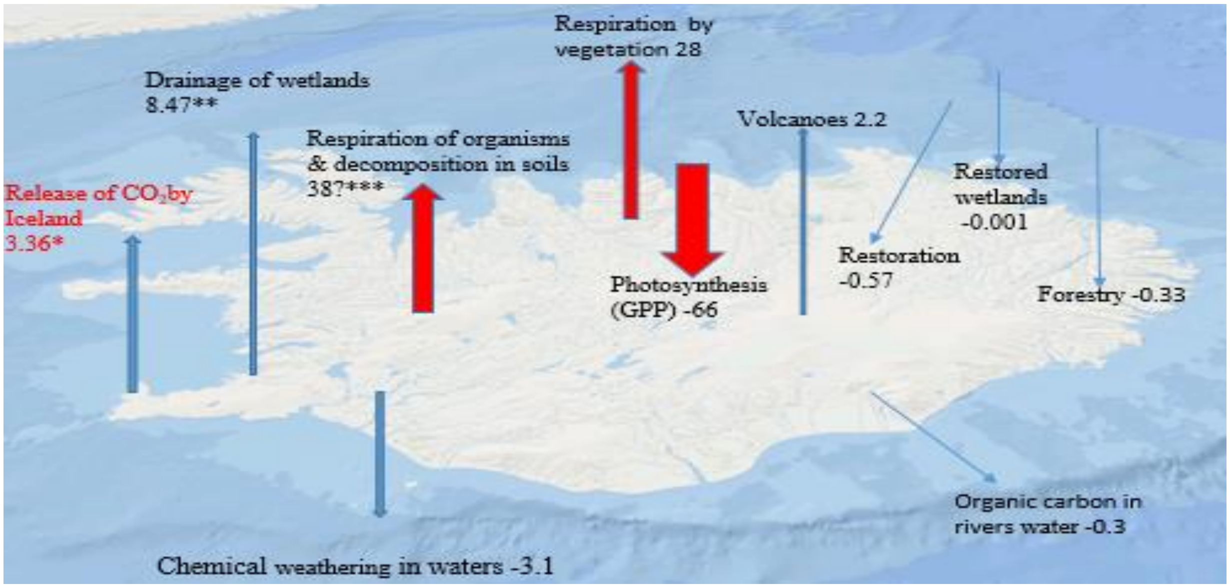


Figure 7. Simplified schematic of Icelandic carbon cycle in 2005-2015 as adapted from (Björnsson et al., 2018). Numbers represent the flow of carbon in millions of tons of CO_2 per year. Positive figures show emissions from land to the atmosphere and negative shows the absorption of atmospheric CO_2 and its transfers into various other pool. Figures on emissions by Icelanders, emissions from wetlands and uptake through afforestation, land reclamation and wetland recovery are for 2015, while other figures are for 2006. * CO_2 emissions from human beings from Iceland was 3.36 million tonnes CO_2 in 2015, when total greenhouse gas emissions totalled 4.54 million tonnes, when other greenhouse gases were added. These figures do not include emissions due to land use change (drainage of wetlands, etc.). In addition, international flights and navigation are not included. ** CO_2 emissions from wetlands, which have either been converted to farmland or are classified as general grassland, were estimated to be 8.47 million tonnes of CO_2 in 2015. This is by far the largest source of greenhouse gas emissions related to land use, with a total of 10.27 million tonnes released. As far as Iceland's carbon balance is concerned, this factor is the most uncertain. In part, it also overlaps with the amount of CO_2 that is expected to be released annually from drainage wetlands.

The annual uptake of CO_2 by terrestrial vegetation through photosynthesis is about 66 million tonnes (Fig. 7). In addition, afforestation, land reclamation and wetland restoration amounted to 0.9 out of millions of tonnes of CO_2 in 2015. Chemical weathering of inorganic CO_2 and transport of organic C by rivers to the sea totalled about 3.4 million tonnes in 2006. It is thus, estimated that the annual uptake is about 70.3 million tonnes of CO_2 which is 1.3% of human induced emissions (afforestation, land reclamations and wetlands restoration). It should be noted that the difference between emission and uptake of CO_2 (i.e. 80-70.3 million tonnes) is the Icelandic carbon balance over the past decades, which is negative at present.

1.5. The national forest inventory

National Forest Inventory (NFI) was introduced in forest management when the world realized the need for concerted effort to provide a complete picture of the resources available for a steady growing population (FAO, 2004). Since then, countries are required to have a five-year interval to collect and publish available information on the world's forest resources. The core purpose of NFI is to generate information for forest planning, taxation and evaluate the contribution of forests to the global carbon cycle (Gabler and Schadauer, 2007). It was observed that NFI play an important role in assessing forest C-stocks, provide information for decision making that helps scientific communities develop techniques and methods for reducing the CO_2 emissions (West et al. 2018). Recognising this importance, the 7th Conference of Parties (COP7) to the United Nations Framework Convention on Climate Change (UNFCCC) and under the Kyoto Protocol required that countries must account for complete forest carbon pools in their annual greenhouse gas inventories (UNFCCC, 2001), i.e. it is the entire forest carbon balance that is linked to the atmosphere, not only the balance of some parts of it. COP then named five major carbon pools in terrestrial ecosystem as; above ground-biomass (trees and vegetation), below-ground biomass (soil organic matter and roots), the dead mass of litter, coarse deadwood of standing trees or fallen and SOC as in (Fig.6). The Protocol emphasises that pools must not be double-counted and significant components of those five should not be excluded. Therefore, the present study aims to develop a scientific methodology to quantify forest C stock more accurately in Iceland. The emphasis of this thesis is to improve the understanding of the response of ground vegetation, soil properties and ecosystem C-stocks in relatively young afforestation areas.

1.6. Effects of afforestation on ground vegetation and C-stock

Ground vegetation (GV) may include not only herbaceous species but also shrubs and low growing plants such as moss, herbs, grass and lichens. Although GV contributes only < 1% of the total biomass of a forest ecosystem (see result section-3.2; Zavitkovski, 1976), a number of studies have looked at the changes in GV following afforestation (Sigurdsson et al., 2005; 1998; Zavitkoski, 1976; Ovington, 1995). It is important to note that, despite the role of GV, relatively few measurements of C content in GV is available (Johnson et al., 2017; Hou et al., 2015; Saitho et al., 2014). Tree species can be expected to influence GV composition and biomass, especially after the tree canopy closes (Barbier et al., 2008). This was confirmed by Sigurdsson et al. (2005) who showed that after canopy closure, the ground vegetation biomass declined and its composition changed in Siberian larch and birch forest as a function of light availability (Sigurdsson et al., 2005). Another decline in GV biomass following afforestation and canopy closure was also reported in mixed deciduous-conifers tree species in Western Hungary (unpublished data by Tinya et al., 2016). Changes in GV biomass, diversity and composition after tree establishment might also sometimes be linked to competition for soil water or nutrients (Anderson and Hanson, 1992), but in northern forest ecosystem, light interception by the emerging tree canopy is one of the most common environmental factor driving the changes (Sigurdsson et al., 2005).

Recent empirical research on forest ecosystem C balance have mostly been based on eddy covariance measurements (CO_2 fluxes) above the canopy, which gives no understanding of the partitioning of C among the five major stocks/pools (Lagergren et al., 2017; Lasslop et al. 2012; Gubler et al., 2019). In Iceland, eddy covariance measurement has been used (e.g. Bjarnadottir et al., 2007) and yet the ground is at least covered by snow for a considerable part of the year, meaning, GV C-stock or fluxes might differ from trees. Hiejmans et al. (2007) confirmed differences in C-stock at various seasonal patterns in ground vegetation and trees. This indicates that including GV in forest ecosystem C balance may be important, also since it is recognised as a component of the above-ground and below-ground biomass pool in the IPCCs Good Practice Guidance (Penman et al. 2006). This thesis therefore presents biomass values, percentage cover, composition and C-stock changes in GV following afforestation with different tree species.

1.7. Effects of litter inputs on SOC

Litter is the layer of dead plants materials that is mainly consisting of shed vegetative parts such as leaves, branches, bark and stems existing in various stages of decomposition, above the mineral soil surface (Richter and Markewitz, 1996). It is one of the most distinctive features in a forest ecosystem and it plays an important role in edaphic communities and in maintaining soil fertility, nutrient accumulation and C-dynamics within 0-10 cm soil depth during stand development (Yavitt et al., 1996; Hilli, 2008). Much of the ecosystem C and the energy fixed by plants is periodically added to the litter layer, from where parts of it become SOC after decomposition, translocation and stabilization, processes that also control the nutrient turnover in terrestrial ecosystems (Cao et al., 2019). During these processes part of the litter C is released back to the atmosphere but part of it becomes SOC that can be stored for a long time (Cao et al., 2019). The SOC does not only accumulate in the soils in the top layers, but litter from roots can replenish it in deeper soil layers, as well as dissolved organic carbon (DOC) that is created in the litter layer can be transferred into soil profile through leaching and where it can be stabilized into SOC (Novara et al., 2015).

It has been observed that about 10-30% of the existing litter layer (fine litterfall, i.e. leaves and reproductive parts) enter the soil annually (Chapin et al., 2002). In Europe this annual flux has been estimated as an average C input of 224 g C m^{-2} yr^{-1}, representing substantial percentage of net primary production of 36% in the north and 32% in central Europe (Neuman et al., 2018). The annual average litter production is about 800 g m^{-2} yr^{-1} in temperate forests and 1,200 g m^{-2} yr^{-1} in the tropical forests, excluding the contribution of woody tissues (Hattenschwiler, 2005).

The quantity of necromass production and its relative C input to SOC is believed to be partly determined by tree species (forest types); especially the composition and structure of the lignin materials found within plant species (Rahman et al., 2013). The variations in decomposition rates control the litter C stocks, which may range from ca. 6 Mg C ha^{-1} for broadleaf to more than 12 Mg C ha^{-1} in conifer forests (Domke et al., 2016). This is contrary to what was long established by Crow et al. (2009) that needle-derived lignin compounds were better preserved in soils of coniferous forest, whereas root-derived lignin compounds were a greater source of SOC in soil of deciduous forests. While, annual litter fall (leaves) inputs to SOC in temperate forests was estimated as 651 g m^{-2} yr^{-1} and 360 g m^{-2} yr^{-1} in mixed conifers and deciduous, annual fine roots

(< 5 mm) biomass was 1.97 g m^{-2} yr^{-1} for conifers and 1.65 g m^{-2} yr^{-1} for deciduous fortes (An et al., 2017). This suggests that litter inputs to soil can differ substantially between forest types and their chemical composition can determine the stability of SOC (Wang et al., 2016).

The above studies and many more show the important of including C-stock in litter when compiling annual estimates of forest C stocks and changes over time. Since 2001, the IFS has been measuring tree and other site-level variables related to forests ecosystem C pool in its NFI, but without including C-stock in litter (Tomppo et al., 2010). The prediction model for litter C-stock changes with age that has been employed by IFS in Iceland's carbon bookkeeping is based on few existing studies (Tomppo et al., 2010), but not associated with systematic litter sample collection throughout the country, which means litter C may not be accurately estimated in forests of Iceland. Direct litter collection from the forests and laboratory measurement of C in litter is the approach used in the current study, and I hope that this information can be used to improve the NFI work. The thesis therefore present changes in litter biomass, depth or thickness, pH, C/N and C and N-stocks, thereby improving on the accuracy of forest C estimates which could be adopted by individual forest owners or NFI.

1.8. Previous knowledge on soil organic carbon

Amount of SOC

Soil organic carbon (SOC) has been recognised as the 'soul' of the soil and a key property for soil quality and it is one of our major 'friends' in sequestering C and thereby decelerating global climate warming (Wiesmeier, et al. 2019). Numerous studies which give global estimates of SOC stocks for the last years have been summarized in Fig. 8. Although most studies report a global SOC estimates of approximately 1,500 petagrams of C (Pg C 10^{15} or billions tons of C), there is a considerable variation in those estimates. Earlier studies of global SOC sometimes yielded higher numbers, e.g. Bohn (1976) who calculated a global SOC pool of 2,946 ± 500 Pg C based on FAO soil maps for South and North America, Asia, Africa, Europe, and Oceania, separately. The latest, most detailed estimates of global SOC stock (using profiles to 1 m depth) have been based on the Harmonized Soil Database from FAO (2009), and reported total stocks of 1,417 Pg C (Hiederer, 2011). It is important to note that the wide variations in estimates of global SOC stocks reflects the disparity in sampling period, intensity and spatial resolution of the soil profile databases, and differences in calculation approaches (Figure 8). For example, the 10,252 georeferenced soil profiles available in the latest version of the World Inventory of Soil Emission Potentials (WISE) collected from the year 1925 onwards are unevenly distributed with most from Africa (41%, largely sub-Saharan), Asia (18%) and South America (18%), and very few profiles for North America (8%), Oceania and the north temperate regions (2%; Batjes, 2009).

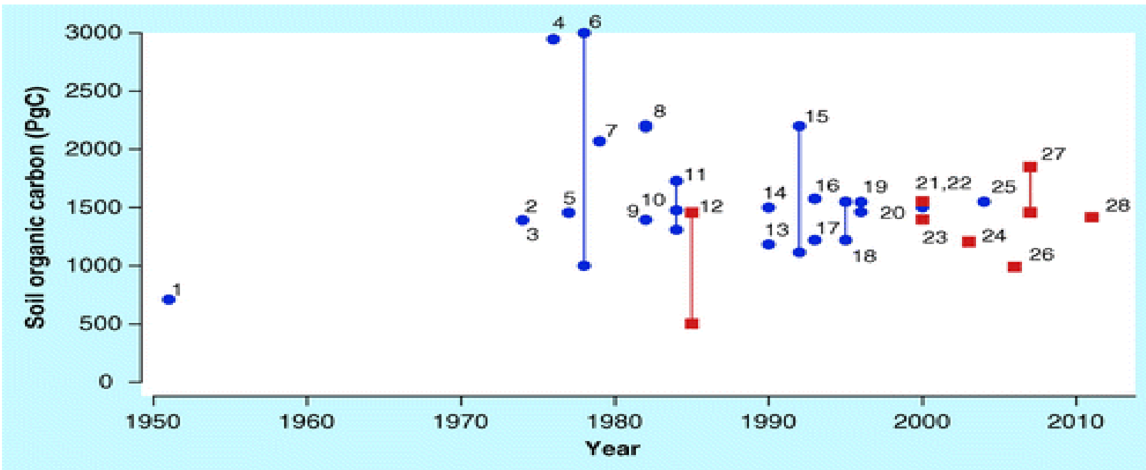


Figure 8. Estimates of global SOC from the literature through time (Source: Scharleman et al., 2014). Median across all estimates is 1460.5 Pg C, range 504–3000 Pg C, n = 27 studies, based on spatially explicit (red; median 1437 Pg C, range 504–2469.5 Pg C, n = 7) and nonspatially explicit methods (blue; median 1388.5 Pg C, range 710–3000 Pg C, n = 20). Lines connect minimum and maximum estimates of soil organic carbon reported by the same study. Numbers refer to references provided by Scharleman et al. (2014).

Vertical distribution of SOC

SOC stock is highly heterogeneous due to complex environmental interactions (Dorji et al., 2014). Several studies give the vertical distribution of SOC at global, regional and national scales (Kern, 1994; Batjes, 1996; Jobbagy and Jackson, 2000; Scott et al., 2002). For example, SOC reserve worldwide in the upper 30 cm is estimated at between 684-724 Pg C and in the upper 100 cm it is estimated at between 1462- 1548 Pg C (Btajes, 1996), i.e. ca. 50% of the SOC stock in the top 1 m is found in the 30 cm layer. Similarly, in New Zealand it was estimated that 1152 (28%), 1439 (34%) and 1602 (38%) Pg C was found in 0-0.1, 0.1-0.3 and 0.3-1.0 m depth, respectively, or 62% in the top 30 cm (Scott et al. 2002). Further, in European forest soils it is estimated that ca. 53% of the 1 m deep SOC stock is found in the top 30 cm (between 11.4 to 12.2 of 21.4 to 22.5 Pg C; De Vos et al., 2015).

The vertical distribution of SOC in Iceland is described as erratic because of variability in environmental conditions during the process of soil formation (Arnalds, 2015). This leads to variations in SOC stock depending on soil types. Oskarsson et al. (2004) used the Agricultural University of Iceland (AUI) soil database and generated the average depth distribution of SOC for different soil types (Figure 9), which indeed show a decreasing trend with depth in Andosols, but not as pronounced as has been reported from Europe or New Zealand.

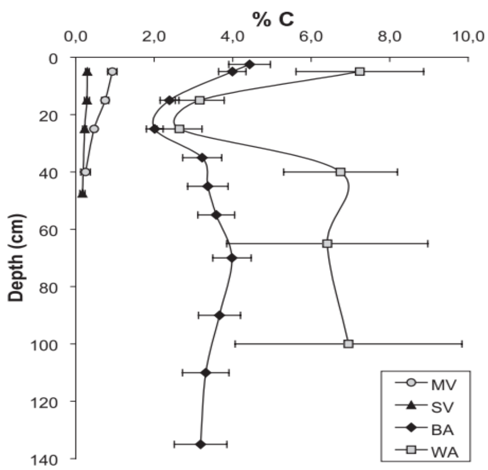


Figure 9. Average vertical distribution of SOC in the four most common soils of Iceland: Combic Vitrisol (MV), Sandy Vitrisols (SV), Brown Andosol (BA) and Gleyic Andosol (WA). (Source: Oskarsson et al. 2004).

Earlier it has been recognised that soil depth plays an important role in SOC distribution, and although SOC concentration generally decreases with increasing depth, tree species has been showed to play a big role in SOC distribution down the soil profile in forests (Oostra et al., 2006). This is partly attributed to the differences in root distributions and above- and below-ground allocation pattern of SOC (Jobbagy and Jackson, 2000). Roots distribution affects the vertical placement of SOC and a study of six tree species in Sweden revealed that in the mineral soils, SOC is ranged in the order of elm (*Ulmus glabra*), oak (*Quercus robur*) > ash (*Fraxinus excelsior*) = hornbeam (*Carpinus betulus*) > Norway spruce > beach *(Fagus silvatica*; Oostra et al., 2006). The dominant plant functional type (i.e. its physical, phylogenetic and phenological) characteristics, can also have a significant effect on the vertical distribution of SOC (Jobbagy and Jackson, 2000). The percentage of SOC in the top 20 cm in shrublands, grassland and forests are reported at 33, 52, and 50% of the 1 m SOC stock, respectively, (Jobbagy and Jackson, 2000). This indicates that the relative distribution of SOC with depth is as strongly associated with vegetation as with climate. Other factors that influence SOC vertical distribution may include local-scale variability in soil environment and microclimate aspects such as soil temperature, moisture, erosion, microorganisms and changes in land use (Begum et al., 2010; Egil et al., 2009; Sharma et al., 2010). The fpcust of this study was estimate SOC stock to 30 cm depth (0-5, 5-10, 10-20, 20-30 cm), including the analysis of potential effects of forest types on SOC distribution.

Accumulation rates of SOC following afforestation

During the past two decades, a number of studies have quantitatively reviewed changes in SOC stock, either with respect to different land-use changes (Post and Kwon, 2000; Lal, 2005; Guo and Gifford, 2002; 2011, Poeplau et al., 2017) or more specifically following afforestation (Paul et al., 2002; Berthrong et al., 2009; Laganiere et al., 2010; Nave et al., 2013; Shi et al., 2018; Pan et al., 2011; Barcena et al., 2014). Depending on the location and the stand and age of the forest, the average rates of SOC sequestration by the world forests is estimated at 2.4+/-0.4 Pg C yr^{-1} (Pan et al., 2011). For European forests, various modelling studies suggest that SOC is being sequestered at 0.3 Pg C yr^{-1} (Liski et al., 2002). At a stand scale (ha scale) these average SOC-sequestration rates have been estimated to 0.9 t C ha^{-1} yr^{-1} (Dijkstra et al., 2009), 0.8 ha^{-1} yr^{-1} (Schils et al., 2008) and 0.41 t C ha^{-1} yr^{-1} (Gruneburg et al., 2014). Another study reported a mean sequestration rate for European forests at 2.98 t C ha^{-1} yr^{-1} which peaked after 38 years and stabilized at an average value of 0.8 t C ha^{-1} yr^{-1} (Nabuurs and Schelhaas, 2002).

In Iceland, SOC sequestration rate is given in the national inventory for the UNFCCC which indicates average soil carbon sequestration factor of 52 g C m^{-2} yr^{-1} following afforestation (Hellsing et al., 2016). The Icelandic National Inventory Report bases this on a number of field studies which found an increase in the SOC stock and therefore assign a positive soil carbon sequestration effect to afforestation. These studies include Snorrason et al. (2002), Ritter (2007), Bjarnadottir (2009 Jónsson and Snorrason, (2018), Aradottir et al. (2000), and Arnalds et al. (2013). They give the average C sequestration rate by revegetation and afforestation in Iceland, irrespective of the region. In addition, the Icelandic Soil Conservation Service continuously reviews this value by ongoing monitoring of C- stocks in revegetation areas (Hellsing et al., 2016). The present study will be a valuable addition to the state of knowledge on these processes in Iceland.

Considering the complete accounting of C stocks in a forest ecosystem (wood, roots, ground vegetation, litter and soil) that was recommended by IPCC (2003,2006), few studies in Iceland have looked at both above and below-ground components together. The first measurement of C stock for both forest above and below-ground biomass was conducted by Jonsson (1985), where lodge pole pine plantation in Northern Iceland was measured. This was followed by Sigurdardottir (2000) who estimated the above- and below-ground C stock in downy birch, lodge pole pine and Siberian larch stands in eastern Iceland. More forests ecosystem C studies have been conducted in eastern Iceland (Bjarnadóttir, 2009; Sigurdsson et al. 2005), southern Iceland (Sigurdsson, 2001) and in western Iceland (Sigurðsson et al., 2008). A recent Nordic overview, including most of the Icelandic data is found in Bárcena et al. (2014).

The above studies demonstrated that the rate of SOC sequestration and magnitude depend on the complex interaction between climate, soils, tree species and management and chemical composition of litter as determined by the tree species (forest types). This thesis therefore presents how SOC sequestration rates vary across forest types with age and tree size (basal area and stem volumes).

Changes in SOC with time following afforestation (chronosequence studies)

One of the common method used to estimate the relationship between SOC sequestration and age is the chronosequence approach (Kashian et al., 2016). A meta-analysis of SOC and litter C-stock with afforestation age in Northern European countries (Denmark, Sweden, Iceland, Finland, Norway, Estonia, Latvia, Lithuania and the British Isles) showed immediate change in C- stock in litter layers following afforestation, but in the mineral soil it took on average 30 years until the changes were statistically significant (Barcena et al., 2014). This means that C can slowly be transferred from dead plant materials on the forest floor (O-horizon) to the deep mineral soils. Paul et al. (2002) and Davis et al. (2002) in their review of previous studies, confirmed that while there were no consistent changes in soil C a few decades after afforestation, but there was typically an increase in C- stock in dead litter layer.

Few studies have also looked at C balance following afforestation on drained organic soils. Uri et al. (2017) studied net ecosystem production (NEP) and net primary production (NPP) in five downy birch stands aged between 12 and 78 years in a well-drained peatland in Estonia. Young and middle-aged birch stands (12-30-years-old) turned the drained wetland balance to a C sink

(1.4 Mg C ha^{-1} yr^{-1}) while with older stands (78-years-old) the drained wetland acted as C source, emitting 0.95 Mg C ha^{-1} yr^{-1}. No drained wetland sites were found within my study areas in SW-Iceland.

Recently, a study of the above- and below-ground biomass production and C balance of a 10-30–year old downy birch stands in peat cutaway areas in Finland showed that after self-thinning, the above-ground biomass increased from 17 to 79-116 Mg ha^{-1} within 25-30 years (Hytonen et al.,2018). A cutaway peatland may represent conditions that are comparable to secondary succession following soil erosion in Iceland. The organic matter in the top O-layer (litter layer) increased linearly with the stand age reaching 29.3 Mg ha^{-1} (ca. 1/3 to ½ of the C-stock change) in the oldest stand, considerably contributing to increased C-sequestration rate in this forest ecosystem (Hytonen et al., 2018).

1.9. Effects of afforestation by different tree species on soil properties

Several studies have studied the effects of trees on soil physical and chemical properties, soil formation and nutrient cycling (e.g. Shear and Stewart 1934; Alriksson and Olsson, 1995; Menyailo et al., 2002; Finzi et al., 1998; Sauer, 2012). Other studies have demonstrated a close relationship among different tree species or forest types on soil properties and organic carbon accumulation (Hobbie, 1992; Jandl et al., 2007; Raulund-Rasmussen and Vejre 1995, Ayres et al., 2009; Wiesmeier et al., 2013). van Breemen (1993) concluded in his review on the influence of plants on soil and microorganisms, that different species appear to affect soil fertility, soil moisture content and other soil properties and such effects are species-specific. This was e.g. confirmed when stands of trembling aspen (*Populus tremuloides*), lodge pole pine and Engelmann spruce (*Picea engelmanii*) were studied (Ayres et al., 2009). Thus, tree species can differ in their influence on soil properties (Lal, 2005), and between native and exotic (Kaye et al., 2005). Even species within the same functional group, such as broadleaf or conifer trees, can differ in their short-term effects on soil (Russel et al., 2004; Knoke et al., 2008). The current thesis shall focus on the extent to which native deciduous and exotic conifer (and mixed) forest types influence physical and chemical properties of soils.

Soil bulk density

Soil bulk density (BD) is one of the physical properties of soil which must be measured (sampled) when estimating SOC stock (Holmes et al., 2011). A number of studies have showed the importance of BD to SOC estimation (Goidst et al., 2009; Schwager and Mikhailova, 2002; Post et al., 2008, Chan et al., 2010). Bulk density is critical in assessing SOC because of the differences in soil mass caused by land use change and management (Ellert et al., 2007). Establishing trees on treeless land may lead to variation in BD values which can greatly influence the estimate of SOC stock when conducting depth-based studies (Ellert et al., 2002). This means that sampling soil to a certain depth when BD values vary (more g soil per cm^3), will lead to more soil taken from one site than in another. This will cause difference in the estimate of the total amount of SOC to a fixed depth. Therefore, is it important to consider differences in BD when comparisons are made, and if that is not done it can easily produce a measurement bias (Wuest, 2009). Using analytical uncertainty, Goidts et al. (2009) observed that BD accounted for up to 25% and 10% of SOC stock variability in stony and non-stony soils, respectively.

It has long been a tradition to calculate SOC to a fixed depth (Stolbovoy et al., 2007), however this method has been showed to introduce substantial errors when comparing sites with different land uses (Ellert and Bettany, 1995; Ellert et al., 2002; Wendt and Hauser, 2013). The error is that, after changing land use (for example, from treeless to afforested or from tillage to non-tillage), sampling soil to a fixed depth (say 0-30 cm) will reduce soil mass by small grams in the top 0-20 cm layer in afforestation or tillage as the case may be. Differences in bulk density and C concentration may be caused by tree roots expansion in the case of afforested soils resulting in large variation in estimates. The error in estimates can be avoided using equivalent soil mass (ESM) or soil mass correction factors (SCF) method which is a recent approach that accounts for the differences in soil masses and bulk density among treatments (Wendt and Hauser 2013; Weismeier et al., 2015). Although difference in BD can be small, correcting soil mass is extremely important to accurately calculate SOC stocks to a known depth and to avoid over- or underestimation of SOC (Barcena et al., 2014).

My study was depth-based and aimed to sample soil to a known fixed depth (30 cm), which is the most accurate and acceptable soil depth when the intension is to estimate effects of land use change on SOC (Baker et al., 2007; Harrison et al., 2003). This kind of study typically requires that bulk density values must be determined (Périé and Ouimet 2008; Dold et al., 2018). The current thesis therefore presents this relationship and shows the importance of including it in the SOC stock calculations.

C and N concentration

There is generally a strong linear relationship between soil C and N concentration (C and N as % of soil dry mass), especially in the 0-30 cm soil depth (Cleveland and Liptzin, 2007; Côté et al., 2000). This relationship depends on the soil profile (Matamala et al., 2008), forest type (Marty el al., 2017) and land use change (Jobbagy and Jackson 2000). Jobbagy and Jackson (2000) showed that changes in land use has a significant impact on C and N concentration in soil and they observed that afforestation can improve C and N concentration in the soil through accumulative organic matter input from litter decomposition and stabilization. Soil degradation, on the other hand, leads to reductions in C and N concentrations. For example, in Icelandic soil, it has been indicated that C concentration vary highly depending on the levels of soil degradation. Highly degraded Andosols has a C concentration of <20 mg g^{-1}, while undisturbed soils have C concentration of 30-70 mg g^{-1} (Oskarsson, et al., 2004 in Ritter, 2007).

What is a "normal" store of N in soils? Danish forest soils has N concentration of 610 N g m^{-2} (Vejre et al., 2003), Finnish forest ranges from 400 to 1,100 C kg m^{-2} (Finér et al., 2003). Carter et al. (1998) found a range of 360 to 1,050 g N m^{-2} in Canadian agricultural soils, while Zinke and Stangenberger (2000) reported a mean value of 610 g N m^{-2} to a 1-m depth in the dense conifer forest and 270 g N m^{-2} in low dense forest on the lower slope of Sierra Navada. Batjes (1996) also calculated soil C and N storage in 1 m soil depth using worldwide database and found soil N content of 520 g N m^{-2} for Andosols and other soil orders had very higher mean values. These values can be kept in mind when the present study is read, but it should be noted that its values are limited to the top 30 cm of soil.

Soil pH

Soil pH, which measures the acidity or alkalinity of soils, is associated with many soil properties including SOC content (Lambkin et al., 2011) and nutrients uptake (Williston and LaFayette, 1978). There are a number of studies from the northern hemisphere that looked at how various reactions in soils are controlled by pH (Wang et al., 2000; Breeman et al., 1998). A good example of this is plant-nutrient interaction. London et al. (2006) explains that, although, the vast majority of tree species can live at a broad range of pH values, some may or may not be able to use certain nutrients depending on how alkaline or acidic (high or low) the soil pH is. They observed that extremes in pH (<4.5 and >8.5) can make some nutrients toxic and others unavailable to plants. The ideal pH is close to neutral, and neutral soils are considered to fall within a range from a slightly acidic pH of 6.5 to slightly alkaline pH of 7.5 and plants nutrients are optimally available to plants within this pH range (6.5 to 7.5; Grayston et al., 1997). Abnormal root development and progressive fall in dry-weight production has been observed in pH above and below 4 and 5 in Sitka spruce seedlings (Leyton, 1952). On the other hand, trees also affect soil pH through the release of organic substances from the roots (root exudes) which influence soil microbial activity (Grayston et al., 1997). Marschner et al. (2004) observed a change in soil pH for the rhizosphere community induced by three different tree species and this affected the bacterial community as bacteria species differ in their pH optimum, i.e. the pH at which they are most competitive.

The aim of this thesis was therefore to compare changes in pH between treeless and afforested soils, and to examine the similarities and differences under different forest types (native deciduous, exotic conifer and mixed). Chemicals emitted from tree roots are reported to modify soil pH (Ralph et al. 1981; Richter et al., 1994) and also litter types (Finzi et al., 1998; Noble et al., 1996). I therefore present changes in the pH of the upper top soil (0-5 cm) depth and litter following afforestation.

Soil C/N ratio

The ratio of carbon to nitrogen concentrations (C/N ratio or C:N ratio) following afforestation have been examined by e.g. Richter et al. (1999), Lemma et al. (2006), Mao et al. (2010), Lagergren et al. (2010) and Cools et al. (2014). They show that globally there are large variations in how afforestation can affect C/N ratios. The global mean C/N ratio is estimated at 37.32 +/- 2.37 (Xu et al., 2016), and for European forests it ranges from 13 in litter layer to 44 in mineral soils (Cools et al., 2014). Influence of tree species and soil depth are the most important explanatory variables for C/N ratio variations (Dise et al., 1989; Emmett et al., 1998; MacDonald et al., 2002). For example, under coniferous tree species, top soil C/N ratio has been proposed as an indicator for nitrate leaching (Vesterdal et al., 2013), while in deciduous forests, litter layer C:N ratio has been showed to be a better indicator of the nitrate status (Gundersen et al., 2009).

In the temperate region, it has been confirmed that conifer tree species growing in monoculture stands alter soil C/N dynamics (Challinor, 1968; Mladenoff, 1987; Boerner and Koslowsky, 1989; Reich et al., 1997). In Iceland however, there are very few studies of C/N beneath different forest types (Sigurdsson et al., 2005).

Intriguingly, litter input with various concentrations of lignin materials greatly affect C:N ratio with changes more confined to the soil surface because of slow transfer of materials to the mineral soil (Kirschbaum et al., 2008). The more recalcitrant the litter is (conifer), the higher the C:N ratio compared to labile deciduous tree litter (Guo and Gifford, 2002). This implies that, as C originating from both fresh litter and dead materials are added to the mineral soils, there is likelihood of influence on C:N ratio. The material originating from fresh litter can be expected to maintain the C:N ratio of the original litter material and thereby providing link between C:N ratio of litter and mineral soils (Finzi et al., 1998). Therefore I was interested to investigate possible changes in C:N following afforestation and also to compare the C:N ratio of litter and mineral soils.

1.10. Objective of the study

The main aim of this research project was to understand the effects of afforestation on vegetation cover, soil properties and the ecosystem C and soil N-stocks at three sites in Iceland. However, I also had the underlying aim to test if it was possible for individual forest owners to validate soil and ground vegetation C-stock changes in their planted forests by measuring minimum number of sampling plots at each site, arranged as chronosequences if the planted forests were of varying age.

The specific research questions were;

Comparison of treeless sites

1. *Are there variations in vegetation cover, soil properties and ecosystem C-stocks between the* ***three control study sites*** *without the effect of the afforestation?* This was important to know if at a later stage I wanted to merge data from different sites.

a. It has been observed that in Iceland, the species composition of vegetation (dwarf bush, herbs, ferns, grass and moss) in geothermal areas often differs from that which grows in cold areas (Icelandic Institute of Natural History, 2019). It was therefore expected that the Nesjavellir area where the Nesjavellir geothermal plant is located would have a different vegetation cover compared to Heiðmörk and Ölfusvatn control sites.

Changes in vegetation cover and soil properties following afforestation

2. *What are the average changes in vegetation cover, litter accumulation, fine root biomass, soil properties and ecosystem C and N* ***following afforestation*** *with different forest types across the three study sites* ***without considering time factor****?*

Bulk density

b. In order to accurately estimate SOC stock to a given depth, understanding changes in soil bulk density following afforestation is very important. Ellert et al. (2007) point out that differences in soil compaction at different sites can cause fluctuations in BD values. It was expected then that soil bulk density would be lower in afforested sites compared to treeless control plots and that dry mass of soil in the top 30 cm in the afforested sites would be correspondingly lower because of soil expansion.

Soil pH

c. A lower soil pH values has been reported in regenerating forest sites in a range of boreal coniferous forest stands (Ste-Marie and Paré, 1999) and in a 30-year Siberian larch chronosequence in Iceland (Ritter et al., 2007), in three commercial tree species (*Pinus radiata, Eucalyptus nitens* and *Curpresus macrocarpa*) in New Zealand (Chirino et al., 2010) as well as in a 53-year old Siberian larch site in Iceland (Haraldsson et al., 2007). Afforestation was therefore expected to change soil pH.

C:N ratio

d. It was hypothesized that C:N ratio would increase following afforestation with different tree species, but only in the litter layer.

Ground vegetation cover

e. Previous studies have showed that new forests exert influences on plant communities under their canopies (Mitchell et al., 2012; Tessema and Belay, 2017). It was therefore expected that ground vegetation consisting of dwarf bush, herbs, ferns, grass and moss would be reduced across all the three study sites following afforestation.

Biomass and litter production

f. Following afforestation, the total aboveground biomass production (ANPP), litter and fine roots biomass was anticipated to increase.

Ecosystem C and N stocks

g. Severely degraded treeless land in Iceland has been showed to have high potential to sequester C in the soil (SOC) following afforestation (Hunziker et al., 2019). It was then hypothesized that SOC stock and sequestration rates would be higher in afforested soils compared to treeless sites and since the soils were generally deeper than 30 cm, I expected that the deeper tree roots could also redistribute immobilized N from deeper soil layers to the top 30 cm.

h. I also expected to find higher C accumulation in standing tree biomass, litter, fine roots and bush in afforested compared to treeless sites.

Influence of forest types

3. *Do forest types (conifer, deciduous and mixed) affects ground vegetation, soil properties, carbon and nitrogen stocks and sequestration rates following afforestation?*

   i. I hypothesised that tree species would differ in the accumulation of C in litter, ground vegetation, trees, bush layer, SOC as well as pH values depending on the chemical composition of their detrital production, tree growth and biomass production. Coniferous trees were expected to have higher ecosystem C stocks compare to broadleaf (deciduous) and mixed forests. Soils were also expected to get acidic under coniferous stands.

Age-related changes

4. *Does tree age influence biomass growth and carbon and nitrogen accumulation in a forest ecosystem?*
    - **j.** It is known that trees do not grow and accumulate C linearly over time. Fluctuations in diameter and height increment (lower growth alternating with periods of increased growth) was expected to strongly depend on tree species. Slower age-dependent growth and C accumulation aboveground and in soil was expected in the native birch compared to the conifers and the mixed forests.

Effects of tree basal area

5. *Does carbon and nitrogen accumulation in above-ground (trees and vegetation) and below-ground biomass (litter, fine roots and soil) increase with tree basal area?*
    - **k.** Basal area and age has been used in forest management studies to provide an accurate quantification of biomass production (diameter increment) and C and N accumulation (Johnson and Abrams, 2009). I expected that basal area would be a better predictor for C stock in vegetation, litter and soil across both the native birch and conifer forest types.

Individual chronosequences

6. *Does it matter whether the trees were planted or naturally regenerating, pure or mixed stand?*
    - **l.** Grouping trees into individual strata or chronosequence (planted, natural and pure stands) affects C and N-stocks (Liu et al., 2017). The strata with planted pure conifer tree species were expected to have higher C and N-stocks compare to the other chronosequences.

m. Only at Nesjavellir was it possible to compare two strata of similar age, where one had been planted and another was naturally regenerated. This was the native downy birch. I expected that the method of regeneration would not have much effect on how the ecosystem C-stocks developed.

Minimum number of measurement plots

7. Can minimum number of measurement plots (n=3) be used to validate soil C sequestration?

n. The few existing studies in Iceland used more plots (n > 3) and have indicated that afforestation of dryland soils always leads to an accumulation of SOC, at least after some years (Barcena et al., 2014). I wanted to test if litter C and SOC could be validated using minimum number of measurement plots in individual forests. If so, it would be practically and economically feasible for individual forest owners to include such measurements in their possible C-sequestration

2. Materials and methods

2.1. Description of experimental sites

The study was conducted at three different afforestation sites covering areas ranging from 316-3,200 hectares. They are all in SW-Iceland and are named Heiðmörk, Nesjavellir and Ölfusvatn forests (Fig. 10).

Figure 10. Location of the three study sites, Heiðmörk (H), Nesjavellir (N) and Ölfusvatn (Ö). (Map from Landmælingar Íslands).

2.1.1. Geology and soils of the study sites

The geology of SW Iceland is characterized by only young basaltic bedrock, postglacial and prehistoric (Jóhannesson & Sæmundsson, 1998), which is within the neo-volcanic zone between the Eurasian and North American tectonic plates (Sæmundsson, 1992; Ward, 1971). In contrast to the bedrock in other parts of Iceland, the bedrock in SW Iceland is highly porous with pores space greater than 30µm (Franzson, 2000), which means that precipitation usually trickles down through the bedrock and emerges as springs in certain places. Therefore, the main sources of running water in southern Iceland are springs.

Icelandic soils are Andosols (soil order), which are volcanic in nature (Arnalds, 2015). At the three study sites in SW Iceland, the main upland soils types are of the sub-order Brown Andosols which according to the World Reference Base for Soil Resources (WRB) are characterized by high water holding capacity, low bulk density, high friability, strong acidity (pH < 5.0) and high Al saturation and Al toxicity to plants (FAO/Unesco, 1988).

It is also important to note that one of the study sites (Nesjavellir forest) is located near the Nesjavellir geothermal power plant and the soil of geothermal areas is often characterized by low fertility and high acidity (Burns, 1997).

2.1.2. Climate of the study sites

The climatic conditions are considered oceanic at all the three experimental sites, which are located a few kilometers away from each other (Fig. 10). The mean monthly temperature of the closest climatic stations to the three study sites ranged from 0.5 to -0.3°C (coldest month) to 11.6 to 12.0 °C (warmest months; Table 1). Their mean annual precipitation ranged from 800 to 1130 mm, as measured in the period 2008-2017 (Table 1). The mean annual temperature was about 0.3 °C higher in Heiðmörk study site than the two inland sites (Nesjavellir and Ölfusvatn), but the mean annual precipitation was ca. 16% higher at the inland sites (Table 1). This difference in precipitation is not so large when compared to the climate variation in Iceland. For example, Akureyri in N-Iceland, which is more continental and in a rain-shadow behind the highland glaciers, has much lower precipitation of ca. 370 mm (Table 1).

Elevation and distance from the ocean generally determine the local climate in Iceland (Björnsson et al., 2007). Even if temperature averages are not so different for the sites, the further inland sites of Nesjavellir and Ölfusvatn are a little less oceanic than the Heiðmörk site. E.g. the Þingvellir area reaches -10°C thirty-five days a year, while in Reykjavík it only happens seven days a year, on average.

Table 1: Mean air temperature and precipitation during 2008-2017 in the study sites in SW-Iceland. The data from Hólmsheidi, situated about 2 km distance from Heiðmörk, Þingvellir is situated about 15-16 km distance from Nejavellir and Ölfusvatn. Akureyri is an example of a weather station in another part of the country.

	Hólmsheiði	Þingvellir	Akureyri	Mývatn
Mean annual air temperature (°C)	4.7	4.4	4.4	2.2
Mean annual precipitation (mm)	815.1	1128.9	378.6	469.8
Mean air temp. (C°)				
January	0.5	-0.3	0.1	-2.8
June	9.8	10.3	9.3	8.7
July	11.6	12.0	11.0	10.4
August	10.7	10.7	10.6	9.3

2.1.3 Vegetation composition and land type of the study sites

Before afforestation, the vegetation cover of the study sites can be described as heathland or unfertile grassland which was characterised by open, slow-growing dwarf shrubs composed mainly of dwarf willow, woody willow, moss campion and other different plant species (grass, ferns, lichens and herbs). Moss accounted for more than half of the vegetation cover prior to afforestation and is particularly dominant where there is little soil and growing conditions are unfavourable. Grassland was more common vegetation cover in Ölfusvatn experimental site where land is relatively more flat and fertile.

2.1.4. Site specific description

Heiðmörk

Heiðmörk is the largest study site which has been protected from livestock grazing since 1950 and is located about 10 km south east of Reykjavik city (Fig. 11). It is a lowland site and all located at < 100 m a.s.l. Historically, tree planting in Heiðmörk began on heathland sites east of the old farm of Elliðavatn in the summer of 1949 (Reykjavik Forest Association, 2019). The total conservation area of Heiðmörk now covers 3,200 hectares of which 820 hectares is planted with coniferous tree species (Fig. 11), including lodge pole pine, Norway spruce, Sitka spruce, black cottonwood (*Populus trichocarpa*), Siberian larch, Engelmann spruce (*Picea engelmannii*) and various other tree and shrub species. At the time of protection, some small remains of native downy birch and tea-leaved willow (*Salix phylicipholia*) patches were found naturally, but later some were also planted or regenerated naturally from the original remains. Felt-leaf willow (*S.*

alaxensis) and few other exotic bush-layer species are found both planted and regenerating naturally. More than four million trees have been planted there since afforestation started and the already existing vegetation has thrived since the area was fenced off (Reykjavik Forest Association, 2019). The area under forest management represent one third of the land constituting Heiðmörk protected area (Fig. 11). Heiðmörk forest provides multiple ecosystem services, among which include regulating service (carbon sequestration and water filtration) as well as provisioning (timber, Christmas trees, mushrooms and berries).

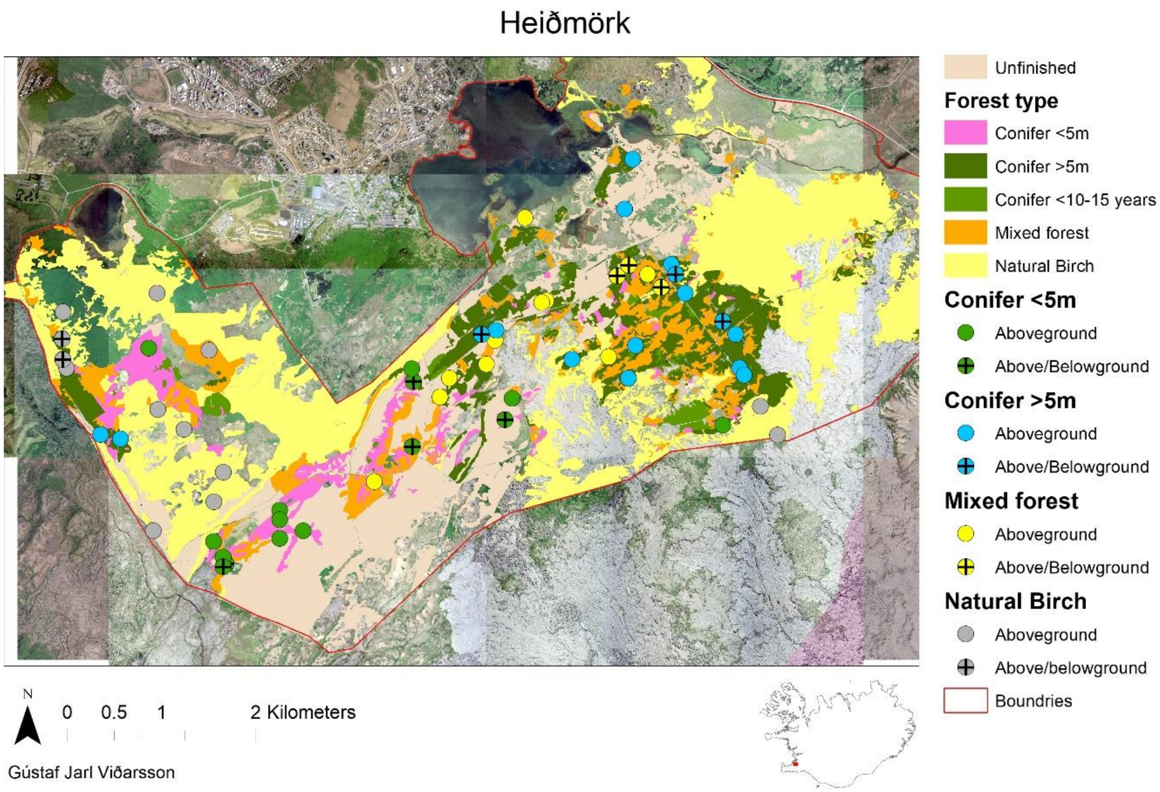


Figure 11. Location of Heiðmörk study site in SW Iceland showing experimental plots. Forest types are marked by different colours and plots included in this study are marked by crosses.

Nesjavellir and Ölfusvatn

These two forests are owned by public utility company (Orkuveita Reykjavikur). Being relatively young compared to Heiðmörk, this is the very first detailed forest inventory study conducted there, and hence, published data are limited. Gíslason and Loftsson (1997) described this area and according to them the lowest elevation is by Lake Þingvallavatn, which is at approx. 100 m a.s.l., and the area's highest point is at Nesjavellir which is approximately 800 m a.s.l. The afforested area in Ölfusvatn is all below 200 m a.s.l and in Nesjavellir it is below 300 m a.s.l. However, some of the natural forest remnants in Nesjavellir reach above that elevation.

Afforestation in both Nesjavellir and Ölfusvatn started in 1997 when the areas were protected and fenced off from grazing (Figs. 13 and 14). The total protected area of Nesjavellir is 2,065 ha of which 92 ha have been planted with downy birch and 126 ha are natural birch woodlands. Geologically, Nesjavellir forest is located around the Nesjavellir geothermal plant (Figs. 12 and 13) which is associated with the Hengil volcano zone. Franzson (2000) describes the dominant rock formation within the Hengill central volcano as basaltic with subordinate amount of more evolved rocks (lava series). On the other hand, Ölfusvatn (Fig. 14) afforested area covers a total protected area of 316 ha, where the planted areas cover about 109 ha. No remains of natural woodlands were found within the Ölfusvatn area prior to the afforestation. Ölfusvatn is also located close to the Nesjavellir geothermal field, within Hengill area, so it has similar geological features as described above.

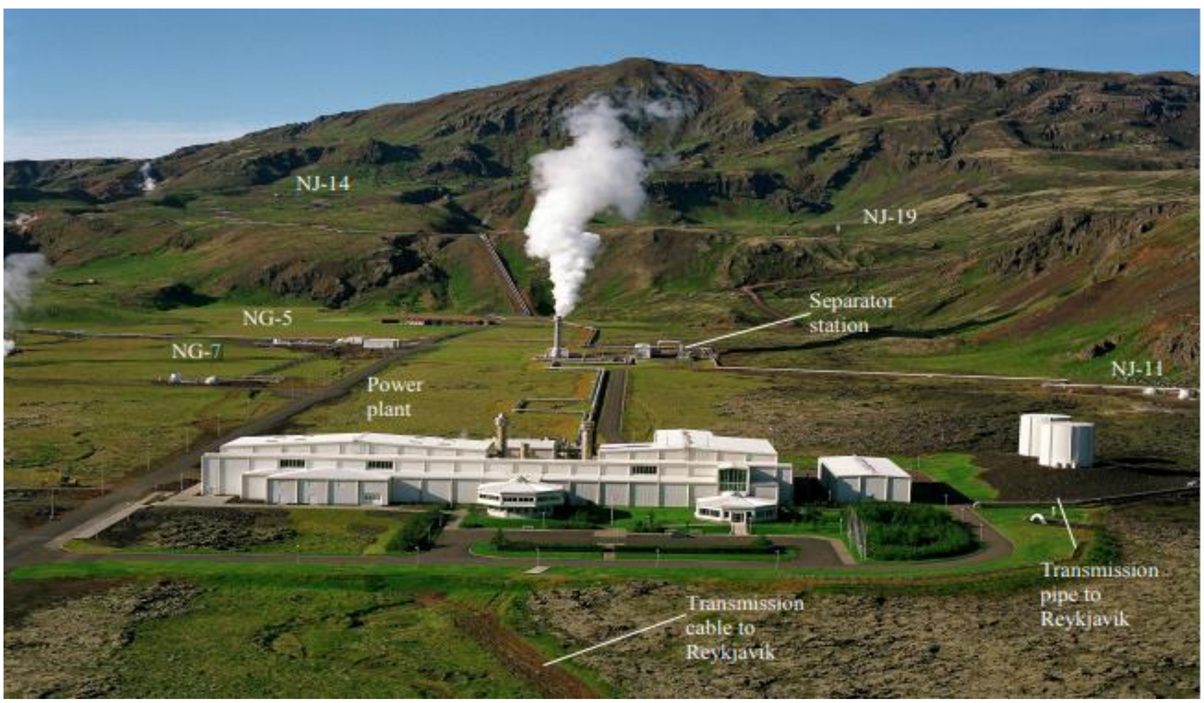


Figure 12. Overview of Nesjavellir geothermal plant site (Photo taken from Ballzus et al., 2000)

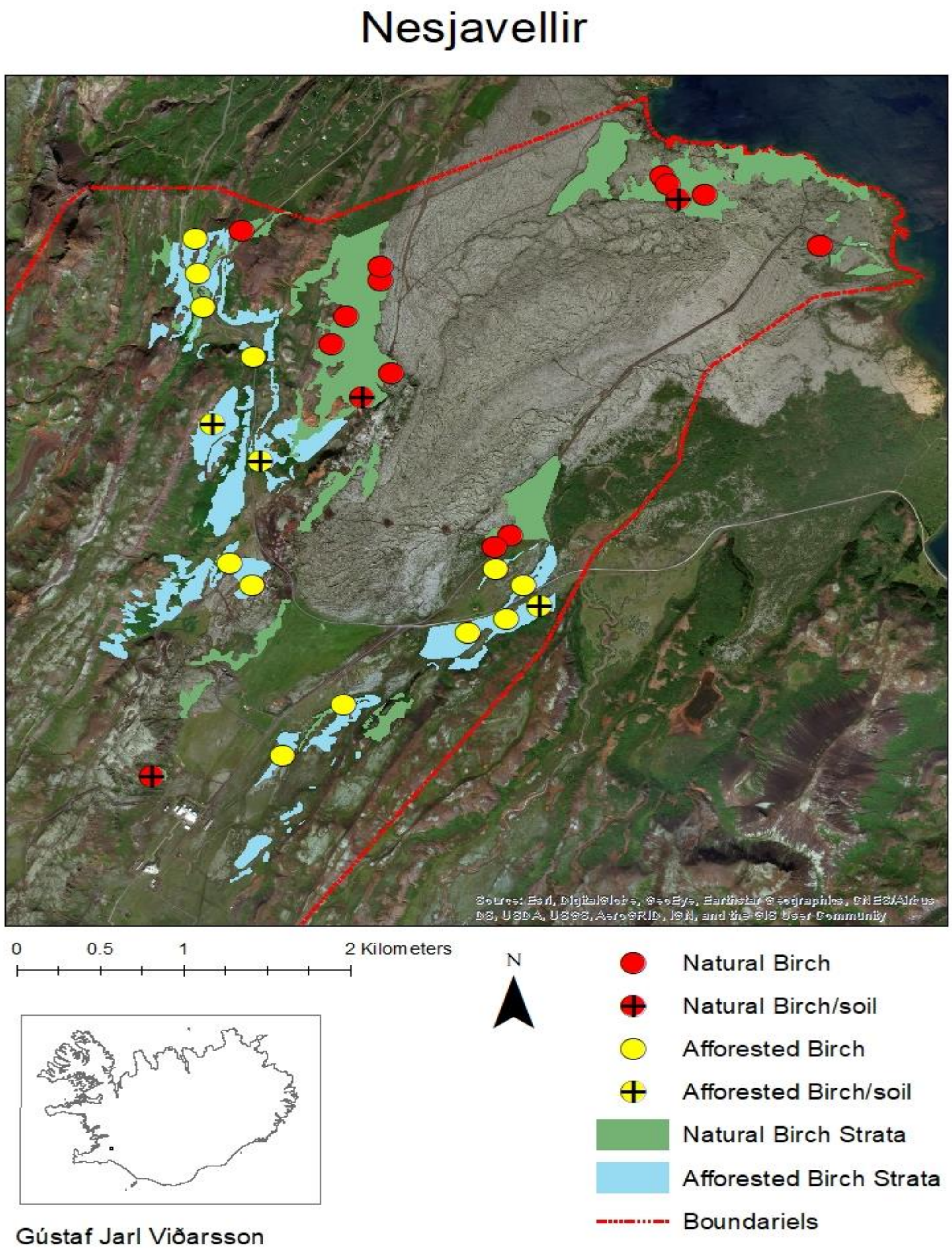


Figure 13. The location of Nesjavellir study site in SW Iceland showing the experimental plots. Forest types are marked by different colours and plots included in this study are marked by crosses.

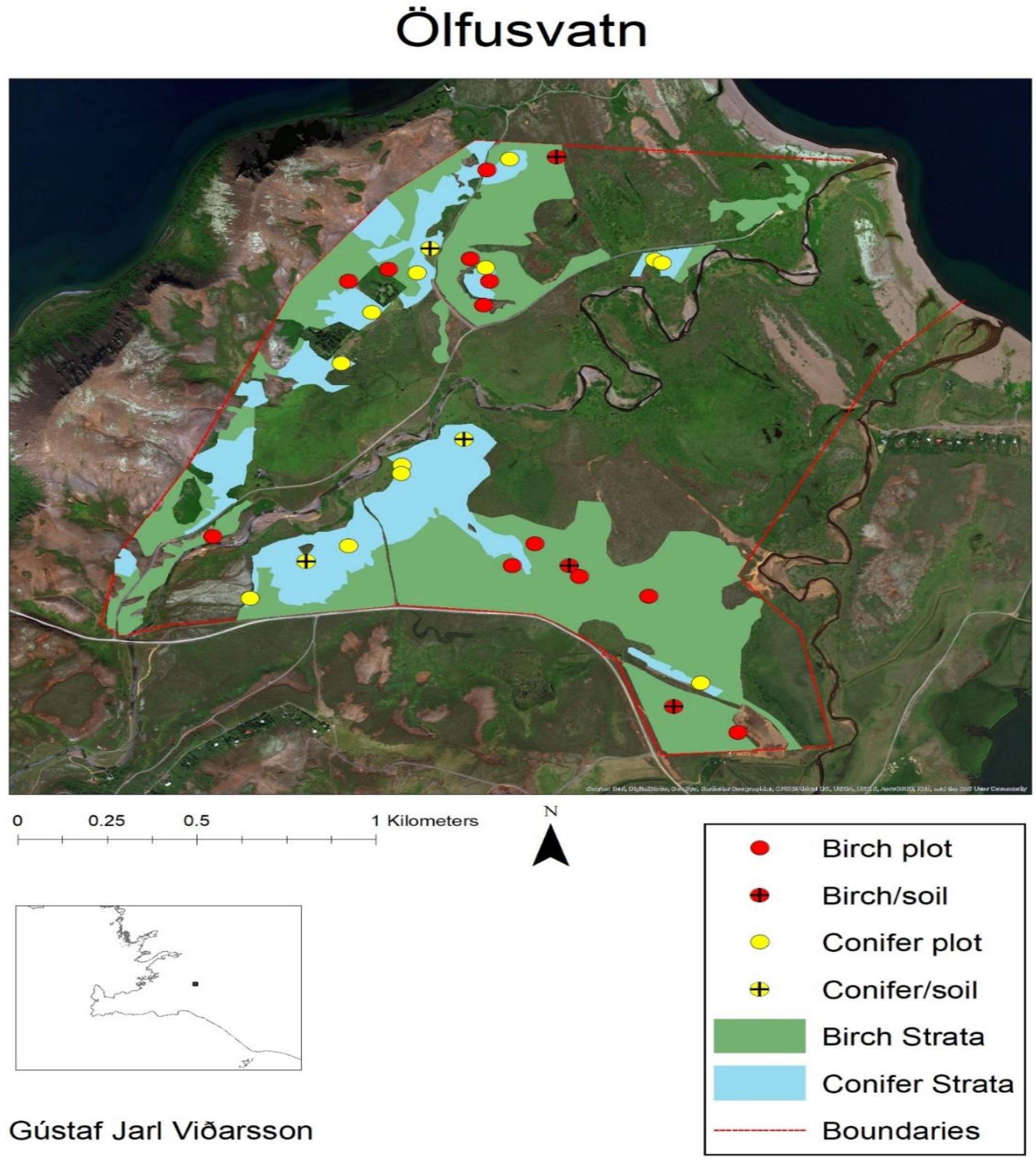


Figure 14. The location of experimental plots in Ölfusvatn experimental site. All trees are planted and forest types are marked by different colours and plots included in this study are marked by crosses.

2.2. Experimental design and data collection

The fieldwork for this project was conducted in collaboration between two M.Sc. students. One student (Gústaf Jarl Viðarsson) had the main responsibility for identifying the set boundaries of forest land and its classification, selecting the study plots, conducting the forest inventory and later scaling up the study's results to landscape level (whole site). For this study, only part of the inventory plots were used for more thorough study on surface cover, litter and soil characteristics, and those are indicated with cross-hatched plots on Figs. 11, 13 and 14. Here below I give a description on how the whole study was conducted; but with emphasis on the parts that are specific to this study.

2.2.1. Forest boundary identification

Several methods and tools are available for identifying and delineating forest land boundaries, including satellite images from optical or radar sensor systems, GIS, aerial photos, GPS, topographic maps and land records (Timothy et al., 2007). In the present study we used aerial photographs provided by Icelandic Forest Service (Figs. 11, 13, 14 and 15). The external boundaries for the study areas were clearly defined on the aerial photographs. Fortunately, the spatial boundaries for Nesjavellir and Ölfusvatn forests are in form of permanently marked fences that could easily be verified through ground-based surveys and collection of 'ground-truth' data with GPS units. We also ensured that the boundaries were not subjected to changes for the duration of the study period. Forests boundaries for Nesjavellir and Ölfusvatn areas were therefore precisely mapped and properly documented from the beginning of the project. This was important to facilitate accurate measurements, monitoring, accounting and verification. For boundary delineation for Heiðmörk study area, we followed the recent works done by Reykjavik town planners and Pic (2009), who explicitly defined the boundaries of this forest. We also used the expert knowledge of the forest managers from Reykjavík Forestry Association who were working in the field at the time of this research who helped in relating the image data to the real feature on the ground.

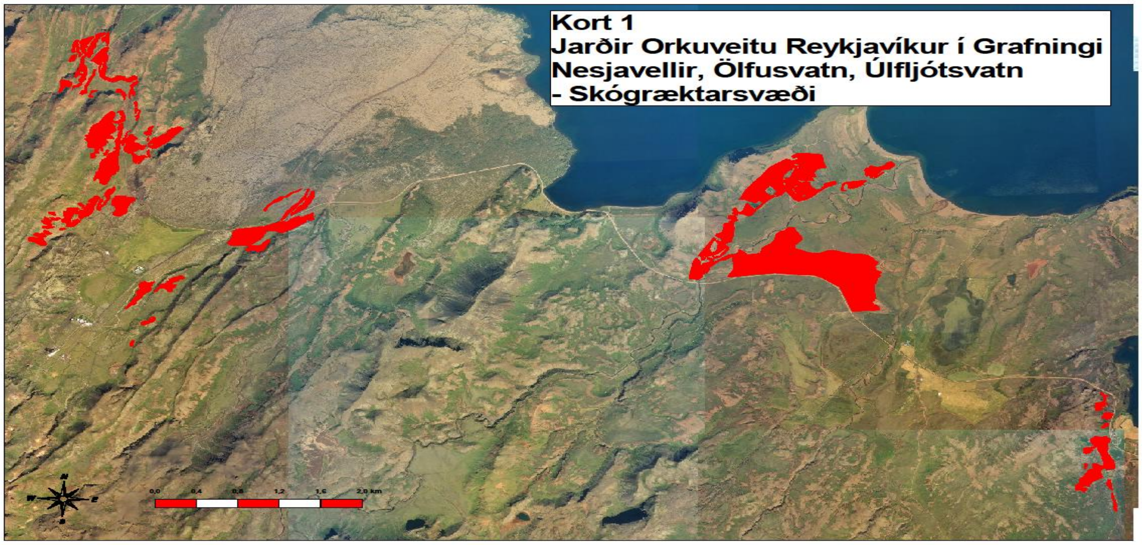


Figure 15. An example of Nesjavellir and Ölfusvatn aerial photograph used for boundary delineation provided by Orkuveita Reykjavikur. Forest areas are marked by red colour.

2.2.2. Stratification of the study areas

The next step after boundaries delineation was stratification, where the lands within each site were divided into more homogeneous units using GIS software Field-Map. The two relatively young afforested areas, (Nesjavellir and Ölfusvatn) were divided into two non-overlapping sub-populations or strata that form relatively homogeneous units based on tree species and whether they are planted or natural (Figs. 13 and 14; Table 14). This was to ensure that measurements are more alike within each stratum than in the sample plot as a whole. This was aimed at facilitating smooth field work and to increase the precision and accuracy of measurements. Before the plots were laid out, 5 m buffer zone was excluded from the forest boundaries and the GIS software was used to randomly place the measurement plots within each stratum. Pic (2009) had already stratified the Heiðmörk forest into four relatively homogenous units representing common features like the stand specificities (tree heights and species). His strata include natural birch (B), planted birch (L), conifer < 3 m (C) and conifer > 3 m (C3). The current study adopted Pic's stratification. However, some amendments were made, especially in the tree heights. The new stratification therefore included; natural birch forest, planted birch trees, conifers over 5 m and conifers under 5 m (Table 2).

Table 2: Forest type (strata) and number of measured plots for each stratum for both afforested and control sites

Sites	Strata	Tree plots	Chronosequence plots	Control plots
Heiðmörk	Natural Birch forest	15	3	3
	Mixed forest	13	3	3
	Conifers over 5 m	19	6	4
	Conifer under 5 m	15	5	3
Nesjavellir	Planted birch forest	15	3	3
	Natural birch forest	15	3	2
Ölfusvatn	Planted conifer	15	3	2
	Planted birch forest	15	3	3
Total	**8**	**122**	**29**	**23**

2.2.3. Inventory plot layout and tree measurements

Mapping of the forests for the aboveground estimate of carbon stock was done using field computer with a range finder, GPS and the software Field-Map (Icelandic Forestry Service, 2017). We established and measured altogether 60 randomly chosen forest plots at Nesjavellir and Ölfusvatn (15 within each stratum) plus 10 nearby treeless control plots at comparable soil, vegetation and landscape characteristics. At Heiðmörk, altogether 62 forest plots (13-19 plots per stratum) were measured, but they had already been randomly placed in another earlier study by Pic (2009). Pic (2009) had in his study used 100 plots, but we randomly selected 62 plots from all those, which explains why the number of plots per stratum is not always the same. We also laid out 13 nearby tree-less control plots at Heiðmörk, 3-4 for each stratum at comparable soil, vegetation and landscape characteristics (Table 3).

At each plot, the central points were found by using a GPS device, plots were marked with a stake and numbered. The trees at Nesjavellir and Ölfusvatn were measured in 50, 100, 150 or 200 m^2 circular plots and the larger plots were used if the stand was sparse, but at minimum 25 trees were to be included in each plot (Fig. 16a). For Heiðmörk, all the plots were 100 m^2 circular plots as established and measured by Pic (2009).

For each tree within each plot, the species was identified and the stem diameter measured at 1.3 m diameter at breast height (dbh), 0.5 m (d 0.5) or at stump height (d 0.1 m) depending on species and size (Fig. 16b). Heights of trees for which diameter is measured were predicted with height/diameter curves based on sub-samples of trees (candidates) that also had height measurement; usually 3-5 per plot.

It should be noted that a different method was used for measuring naturally regenerating downy birch woodlands which are < 2.0 m tall/long. This was done by establishing four equal size subplots within the main plots. One representative mean stem from one of the four subplots was then randomly selected for detail measurements of different variables (height/length, diameter, yearly increment) and the rest of the stems where only counted. The result was then extrapolated to the whole plot.

Either tree core or stem disk from a similar tree outside the plot were taken from diameter at breast height (*dbh)* or d 0.5 or d 0.1 depending on tree species. The samples were taken from the basal area mean tree, if there were more than one species in the plot then a sample was taken from a representative of that species. Width of the annual rings from the disks or cores was measured to 0.001 mm using TSAP-Win software (Copyright © 2003 Frank Rinn, Heidelberg, Rinn Tech). For Nesjavellir and Ölfusvatn additional cores were taken at d 0.1 from dominant trees within each forest plot and their tree rings were counted for determination of planting age. Age determination existed for all the Heiðmörk forest plots by the same method (Ólafur Eggertsson, unpublished data).

The tree-layer variables that were calculated using this data were stand density, basal area, stem volume, stem quality and total tree biomass (stem + coarse roots), and crown cover was also determined. Single tree stem volume and biomass functions for tree species grown in Iceland were used to calculate stem volume and biomass (Snorrason and Einarsson 2006; Snorrason 2010; Jónsson and Snorrason 2018). These functions predict stem volume defined as the volume over bark and above stump to the top of the tree in dm^3 $stem^{-1}$ or dry mass as kg $stem^{-1}$. Total tree biomass was found by adding 25% to the aboveground values to account for the stump and the coarse roots, which is a ratio found for Icelandic planted trees by Snorrason (2002). The current annual increment (CAI) of trees was estimated on the basis of 5 last annual rings taken from trees within the plots and measuring their 5 years' height increment. Mean annual increment

(MAI) of trees was found by dividing standing stem volume or total tree biomass or carbon stock with the plot age found by tree ring analysis. For the purpose of this thesis, I only used basal area and total tree biomass and carbon stock as variables. The details of the above-ground biomass estimation and C sequestration in biomass stocks will be further addressed in another MSc thesis by Gustaf Jarl Viðarsson.

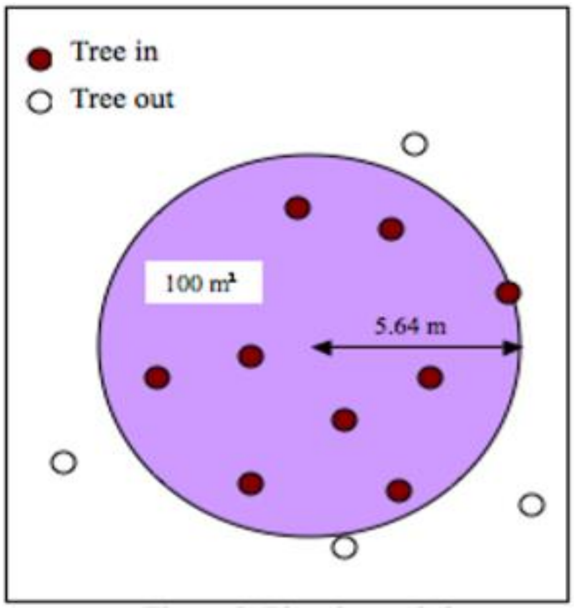


Figure 16 a). Example of sample plot in Heiðmörk. Red symbols indicate trees found within a 100 m^2 sample plot.

b). Forest measurement in Heiðmörk using field computer (*Photo: Kimmo Vanhavifta*)

2.3. Surface- and below-ground field measurement and laboratory work

For the sub-sample of plots that were selected for further measurements of surface and soil characteristics (Table 2; Figs. 11, 13, and 14) more measurements took place.

2.3.1. Harvesting bush layer

Bush or shrubs layer samples (woody plants >50 cm and < 1.3 – 2.0 m in diameter and height) were collected from plots with only non-native tree species (native tree species were determined following the definition by Eysteinsson (2017)). Within the circular plots, 4 m^2 circular subplots were established at a distance of 3 m north and south from the centre, using 1.13 m long measurement tape and all the bush/shrub stems were counted moving in a full circle within the subplots. One main stem was then harvested and the samples were transported to the lab at Keldnaholt and oven dried at 70°C for at least 72 hours. After separating leaves and wood, the samples were weighed and their dry mass (0.00g) was recorded after which the samples was discarded.

2.3.2 Ground vegetation (GV)

The percentage cover of ground vegetation main taxonomic classes (moss, ferns, monocot grasses, dicot herbs, dicot dwarf bushes) and surface characteristics (rock, unvegetated and deadwood) was estimated visually using 20 x 50 cm frame within each 4 m^2 subplot used for measuring the bush/shrub layer. The ground vegetation was then harvested using electric grass cutter, including the living part of moss layer (Fig. 17). In the lab, it was oven dried at 70 °C for 48 hours, weighed its dry mass in (0.00g) and was discarded afterward.

Figure 17. Ground vegetation was collected from within a 20 x 50 frame using electronic grass cutter (*Photo: Kimmo Vanhavifta)*

2.3.3 Woody and fine litter layer

Coarse woody litter (fallen trees, branches and pieces of wood >2 cm and <10 cm in diameter) was extracted/cut at the 20 x 50 cm frame's edge and put in paper bag and transported to the lab, oven dried at 70°C for 72 hours, its dry mass weighed in (0.00g) and the sample was discarded. The fine litter (O-horizon) and dead moss were harvested from within 20 x 20 cm subplot and put in paper bag (Fig.18). Here, difficulties in accurately separating the litter and mineral soils were sometimes experienced. Depths of litter layer as well as depth of soil below were registered using measurement tape and 1 m long probe, respectively. In the lab, the fine litter sample was oven dried at 40°C for 48 hours, roots removed and weighed separately.

Approximately 5 g subsamples of litter were taken for pH measurements. The pH was measured by adding 25 ml of de-ionised water to the litter sample to give an 8:1 volume ratio of water to litter sample. This was done in two replicates. Shaking was done using the laboratory shaker for at least 2 hours and centrifuged until there was a clear separation of the sediment and the supernatant. The supernatant was then measured for pH using the pH meter (Oakton/pH/mV/°C, Oakton Company, Australia) with adequate calibration (using buffer solutions of pH 4 and pH 7). Another 5 g subsamples of litter were taken for ball milling to obtain finely ground, homogeneous samples for chemical analysis by direct combustion procedure using (Vario MAX CN Element Analyser, Elementar Analysensystem GmbH, Germany) for determination of total C and total N concentration.

Figure 18. Fine litter was harvested from a 20 x 20 cm frame and in the middle litter depth was measured (*Photo: Kimmo Vanhavifta)*

2.3.4 Soil sampling for calculation of soil bulk density (BD)

One soil samples per plot were collected for BD within a frame of 20 x 20 cm subplot, (one per main plot, the one within the deeper soil) using 30 cm long and 5.0 cm diameter soil corer, and separated into 0-5, 5-10, 10-20 and 20-30 cm layers (Figs. 19a and b). Samples were placed in plastic bags and oven-dried in the lab at 40°C for at least 48 hours, sieved through 2 mm to separate fine fraction (<2 mm), coarse fraction (>2 mm) and roots. Their dry mass was weighed (0.00g) separately. Volume of the coarse fraction of soil sample (stoniness) and roots was measured by water displacement method. Soil bulk density was then calculated using this formula;

$BD = M_s / (V_t - (V_s + V_r))$,

where *BD* is in g cm^{-3}, M_s is the weight of the oven-dried soil sample in g, V_t is the total sample volume in cm^3, V_s is the volume of the cores fraction of soil sample and V_r is the volume of fine roots in cm^3. V_r was estimated from its dry mass, using a factor of 4.487 cm^3 g^{-1} (Bjarni D. Sigurdsson, unpublished data).

Figure 19 a). Soil sampling for BD at treeless control site at Nesjavellir

b). Soil sampling for BD at Ölfusvatn afforested site. (*Photos: Kimmo Vanhavifta)*

2.3.5 Soil sampling for SOC

Eight samples for chemical analysis were taken from ca. 3 m north, south, east and west of the central point with a 2 cm wide and 30 cm long corer, as well as from the same frame where fine litter was harvested (Fig. 20). The samples were divided into 0-5, 5-10, 10-20 and 20-30 cm layers from each subplot and all samples per depth (layer) were combined and put in marked plastic bag and transported to the lab, oven-dried at 40°C for 48 hours and sieved through 2 mm. Approximately 5 g subsample of fine soil was taken for ball milling and chemical analysis and for dry matter determination. The finely ground soil samples were placed in a redcap tube and taken for chemical analysis (C and N measurement) and dry matter after drying at 103°C for 48 hours. Another 10 g of soil sample from the top layer (0-5 cm) were taken for pH measurements. The pH was measured by adding 50 ml of de-ionised water to the sample to give a 1:5 volume ratio of water to soil sample. This is done in 2 replicate. Shaking was done using the laboratory shaker for at least 2 hours and allowed to settle into supernatant solution. The supernatant was then measured for pH using the pH meter (Oakton/pH/mV/°C, Oakton company, Australia) with adequate calibration (using buffer solutions of pH 4 and pH 7).

Figure 20. Soil sampled from beneath the litter layer for chemical analysis from Nesjavellir. (*Photo: Kimmo Vanhavifta)*

2.3.6 Soil mass correction (SMC)

First, the carbon stock was calculated to 30 cm depth without soil mass correction. But, since areas with different land use (afforested and treeless) apparently had differences in total soil mass and soil bulk density of different layers, calculating the carbon stock based on fixed depth could lead to errors (see the Introduction chapter 1.9). Since BD samples included undisturbed depth profiles down to 30 cm depth where all the soil from each layer had been carefully conserved during laboratory processing, the soil dry mass in the forest plots was corrected to equivalent soil mass from the corresponding treeless control plots, directly using the average mass ratio between control and afforested soils for each layer separately, rather than to calculate such factors from the BD values as was done by e.g. Bracena et al. (2014) and Leblans et al. (2016). The correction factors were calculated directly from the average mass ratio between treeless and afforested soils for each layer separately, as showed below;

$$CF = \sum \frac{TL\alpha}{AF\alpha}$$

where CF is the correction factors, Ʃ is the sum, TLα is soil mass at layer α in treeless site (g DM m^{-2}) and AFα is soil mass at layer α in afforested sites (g DM m^{-2}). This was noted to be a sound method as long as the calculation only includes the fine fraction of the soil and sites do not significantly differ in stoniness (Bárcena et al., 2014). Further, the soil mass correction factors were based on averages for each depth layer across all control and forest plots.

2.4 Calculations of C-stocks

C content in tree biomass (Tc), bush layer (Bc) and ground vegetation (Gc)

To estimate woody biomass carbon (total tree carbon aboveground and coarse roots, (Tc); bush layer; coarse woody litter; dwarf bushes), I used 50% C concentration of woody biomass (Snorrason et al., 2002). Undecomposed woody biomass is always close to this value (Chapin et al., 2002). To estimate the carbon contents of deciduous vegetation, moss and fine roots from oven dry mass we used 40% C concentration which is based on Icelandic research (Snorrason et al., 2002; Leblans, 2016)

C content in fine litter and soil

Chemical analysis was used to find site-specific C concentrations for both litter and individual soil samples. The C content in fine litter was then estimated by multiplying its dry mass by the measured C concentration and converting it to g C m^{-2} using the frame size. The rate of C sequestration was given in g C m^{-2} yr^{-1} and C stock in g C m^{-2}, BD is soil bulk density (g cm^{-3}) and S_{dm} is the corrected soil mass (g).

Total ecosystem C stocks (TOC)

The total ecosystem C stocks per unit area were estimated as;

$TOC= Tc+SOC+Lc+Gc+Bc+Rc$ *(g C* m^{-2}*),* where *TOC* is total organic carbon, *Tc* is carbon content is above-ground tree biomass and coarse roots, SOC is soil organic carbon, *Lc* is carbon content is litter, *Gc* is carbon content in vegetation, *Bc* carbon content is bush layer and *Rc* is carbon content is fine roots.

2.5 Data handling and statistics

SAS software (SAS system version 9.4) was used to calculate averages and standard error for the soil and surface cover data and to analyse differences across treeless sites, different tree species or other fixed attributes. The latter was done with one-way analysis of variance (ANOVA). When number of groups was > 2 then Fisher's Least Significant Difference tests were used to derive pairwise differences, when the ANOVA was significant ($p<0.05$). This analysis of variance was done for all vegetation cover (dwarf bush, shrubs, ferns, moss and grasses), soil chemical properties (C and N stocks, soil pH, percentage of weight C and N, C/N ratios) and physical properties (bulk density, stoniness soil depths,) and other ground characteristics (rock, deadwood and bare soil).

For the forest data, Excel spread sheets were used to calculate means and standard errors, including stand density, basal area, stem volume, total tree biomass, tree heights and diameter at breast height. I also used the same model of one-way ANOVA followed by Fisher's LSD to compare differences between vegetation cover, soil properties and ecosystem C and N-stocks within afforested and compared with treeless sites.

Relationships between different variables and age, forest basal area or stem volume were analysed by linear regression using Sigma Plot version 12.0.

3. Results

3.1. Differences in vegetation composition and soil properties at three treeless control sites

The result show that for the majority of the variables studied (vegetation composition, soil physical and chemical properties as well as unvegetated rocky surfaces), there were no significant differences between the three control sites (Table 3), indicating that they were all comparable (similar) before the afforestation started.

The vegetation of the three treeless sites was, on average, composed of 38% dicotyledonous plants (dwarf bushes and herbs), 23% monocotyledonous plants (ferns, grasses) and 36% mosses (Table 3). Only the grass cover was significantly less in Heiðmörk treeless site, compared to the other two sites. The treeless plots at the three sites were on average 28% unvegetated and on average the summed surface cover of the different vegetation classes was 72% on average. The dry mass of vegetation was not significantly different across the three treeless sites and on average had a mean value of 518 g m^{-2}, but Heiðmörk had significantly less fine roots (0.4 g m^2) than the other two sites, which fits with Heiðmörk’s lower average grass cover.

The soil chemical properties (pH, C/N ratio, C and N concentrations) as well as soil physical properties (bulk density, stoniness, soil depth, litter depth and litter dry mass) did not vary, except soil dry mass in the top 30 cm was significantly lower in the Nesjavellir treeless control plots (Table 3).

Similar to many other variables measured, there were no significant difference in the ecosystem C-stocks among the three control sites (Table 4). The average C-stock in ground vegetation was 202 g C m^{-2}, litter was 336 g C m^{-2}, fine roots was 0.5 g C m^2 and soil within 30 cm depth was 10.429 g C m^2. Nitrogen was only measured in soil (mean value 661 g C m^2) and litter (mean value 6.4 g C m^{2}) and did not significantly vary across all the three sites.

Table 3. Comparison of ground vegetation and soil properties in treeless sites in SW-Iceland in autumn 2017. Values are means and standard errors of Heiðmörk (n = 13 plots), Ölfusvatn (n = 5 plots) and Nesjavellir (n = 5 plots). Significant values are highlighted as bold ($P < 0.05$). Ground vegetation cover in percentages are denoted by D_{bs} (dwarf bush), H_r (herbs), F_r (ferns), G_r (grass) and M_o (moss). Surface classes are denoted by R_k (rock), U_v (Unvegated) and dry mass of biomass are symbolised by V_{dm} (vegetation) and R_{dm} (roots). Soil physical properties are symbolised by L_{dm} (litter dry mass), S_{dm} (soil dry mass), BD (soil bulk density), S_t (percentage of stones in the soil), S_{dt} (soil depth), L_{dt} (litter depth), and soil chemical properties are symbolised by pH_s (soil pH), pH_L (litter pH), C/N_s (carbon to nitrogen ratio in soil), C/N_L (carbon to nitrogen ratio in litter), C_s (percentage weighted carbon in soil), N_s (percentage weighted nitrogen in soil), C_L (percentage weighted carbon in litter) and N_L (percentage weighted nitrogen in litter). Different letters indicate the significant variations among treeless sites (one-way ANOVA, followed by Fisher's LDS used to test significance).

	Heiðmörk – treeless control		**Ölfusvatn treeless control**		**Nesjavellir treeless control**		**ANOVA**
Variables	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**	**P-value**
Vegetation cover							
D_{bs} (%)	48.5	8.30	50.0	11.0	32.5	11.8	0.57
H_r (%)	7.00	1.10	7.00	3.0	6.20	3.80	0.98
F_r (%)	0.40	0.40	4.00	3.0	0.00	0.00	0.10
G_r (%)	**13.8**	**2.40**	**30.0**	**8.5**	**42.5**	**13.6**	**0.01**
M_o (%)	39.2	6.30	31.0	8.4	37.5	17.6	0.81
Ground Surface classes							
U_v (%)	28.8	7.00	25.0	7.20	29.0	7.00	0.10
R_k (%)	0.10	6.40	0.00	0.00	0.00	0.00	0.28
Vegetation* surface cover (%)	71.1		75		71		
Biomass							
V_{dm} (g DM m^{-2})	438	64	620	221	496	103	0.53
R_{dm} (g DM m^{-2})	**0.4**	**0.1**	**2**	**1.0**	**1.7**	**1**	**0.04**
Soil physical properties							
Li_{dt} (cm)	4.1	0.7	5.7	1.4	2.6	1	0.21
S_{dt} (cm)	56.4	5.9	79.7	7.2	46.7	13	0.06
Li_{dm} (g DM m^{-2})	803	197	859	132	1179	307	0.59
S_{dm} (g DM m^{-2})	**150333^{ac}**	**9368**	**154843^{bc}**	**9436**	**103417^{bc}**	**6670**	**0.01**
S_t (%)	10.3	4.3	1.6	1.4	10	6.6	0.46
$BD_{0\text{-}30}$ (g cm^{-3})	0.49	0.03	0.49	0.03	0.37	0.03	0.10

*Percentage of surface vegetation cover was calculated by subtracting the percentage of unvegetated from vegetated surfaces.

Table 3. Continued

Variables	Heiðmörk – treeless control Mean	SE	Ölfusvatn treeless control Mean	SE	Nesjavellir treeless control Mean	SE	ANOVA P-value
Soil chemical properties							
pH_L	4.6	0.08	4.5	0.09	4.4	0.06	0.22
$pH_{s,(0\text{-}5cm)}$	5.4	0.07	5.3	0.07	5.1	0.2	0.10
$C/N_{s,\ (0\text{-}30\ cm)}$	19.2	1.3	18.07	1.5	22.2	3.5	0.44
C/N_L	47.5	5.5	45.5	4.5	49.2	11	0.95
C_s (%)	7.4	0.8	8.5	0.9	9.06	1.1	0.48
N_s (%)	0.5	0.05	0.5	0.05	0.6	0.1	0.68
C_L (%)	30.3	3.8	36.9	2.4	39.7	1.1	0.48
N_L (%)	0.6	0.07	0.8	0.1	0.9	14.1	0.08

Table 4. Ecosystem C-stocks and soil N at three treeless sites in SW-Iceland in autumn 2017. Values are means and standard errors of Heiðmörk (n = 13 plots), Ölfusvatn (n=5 plots) and Nesjavellir (n=5 plots). VC denotes carbon in vegetation, LC carbon in litter, RC carbon in roots, SOC soil organic carbon, LN nitrogen in litter and SON soil organic nitrogen.

Variables ($g\ C\ m^{-2}$)	Heiðmörk – treeless control Mean	SE	Ölfusvatn treeless control Mean	SE	Nesjavellir treeless control Mean	SE	ANOVA P-value
VC	175	26	248	88	185	35	0.50
LC	265	56	321	41	422	107	0.30
RC	0.19	0.04	0.73	0.33	0.46	0.27	0.90
$SOC_{0\text{-}30\ cm}$	10031	847	12303	930	9592	852	0.22
LN	4.9	1.1	6.2	0.7	8.1	1.6	0.20
$SON_{0\text{-}30cm}$	653	60	795	55	550	71	0.15

3.2. The mean changes following afforestation across the three sites

3.2.1. Tree characteristics

During the study, I grouped the tree species into five main forest types which included; conifer >5 m, conifer <5 m, mixed, planted birch and natural birch (Table 5). Conifers >5 m tall, mixed and old growth birch forests were only found in Heiðmörk, while conifer <5 m were found in both Heiðmörk and Ölfusvatn afforestation sites. Birch forest, either planted or naturally regenerated, were found in all the sites but most predominantly (71%) were found in Nesjavellir.

For all the measurement plots, I included data on tree characteristics including stand density, basal area, stem volume, total tree biomass, tree heights and diameter at breast height and their mean values are shown in Table 5. At Ölfusvatn, conifer < 5 m had more stems per hectare than in Heiðmörk, while in Nesjavellir natural birch was denser than planted stands (i.e. 93% vs 07%). Across all sites, pure conifer had an average tree diameters ranging from 10.7 to 11.0 cm and the diameter of planted birch ranges from 1.9 to 2.9 cm, while the mean diameter for planted birch had higher diameter (ca. 50%) than natural ones in Nesjavellir.

Planted birch in Nesjavellir were taller that natural ones and even old growth trees by 1.6 and 0.1 cm, respectively, while conifer > 5 m were taller than conifer < 5m and mixed conifer by 6.8 and 5.7 cm, respectively (Table 5). Conifer > 5 m had the highest mean total tree biomass, while conifer > 5 m found in Heiðmörk were 58% more productive than those found in Ölfusvatn forest. Birch planted in Nesjavellir was 30% more productive than those growing in Ölfusvatn.

Table 5. Description of forests characteristics across five forest types at three sites in SW-Iceland in autumn 2017. Values are means and standard errors of Heiðmörk (n = 17 plots), Ölfusvatn (n = 6 plots) and Nesjavellir (n = 6 plots). NF denotes forest types not found growing and NM denotes forest type found growing but not included in the measurement.

	Heiðmörk		**Ölfusvatn**		**Nesjavellir**	
Forest type	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
Stand density (trees ha^{-1})						
Conifer > 5 m	2575	774.0	NF	-	NF	-
Conifer < 5 m	1920	297.2	2200	346.4	NF	-
Mixed	933	290.6	NF	-	NF	-
Planted birch	NM	-	1467	371.2	1400	284
Natural birch	NM	-	NF	-	17,467	8,042
Birch, old	10833	5554.7	NF	-	NF	-
Basal area(m^2 ha^{-1})						
Conifer > 5 m	42.9	6.74	NF	-	NF	-
Conifer < 5 m	16.8	4.32	13.55	4.05	NF	-
Mixed	5.9	3.84	NF	-	NF	-
Planted birch	NM	-	0.54	0.19	1.5	0.8
Natural birch	NM	-	NF	-	4.1	1.3
Birch, old	5.8	0.55	NF	-	NF	-

Table 5. Continued

Stem volume (m^{-3} ha^{-1})						
Conifer > 5 m	217.0	35.07	NF	-	NF	-
Conifer < 5 m	54.1	14.31	42.82	12.71	NF	-
Mixed	24.3	17.74	NF	-	NF	-
Planted birch	NM	-	0.62	0.25	2.0	1.04
Natural birch	NM	-	NF	-	6.1	4.0
Birch, old	7.08	0.78	NF	-	NF	-
Total tree biomass (g DM m^{-2})						
Conifer > 5 m	17563	2783.6	NF	-	NF	-
Conifer < 5 m	6038	1585.1	4712	1446.2	NF	-
Mixed	2189	1465.0	NF	-	NF	-
Planted birch	NM	-	154.6	44.0	289	120
Natural birch	NM	-	NF	-	789	445
Birch, old	1157.7	196.2	NF	-	NF	-
Diameter (cm)						
Conifer > 5 m	11.0	2.2	NF	-	NF	-
Conifer < 5 m	10.7	1.2	6.1	3.1	NF	-
Mixed	11.3	1.7	NF	-	NF	-
Planted birch	NF	-	1.9	0.3	2.9	0.9
Natural birch	NF	-	NF	-	1.9	0.3
Birch, old	3.3	0.6	NF	-	NF	-
Dominant height (m)						
Conifer > 5 m	13.0	1.13	NF	-	NF	-
Conifer < 5 m	6.2	0.6	6.0	0.5	NF	-
Mixed	7.3	1.6	NF	-	NF	-
Planted birch	NM	-	3.3	0.5	3.7	0.9
Natural birch	NM	-	NF	-	2.1	0.2
Birch, old	3.6	0.6	NF	-	NF	-
Total tree C-stock (g C m^{-2})						
Conifer > 5 m	5746	1931	NF	-	NF	-
Conifer < 5 m	3019	886	2356	723	NF	-
Mixed	1094	732	NF	-	NF	-
Planted birch	NM	-	77	22	144	60
Natural birch	NM	-	NF	-	394	228
Birch, old	579	98	NF	-	NF	-

The average tree C-stock across all the forest types at all the three sites was 1,877 g C m^{-2}. There was higher woody C-stock in conifers (both < and > 5 m; 3,707 g C m^{-2}), followed by mixed (1,094 g C m^{-2}) and deciduous (both planted and natural = 308 g C m^{-2}). Heiðmörk had the largest amount of standing C-stock in aboveground biomass and coarse roots per m^{-2} (64%), followed by Ölfusvatn (30%) and Nesjavellir (6%) as shown in Table 5.

3.2.2. Vegetation composition

Following afforestation, vegetation composition changed. Dwarf bush and shrubs cover significantly decreased from 42.6% in treeless sites to 23% in afforested sites, but the fern cover significantly increased in the afforested sites (Table 6). There was a strong trend for more cover of herbs ($P = 0.06$) and grasses ($P = 0.08$) following afforestation, but the moss layer did not significantly change.

The surfaces covered by dead wood significantly increased at afforested sites, as was expected, and that explains most of the observed reduction in total ground vegetation cover (Table 6). There was a strong trend for less rocky surfaces ($P = 0.05$) in the afforested areas but the cover of unvegetated open soil was not different.

The bush layer, measuring >50 cm in height and lower than ca. 1.3-2.0 m, comprised mostly of willow (*Salix* spp.) accumulated only on average net biomass of 14.0 g DM m^{-2}, while no bush layer was encountered on the treeless control sites (Table 6). Although not significant, ground vegetation dry mass showed an average decreasing trend of ca. 134 g DM m^{-2} ($P = 0.10$) in the afforested sites, but average fine root biomass did not change (Table 6).

3.2.3. Soil properties

Soil depth was not significantly different between treeless and afforested sites and was on average 60 cm (Table 7). Soils in both treeless and afforested sites were on average 30 cm deeper than the sampling depth. The soil mass, when summed for all measured soil layers, showed no significant change following afforestation, but there was a significant increase in litter biomass and litter depth following afforestation by 760 g DM m^{-2} and 1.7 cm, respectively (Table 7). The influence of afforestation on the mean bulk density (BD) in the top 30 cm of soil also showed no significant difference after afforestation, and stoniness in the top 30 cm was not significantly different and was on average 7.2%.

Litter and mineral soil in the top 5 cm did not show a significant change in their pH values across all forest plots (Table 7). The average C/N ratio of the mineral soil and the percentage N concentration in soil (0-30cm) did not change significantly, however, the C/N ratio of the litter layer had a strong trend for lower values in the afforestation areas (P = 0.05). The carbon concentration in the 30 cm of mineral soil showed a trend for increased values following afforestation (P = 0.10). Also, the percentage concentration of C and N in litter significantly increased following afforestation (Table 7).

Table 6. Changes in vegetation composition and other surface characteristics following afforestation at three sites in SW-Iceland in autumn 2017. Values are means and standard errors for treeless (n = 23 plots) and afforested sites (n = 29 plots). Significant values are highlighted as bold (P < 0.05). Vegetation cover are denoted D_{bs} (dwarf bush), H_r (herbs), F_r (ferns), G_r (grass), M_o (moss). Surface classes are symbolised by Deadwood (D_w), R_k (rock), U_v (Unvegated) and dry mass of biomass are denoted by V_{dm} (vegetation) R_{dm} roots.

	Treeless sites		**Afforested sites***		
Variables	**Mean**	**SE**	**Mean**	**SE**	**P-values**
Vegetation cover					
D_{bs} (%)	**42.6**	6.2	**22.5**	5.2	**0.01**
H_r (%)	7.5	1.1	13.9	3.2	0.06
F_r (%)	**1.1**	0.7	**7.5**	1.7	**0.01**
G_r (%)	19.8	3.4	30.5	4.7	0.08
M_o (%)	37.2	4.9	40.5	4.3	0.60
Ground vegetation and surface classes					
D_w (%)	**1.8**	1	**14.1**	3.7	**0.01**
R_k (%)	**8.3**	3.9	**0.7**	0.7	0.05
U_v (%)	25.9	5.2	25.2	5.2	0.92
Vegetated (%)	64.3		60		
Vegetation biomass					
Bush layer*	-	-	14.0	9.7	-
V_{dm} g DM m^{-2}	490	63.6	356	50.0	0.10
R_{dm} g DM m^{-2}	1.0	0.3	1.2	0.2	0.54

* No bush layer was not encountered on the treeless control sites

Table 7. Changes in soil physical and chemical properties following afforestation at three sites in SW-Iceland in autumn 2017. Values are means and standard errors of treeless (n = 23 plots) and afforested sites (n = 29 plots). Significant values are highlighted as bold (P < 0.05). Soil physical properties are symbolised by L_{dt} (litter depth in cm), S_{dt} (soil depth in cm), L_{idm} (dry mass of litter in g DM m^{-2}), US_{dm} (uncorrected soil mass in g DM m^{-2}), CS_{dm} (corrected soil mass in g DM m^{-2}), S_t (percentage of stones in the soil), BD (soil bulk density in g cm^{-3}). Chemical properties are denoted by pH_s (soil $pH_{0\text{-}5\ cm}$), pH_L (litter pH), C/N_s (carbon to nitrogen ration in soil 0-30cm), C/N_L (carbon to nitrogen ratio in litter), C_s (percentage weighted carbon in soil), N_s (percentage weighted nitrogen in soil), C_L (percentage weighted carbon in litter) and N_L (parentage weighted nitrogen in litter).

Variables	Treeless sites Mean	SE	Afforested sites Mean	SE	P-values
Soil physical properties					
Li_{dt}	4.2	0.5	5.9	0.5	**0.03**
S_{dt}	60	4.9	60	3.8	0.65
Li_{dm}	884	132	1644	168	**<0.01**
US_{dm}	141115	7101	131753	59987	0.32
CS_{dm**}	-	-	151,053	6,970	0.33
S_t	8.3	2.8	6.0	2.9	0.63
BD	0.47	0.02	0.43	0.02	0.19
Soil chemical properties					
pH_s	5.32	0.05	5.21	0.05	0.13
pH_L	4.54	0.05	4.63	0.05	0.26
C/N_s	16.1	0.46	15.9	0.38	0.74
C/N_L	47.2	3.63	38.9	2.36	**0.05**
C_s	34.6	2.3	39.3	1.6	0.10
N_s	0.50	0.03	0.56	0.02	0.13
C_L	33.7	2.3	38.7	1.06	**0.04**
N_L	0.72	0.05	1.17	0.16	**0.02**

** Soil mass for treeless sites was not corrected

3.2.4 Soil mass correction

To avoid underestimation of C-stock in soil, the soil dry mass for afforested sites was corrected for each individual layer using correction factors (Fig. 21). Birch forest had correction factors of 1.32, 1.45, 1.09 and 0.97 g DM m^{-2} for 0-5, 5-10, 10-20 and 20-30 soil depths, respectively, and conifers and mixed forests combined had correction factors of 1.32, 1.18, 1.25 and 1.01 g DM m^{-2} for the same depths as birch forest.

The average uncorrected soil mass in the top 30 cm of soil for both afforested and treeless was 136,434 g DM m^{-2} and 19,300 g DM m^2 was added to the afforested soil mass (Table 7). The mass correction affected mainly the top soil layers (0-10cm) in afforested sites (Figure 21).

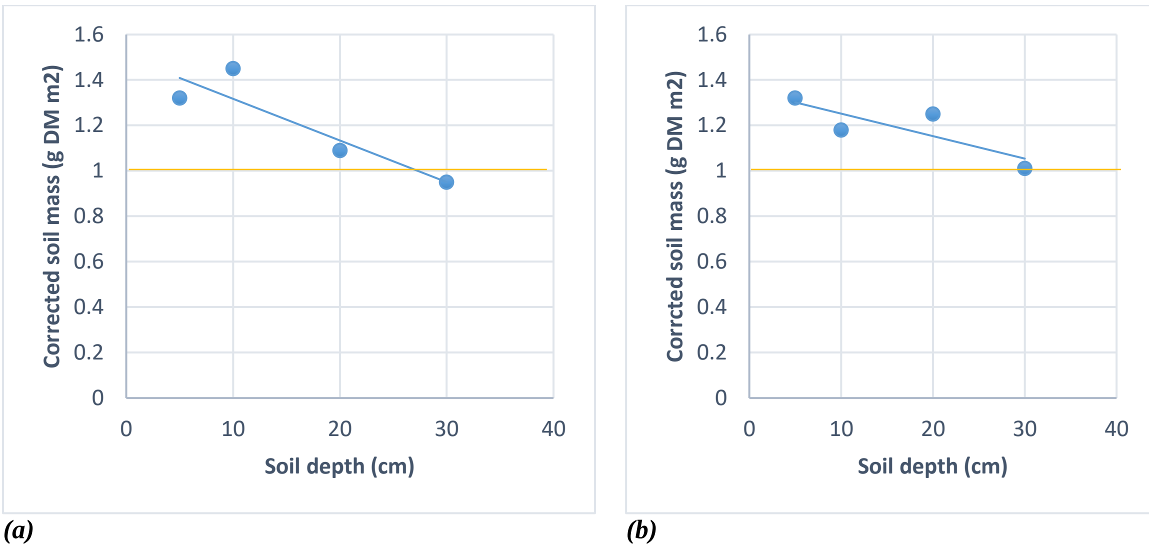


Figure 21. Effects of tree roots on soil mass in top layer (0-30 cm) depth following afforestation at three forested sites. Letter (a) denotes birch and (b) represent conifers and mixed forests.

3.2.5. Ecosystem C-stocks and soil N

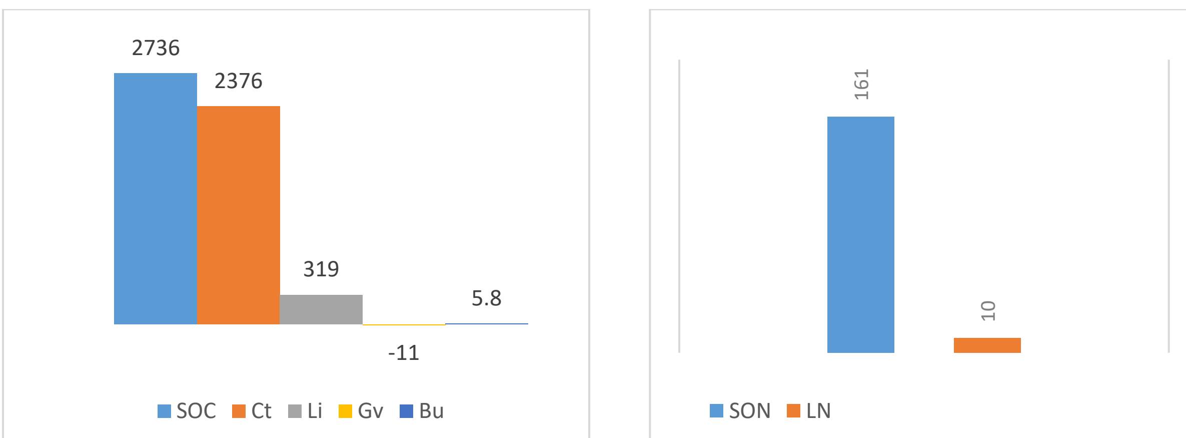
The total mass-corrected SOC-stock in the top 30 cm of soil and litter layer significantly increased following afforestation (Table 8), with higher percentage change measured in litter (+34%; P=0.001) and lower in soil (+10%; P=0.003; Table 6; Fig. 22). Similar to soil mass, the carbon stock change was not significantly different when not corrected for mass differences (Table 7). It is also worth noting that the average C stock change in ground vegetation showed a strong decreasing trend (P=0.07), but fine roots showed no significant difference. Since there was no bush layer on the treeless control sites, the increase in the bush layer was a net increase (Table 8; Fig. 22).

It came as a surprise that the soil organic nitrogen in the 0-30 cm was significantly increased by +10% or 161 g N m^{-2} and almost doubled in litter (+46%) following afforestation (Table 8; Fig. 22).

Table 8. Changes in ecosystem C-stocks and soil N following afforestation. Values are means and standard errors of n = 23 (treeless) and n=29 (afforested) plots. Significant values are highlighted as bold (P< 0.05). Carbon in trees (aboveground and coarse roots) is represented by (Ct), soil organic carbon (SOC g C m^{-2}), uncorrected soil organic carbon (USOC** g C m^{-2}), soil organic nitrogen (SON g C m^{-2}), carbon in vegetation (G_v g C m^{-2}), carbon in litter (L_iC g C m^{-2}), nitrogen in litter (L_iN g C m^{-2}), carbon in roots (R_t g cm^{-2}) and carbon in bush (B_uC g cm^{-2})

Variables	Treeless sites Mean	SE	Afforested sites* Mean	SE	P-values
Ct	-	-	2376	668	-
SOC	**10429**	574	**13165**	619	**<0.03**
USOC**	-	-	11290	553	-
L_iC	**311**	41	**630**	74	**<0.01**
G_vC	193	24	137	20	0.07
R_tC	0.4	0.2	26.9	26.4	0.37
B_uC	-	-	5.8	3.9	-
SON	**661**	42	**822**	35	**<0.04**
L_iN	5.9	0.7	15.9	1.7	**<0.01**

* Including three older birch plots in Heidmork and three naturally regenerated birch plots in Nesjavellir



(a) *(b)*

Figure 22. Average changes in ecosystem C (a) and N-stocks (b) between treeless and three afforested sites in SW- Iceland in autumn 2017 following afforestation (sequestration = positive; loss = negative). Soil organic carbon is denoted as SOC, tree carbon and coarse roots as Ct, litter carbon as Li, vegetation carbon as Gv, and bush layer carbon as Bu. All the variables were measured in (g C m^{-2}).

3.2.6. Age related changes during forest growth (15-92 years) across all species

None of the measured tree characteristics significantly increased with age of the forest when analysed across all the forest types, which was an unexpected finding (Table 9). I ran both linear and exponential regression analyses with different exponential growth formulas, but were no significant relationships between age and forest growth variables (data not shown). When Fig. 23 was studied then it became clear that the trend lines for the age-relationship for e.g. basal area (density of the forest) were very different for conifer compared to birch or mixed forests. The conclusion is that when analysing all forest types and sites together, there was too much variability in the forest characteristics to see any clear relationship with time (age). Forest types should therefore always be separated in further analysis.

Table 9. The outcome of linear regression analysis between age (x: years) and tree characteristics that were studied in three forests in SW Iceland in autumn 2017; P: ANOVA significance of regression; R^2 coefficient of determination, n = 52 plots, y^0: intercept of linear function and *a*: slope or exponent of linear or exponential function, respectively.

Variables	R^2	P	y^0	*a*
Stem volume (litter)	0.03	0.37	23.36	0.5
Total tree biomass (kg cm^{-2})	0.02	0.45	2678	35.8
Basal area (cm^{-2})	0.03	0.41	7.63	0.10
Diameter (DBH; cm)	0.03	0.36	5.39	0.05
Dominant height (m)	0.06	0.23	4.35	0.04
Tree carbon (g C m^{-2})	0.02	0.45	1339	17.9

* Including three older birch plots and mixed forest plots in Heidmork and three naturally regenerated birch plots in Nesjavellir

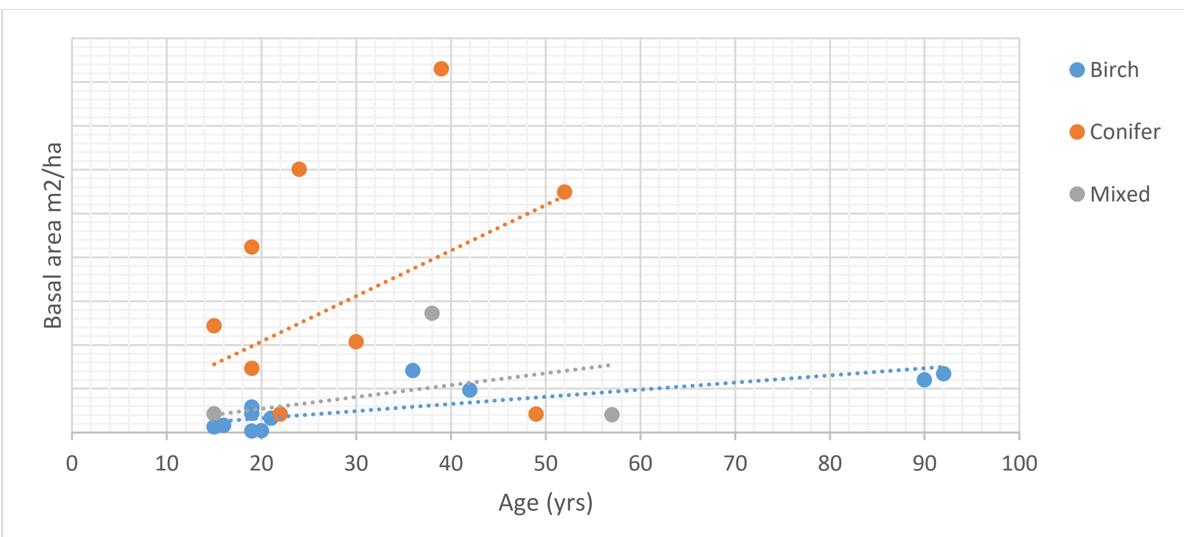


Figure 23. The outcome of none-linear regression relationship between age and basal area across all sites and all species.

Some other ecosystem characteristics did, however, show more regular changes with age when the whole forest dataset was analysed together. Total organic carbon, tree carbon, soil organic carbon, and carbon and nitrogen in litter significantly increased with age, but soil organic nitrogen (P = 0.06) and litter pH (P = 0.07) only showed a strong positive trend as soil pH (P = 0.08) showed a negative trend with age (Table 10). Carbon stock in ground vegetation significantly decreased (P = 0.04), while soil dry mass showed no significant change with age.

Table 10. The outcome of linear regression analysis between age (x: years), soil properties and ecosystem C-stocks that were studied in three forests in SW Iceland in Autumn 2017; P: ANOVA significance of regression; R^2 coefficient of determination, (n = 51 plots), y^0: intercept of linear function and *a:* slope of liner function. Significant values are highlighted as bold (P < 0.05). Total Organic Carbon includes SOC in the top 30 cm and litter C stock.

Variables	$\mathbf{R^2}$	**P**	$\mathbf{y^0}$	***a***
Soil dry mass (g DM m^{-2})	0.0003	0.89	146110	28
Soil organic nitrogen (g C m^{-2})	0.07	0.06	706.8	2.2
pH of litter	0.07	0.07	4.5	0.004
pH of soil	0.06	0.08	5.3	-0.003
Litter carbon (g C m^{-2})	0.23	**<0.01**	374	7.6
Soil organic carbon (g c m^{-2})	0.08	**0.03**	11147	41
Total organic carbon (g C m^{-2})	0.12	**0.01**	11877	54
Vegetation carbon (g C m^{-2})	0.09	**0.04**	195	-1.80
Roots carbon (g C m^{-2})	0.08	0.54	0.32	0.006
Litter nitrogen (g C m^{-2})	0.45	**<0.01**	6.46	0.300
Tree carbon (g C m^{-2})	0.13	**0.01**	401	30.7

* Including three older birch plots and mixed forest plots in Heidmork and three naturally regenerated birch plots in Nesjavellir

3.2.7. Other predictors for ecosystem changes during forest growth across all species

Because forest characteristics did not change regularly with age across all forest plots, it was interesting to look at other drivers that are more related to biomass production in the forests. Both tree basal area and stem volume showed a significant change with C-stock in trees, ground vegetation and litter (Tables 11 and 12), but, C-stock in fine roots and N-stock in soil did not change significantly with tree basal area. C-stock in soil showed positive trend with basal area (P = 0.09).

Table 11. The outcome of linear regression analysis between tree basal area and ecosystem C-stock and soil nitrogen that were studied in three forests in SW Iceland in Autumn 2017; P: ANOVA significance of regression; R^2 coefficient of determination, (n = 27 plots), y^0: intercept of linear function and *a*: slope of liner or function. Significant values are highlighted as bold (P< 0.05).

Variables	$\mathbf{R^2}$	**P**	$\mathbf{y^0}$	**a**
Tree carbon (g C m^{-2})	1,00	**<0.01**	-101	196
Vegetation carbon (g C m^{-2})	0.35	**0.01**	189.44	-4.90
Litter carbon (g C m^{-2})	0.48	**<0.01**	425.89	21.62
Roots carbon (g C m^{-2})	0.009	0.63	0.42	0.003
C in soil (g C m^{-2})	0.12	0.09	12141	88.36
N in soil (g C m^{-2})	0.04	0.30	789.99	3.11

*Excluding control plots at all the three sites

Table 12. The outcome of linear regression analysis between tree stem volume and ecosystem C-stock that were studied in three forests in SW Iceland in Autumn 2017; P: ANOVA significance of regression; R^2 coefficient of determination, (n = 27 plots), y^0: intercept of linear function and *a*: slope of liner function. Significant values are highlighted as bold (P < 0.05).

Variables	$\mathbf{R^2}$	**P**	$\mathbf{y^0}$	**a**
Tree carbon (g C m^{-2})	**0.97**	**<0.01**	**114**	**42.67**
Vegetation carbon (g C m^{-2})	0.32	**<0.02**	175.2	-0.96
Litter carbon (g C m^{-2})	0.56	**<0.01**	464.8	4.85
Roots carbon (g C m^{-2})	0.008	0.65	0.42	0.0005
Soil carbon (g C m^{-2})	0.07	0.20	12529	14.13
Soil nitrogen (g C m^{-2})	0.03	0.41	803.0	0.51

*Excluding control plots at all the three sites

3.3. Changes following afforestation by different forest types

3.3.1. Forest characteristics

Deciduous stands that were studied consisted mainly of downy birch, with some few rowan (*Sorbus aucuparia*) and willows which had a 53% higher stand density compared to planted conifer and 63% compared to mixed stands (Table 13). Conifer exhibited, however, significantly higher basal area, total stem volume and total tree biomass than deciduous stands, while mixed forests accumulated significantly higher tree biomass and tree C-stock compared to the other two forest types. The mean diameter at breast height ($DBH_{1.3m}$) for conifer was significantly higher than that of mixed stand by1.6 cm (Table 13).

3.3.2. Ground vegetation and soil properties

There was no significant variation in the surface cover of ground vegetation classes and other surface characteristics, however, deciduous forest had higher total ground vegetation surface cover (72%) compared to conifer (40%) and mixed (57%) (Table 14). Herbs cover showed an increasing trend in deciduous stand ($P = 0.06$).

Ground vegetation biomass showed a significantly higher accumulation in deciduous stands, while bush biomass showed an increasing trend in mixed stands ($P = 0.09$) (Table 14). Fine root biomass did not differ significantly across all forest types.

Soil chemical properties (pH and C/N ratio) and soil physical properties (soil dry mass, BD, stoniness, soil and litter depths) did not vary significantly across all stands, but coniferous stands contained a significantly higher amount of litter biomass (Table 15). There was an increasing trend in the soil carbon concentration ($P = 0.09$) and nitrogen ($P = 0.07$) in mixed stands compared to deciduous and conifer.

Table 13. Tree characteristics across all forest types at three afforestation sites in SW-Iceland in autumn 2017. Values are means and standard errors of deciduous (n = 12 plots), conifer (n = 12 plots) and mixed (n = 3 plots). Significant values are highlighted as bold ($P < 0.05$). Stand density is denoted by (S_d trees ha^{-1}), diameter at breast height ($DBH_{1.3}$ cm), dominant height (H_d m), basal area (BA m^{-2} ha^{-2}), stem volume (S_v m^{-3} ha^{-2}), total tree biomass (B_t g DM m^{-2}), and tree carbon (C_t g C m^{-2}).

	Deciduous *		**Conifer**		**Mixed**		
Variables	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**	**P-values**
No. plots	12		12		03		
S_d	7792	2920	2075	291	933	290	0.10
$DBH_{1.3}$ **	2.5a	0.3	10.2b	1.5	11.3ac	1.7	**<0.01**
H_d	3.2a	0.	7.6ab	1.1	7.3ac	1.6	**<0.02**
BA	2.9a	0.7	19.8ab	3.9	5.9c	3.9	**<0.01**
S_v	3.9a	1.2	80.2a	20.6	24.3b	17.7	**<0.03**
B_t	597.6ab	167.1	7524.9ba	1633.2	2189ca	14765	**<0.01**
C_t	298.7a	83.6	3762.3ba	816.6	1094.3ca	732.4	**<0.01**

* Including three older birch plots in Heidmork and three young naturally regenerated birch plots in Nesjavellir. ** measured at knee height for birch ($D_{0.5m}$)

Table 14. Effects of forest types on understory vegetation and other surface characteristics across three afforestation sites in SW-Iceland in autumn 2017. Values are means and standard errors of deciduous (n = 23 plots), conifer (n = 23 plots) and mixed (n = 6 plots). Significant values are highlighted as bold (P < 0.05). Vegetation composition was symbolised as; dwarf bush (D_{bs}), herbs (H_r), ferns, (F_r), grass (G_r), moss (M_o). Other surface classes were unvegated surfaces (U_v) and rock (R_k). Vegetation biomass was represented as (V_{dm} g DM m^{-2}) and roots in dry mas (R_{dm} g DM m^{-2}).

Variables	Deciduous Mean	SE	Conifers Mean	SE	Mixed Mean	SE	P-values
Ground layer							
D_{bs} (%)	30.00	7.57	18.46	7.85	3.33	3.33	0.26
H_r (%)	21.15	5.56	5.21	1.50	13.33	3.33	0.06
F_r (%)	5.76	2.64	7.69	2.23	1167	7.26	0.59
G_r (%)	31.92	6.94	22.11	7.21	0.00	0.00	0.59
M_o (%)	45.46	5.70	36.35	6.71	35.00	16.07	0.56
Ground vegetation and surface classes							
D_w (%)	11.92	3.37	24.61	8.53	16.67	8.82	0.38
R_k (%)	1.54	1.54	0.00	0.00	0.00	0.00	0.56
U_v (%)	14.23	4.41	35.77	9.53	26.67	8.33	0.13
Vegetated (%)	72		40		57		
Vegetation biomass							
Bu_{dm}	0.00	0.00	13.92	13.92	70.00	70.00	0.09
V_{dm}	546 a	66	161 b	43	280 b	63	**<0.01**
R_{dm}	1.22	0.37	1.09	0.25	1.34	0.41	0.91

* Including control, three older birch plots in Heidmork and three naturally regenerated birch plots in Nesjavellir

Table 15. Influence of forest types on soil physical and chemical properties at three afforestation sites that were studied in SW-Iceland in autumn 2017. Values are means and standard errors of deciduous (n = 23 plots), conifer (n = 23 plots) and mixed (n = 6 plots). Significant values are highlighted as bold (P< 0.05). Physical properties were symbolised as; litter depth (L_{dt} cm), soil depth (S_{dt} cm), litter in dry mass (L_{idm} g DM m^{-2}), soil dry mass (S_{dm} g DM m^{-2}), percentage of stones in the in dry weight of soil (S_t%), bulk density (BD g cm^{-3}). Soil chemical properties were symbolised as; (soil pH ($pH_{s\ 0\text{-}5cm}$,) litter pH (pH_L), carbon to nitrogen ration in soil ($C/N_{s\ 0\text{-}30cm}$), carbon to nitrogen ratio in litter (C/N_L), carbon concentration in soil (C_s%), percentage weighted Nitrogen in soil (N_s%), percentage weighted carbon in litter (C_L%) and parentage weighted Nitrogen in litter (N_L%).

	Deciduous		**Conifers**		**Mixed**		
Variables	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**	**P-values**
Soil physical properties							
Li_{dt}	5.96	0.57	6.58	0.71	4.17	0.17	0.26
S_{dt}	56.08	6.61	58.61	4.38	56.00	12.12	0.94
Li_{dm}	1258	191	2127	303	1409.7	370	0.06
S_{dm}	142692	7935	166216	14788	139535	8981	0.27
S_t	11.76	5.92	1.79	0.86	0.26	0.26	0.19
BD	0.42	0.02	0.44	0.04	0.39	0.02	0.67
Soil chemical properties							
pH_s	5.22	0.07	5.16	0.06	5.04	0.04	0.25
pH_L	4.68	0.10	4.58	0.08	4.700	0.12	0.67
C/N_s	16.23	1.0	15.9	0.3	15.7	0.5	0.90
C/N_L	40.86	5.04	40.93	3.90	33.10	1.81	0.72
C_s	8.47	0.31	9.36	0.69	11.15	0.92	0.09
N_s	0.53	0.03	0.56	0.04	0.71	0.05	0.07
C_L	38.05	1.55	39.79	1.71	36.75	2.68	0.62
N_L	1.37	0.35	0.98	0.09	1.11	0.08	0.55

* Including control, three older birch plots in Heidmork and three naturally regenerated birch plots in Nesjavellir

3.3.3. Ecosystem C-stocks and soil N

Total ecosystem organic carbon varied significantly in soil, litter and ground vegetation across all forests types (Table 16). There was significantly higher C stock in litter in conifer, but deciduous trees contained significantly higher amount of C in vegetation. There was no significant variation in C stock in fine roots across all forest type. However, carbon stock in bush layer showed an increasing trend (P=0.09) in mixed forest stands compared to the other forest types. There was not significant variation in soil organic nitrogen stock and litter nitrogen across all forest types (Table 16).

Table 16. C-stocks and soil N in the top 30 cm in different forests types in SW Iceland. Values are means and standard errors of deciduous (n = 23 plots), conifer (n=23 plots) and mixed (n=6 plots). Significant values are highlighted as bold (P < 0.05). Ecosystem C-stock were denoted as; soil organic carbon (SOC g m^{-2}), carbon in litter (LC g m^{-2}), carbon in ground vegetation (VC g m^{-2}), carbon in roots (RC g cm^{-2}), carbon in bush layer (BC g cm^{-2}) and total organic carbon (TOC g m^{-2}). Soil N-stock was denoted as; soil organic nitrogen (SON g m^{-2}) and nitrogen in litter (LN g m^{-2}).

	Deciduous		**Conifers**		**Mixed**		
Variables	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**	**P-values**
SOC	11,679	769	13781	1174	15574	1619	0.13
LC	491 ^{a}	61	924 ^{b}	132	530 ^{ab}	128	**0.01**
VC	218	26	64	17	106	28	**<0.02**
RC	0.49	0.14	0.43	0.10	0.54	0.17	0.91
BC	0.00	0.00	5.55	5.55	27.94	27.94	0.09
TOC	12741	598	17793	1583	19529	3018	**0.01**
SON	746	69	850	54	987	0.14	0.13
LN	13.23	2.55	19.08	2.59	13.43	3.03	0.24

* Including control, three older birch plots in Heidmork and three naturally regenerated birch plots in Nesjavellir

3.4. Age-related changes following afforestation

3.4.1. Forest characteristics

Three birch plots in Heiðmörk had older trees than from 1950, when the area was fenced and the afforestation activities started. Those plots were excluded when effects of afforestation were studied, i.e. trees existed before the afforestation began and the time that the area had been under forest could therefore possibly be much longer than the tree age indicated, if those trees had regenerated from pre-existing stands. When basal area, stem volume, diameter at breast height and total tree biomass were correlated with age separately, the result showed an exponential and linear increment except in DBH for birch forest (Table 17). The exponential model was better for the deciduous forests, but both models gave similar results for the basal area and stem volume in coniferous forests, as judged from the R^2 values, but the exponential model was also slightly better for tree biomass and DBH in conifer (Table 17).

Table 17. Age related changes in coniferous (n = 23 plots) and deciduous forest (n=20 plots) in three afforestation sites in SW-Iceland in autumn 2017. Treeless sites, old growth and mixed forests were excluded. The outcome of a linear and exponential analysis between age (x: years) and tree variable; ANOVA significance of regression (P < 0.001); R^2: coefficient of determination; a: intercept of linear or exponential function; y^0: slope of linear function. Tree biomass (tree woody biomass aboveground and in coarse roots) and diameter at breast height (DBH).

	Conifer				**Deciduous***			
Variables	**R^2**	**P**	**y^0 or *b***	***a***	**R2**	**P**	**y^0**	***a***
Exponential model (f = a*exp(b*x))								
Basal area **	0.54	**0.01**	6.92	0.031	0.78	**<0.02**	0.247	0.093
Stem volume	0.68	**<0.01**	16.53	0.046	0.93	**<0.01**	0.134	0.129
T-biomass	0.61	**<0.03**	2217.13	0.036	0.94	**<0.01**	32.154	0.110
DBH	0.51	**0.01**	4.68	0.023	0.02	0.74	2.115	0.000
Linear model (f = y0+a*x)								
Basal area	0.54	**0.01**	-1.75	0.702	0.76	**<0.02**	-4.249	0.307
Stem volume	0.66	**<0.01**	-43.76	0.031	0.88	**<0.02**	-10.764	0.669
T-biomass	0.56	**<0.03**	-1838.61	304.50	0.92	**<0.01**	-1193.8	78.488
DBH	0.49	**0.01**	2.26	0.258	0.003	0.89	2.033	0.008

* Excluding three older birch plots and the three mixed plots in Heidmork; **also shown in Fig. 21

3.4.2. Ground vegetation and soil characteristics across all species and sites

In coniferous stands, ferns, moss and grass cover did not significantly change with age whereas dwarf bushes and herbs cover significantly decreased (Table 18). While other surface characteristics (dead wood and unvegetated) significantly increased with forest age, the surfaces covered by rock did not significantly vary. Soil pH, bulk density, ground vegetation dry mass significantly decreased with age, litter dry mass increased, while soil dry mass, soil and litter depths did not significantly change with age in coniferous stands (Table18).

Deciduous stands showed no significant change in ground vegetation composition, surface characteristics or soil properties with age, but mixed stand showed a significant increase in litter and root dry mass as well as surfaces covered by deadwood (Table 18). The present analysis does not compare the regression models of the forest types.

Table 18. The outcome of linear regression analysis between age (x: years), ground vegetation, other surface classes and soil characteristics that were studied in conifers (n = 23 plots) and deciduous (n=20 plots) forests in SW Iceland in Autumn 2017; P: ANOVA significance of regression; R^2 coefficient of determination, y^0: intercept of linear function and a: slope of liner or function. Ground vegetation group are denoted as dwarf bush (D_{bs}%), herbs (H_r %), ferns, (F_r %), grass (G_r %, moss (M_o%). Other surface classes are denoted as rock (R_k %) and unvegated (U_v%). Dry mass of biomass is symbolised as litter in dry mass (L_{dm} g DM m), vegetation in dry mass (V_{dm} g DM m^{-2}), roots in dry mas (R_{dm} g DM m^{-2}), soil physical properties are represented as soil dry mass (S_{dm} g DM m^{-2}), soil bulk density (BD_{0-30} g cm^{-2}).

Variables	**Conifer**				**Deciduous**				**Mixed**			
Vegetation	R^2	P	y^0	a	R^2	P	y0	a	R^2	P	y^0	a
Db	0.40	**<0.01**	42.99	-0.91	0.02	0.57	30.68	0.35	0.62	0.06	47.08	-0.022
Hr	0.27	**0.01**	9.30	-0.14	0.03	0.54	9.32	0.11	0.09	0.56	7.64	0.083
Fr	0.03	0.47	4.150	0.06	0.03	0.52	4.63	0.17	0.01	0.86	5.12	0.04
Gr	0.07	0.24	26.00	-0.27	NR	-	-	-	0.56	0.09	13.05	0.379
Mo	0.10	0.13	46.22	-0.34	0.07	0.31	34.90	0.61	0.04	0.72	33.89	-0.212
Other surface classes												
Dw	0.48	**<0.02**	-1.18	0.89	0.01	0.66	11.03	-0.17	0.94	**<0.01**	-1.45	0.535
Rk	0.06	0.25	3.70	-0.09	0.05	0.41	0.96	0.13	0.17	0.42	13.36	-0.41
Uv	0.33	**0.04**	15.17	0.82	0.02	0.62	23.58	-0.23	0.12	0.82	26.07	0.123
Soil chemical properties												
pHL	0.03	0.39	4.54	0.002	0.04	0.47	4.45	0.01	0.31	0.25	4.27	0.032
pHs	0.45	**<0.01**	5.43	-0.01	0.101	0.68	5.14	-0.002	0.01	0.89	5.31	0.001
Soil physical properties												
Gdm	0.72	**<0.01**	45.88	-0.87	0.02	0.64	55.82	0.35	0.26	0.30	54.58	-0.68
Rdm	0.03	0.41	0.81	0.01	0.02	0.65	1.37	-0.02	0.75	**0.03**	0.34	0.03
Ldm	0.63	**<0.01**	816.16	45.11	0.05	0.41	1016	12.0	0.82	**0.01**	429.04	25.95
Sdm	0.02	0.51	164375	-288	0.08	0.29	121195	658	0.001	0.95	140751	62.00
BD	0.19	**0.03**	0.52	0.003	0.16	0.13	0.37	0.003	0.02	0.77	0.46	-0.001
Sdt	0.07	0.21	67.60	-0.26	0.01	0.76	55.02	0.17	0.08	0.58	42.85	0.21
Ldt	0.05	0.29	4.88	0.03	0.12	0.18	4.42	0.10	0.003	0.95	3.95	0.003

3.4.3. C and N-stocks in conifer, deciduous and mixed stands

Conifer showed a significant increase in all variables studied except in bush layer and fine roots, while in deciduous stands, only tree C stock changed with age (Table 19). In mixed forests, there was a significant increase in C-stock in litter and fine roots as well as litter N.

Table 19. The outcome of linear regression analysis between age (x: years) and C and N-stocks that were studied in conifers (n = 23 plots), deciduous (n = 23 plots) and mixed (n = 6 plots) stands in three forests in SW Iceland in Autumn 2017; P: ANOVA significance of regression; R^2 coefficient of determination, y^0: intercept of linear function and *a*: slope of liner or function. Ecosystem C-stock was denoted as; tree carbon above ground and coarse roots (TC), vegetation carbon (VC), bush layer carbon (BC), soil organic carbon (SOC), litter carbon (LC), root carbon (RC) and N-stock in soil was symbolised as; litter nitrogen (LN) and soil organic nitrogen (SON).

	Conifer				**Deciduous**				**Mixed**			
	R2	P	Y0	a	R2	p	Y0	a	R2	P	Y0	a
TC	0.41	**<0.01**	317.88	89.66	0.62	**<0.03**	-51.80	14.93	0.09	0.29	151.2	21.60
VC	0.72	**<0.01**	183.50	-3.48	0.01	0.67	224.00	1.27	0.29	0.27	219.71	-2.95
BC**	0.004	0.78	2.28	0.05	-	-	-	-	0.004	0.90	15.74	-0.09
SOC	0.17	**0.05**	10991	84.15	0.11	0.21	103340	64.23	0.31	0.25	11692	147
LC	0.79	**<0.01**	247.65	21.88	0.06	0.35	392	4.92	0.73	**0.03**	183.79	8.93
RC	0.02	0.55	0.36	0.002	0.02	0.65	0.55	-0.01	0.75	**0.03**	0.14	0.011
LN	0.25	**0.02**	8.53	0.25	0.13	0.16	7.49	0.23	0.71	**0.02**	3.32	0.25
SON	0.91	**<0.01**	129.3	0.05	0.08	0.30	663	4.82	0.48	0.13	723.56	5.65

** Bush layers were not encountered in deciduous forest

3.4.4. Total ecosystem C and N stocks

Since the afforestation started, there has been a significant increase in the amount of carbon and nitrogen sequestered per year both above and below-ground (Table 20). Soil, litter and fine roots (below-ground) combined were sequestering C at the rate of 49.0 C g m^{-2} yr^{-1} reaching a total soil C-stock of 11,521 g C m^{-2} and trees (including coarse roots), ground vegetation as well as bush (above-ground biomass) combined, were sequestering C at the rate of 29.3 C g m^{-2} yr^{-1} reaching a total C-stock of 595 g C m^{-2} by 2017 (Table 20).

N in the top 30 cm of mineral soil and litter combined also continued to increase significantly with age at the rate of 2.41 g N m^{-2} yr^{-1}. However, N-stock in soil alone showed only a very strong trend (P = 0.06), while C-stock in bush layer did not significantly increase with age (P = 0.98; Table 20).

Table 20. The outcome of regression analysis between age (x: years) and ecosystem C-stock that were studied in treeless (n=23 plots) and the three afforested (n=29 plots) sites in SW Iceland in Autumn 2017; P: ANOVA significance of regression; R^2 coefficient of determination, y^0: intercept of linear function and *a*: slope of liner or function.

Variables	R^2	P	y^0	*a*
Tree carbon	0.13	**0.01**	400.76	30.75
Ground vegetation C	0.10	**0.02**	191.12	-1.49
Bush C	0.89	0.98	3.04	-0.002
Total above-ground C	0.12	**0.01**	594.9	29.25
Soil carbon (0-30cm)	0.09	**0.03**	11146.9	*41.45*
Litter carbon	0.23	**<0.03**	374.1	*7.58*
Fine roots carbon	0.08	**0.05**	0.33	*0.01*
Total below-ground C	0.11	**0.02**	11521.0	49.03
Soil nitrogen	0.07	0.06	706.89	2.24
Litter N	0.24	**<0.03**	8.00	0.18
Total below-ground N	0.08	**0.05**	714.9	2.41

* Including three older birch plots in Heidmork and three naturally regenerated birch plots in Nesjavellir

3.5. **Tree basal area-related changes**

3.5.1. Total ecosystem C and N stocks

As regression analysis did not explain the changes with age, except the data was broken into different forest types, it was of interest if other drivers than age, such as basal area would be a better predictor across all sites. Above-ground C stock (trees, vegetation and bush combined) as well as below-ground C stock (soil, litter and roots combined) significantly increased with basal area growth (Table 21). Tree biomass alone was significantly accumulating approximately 195 g C m^{-2} BA^{-1} growth, as expected. However, ground vegetation significantly decreased by 4.91 g C m^{-2} BA^{-1}, soil within 30 cm depth was significantly sequestering approximately 124 g C m^{-2} BA^{-1}, litter significantly increased by 23 g C m^{-2} $BA^{-1}$by Autumn 2017. C-stock in fine roots and bush layer did not show a significant change with basal area (P = 0.61) and (P = 0.41, respectively (Table 21).

Total nitrogen (top 30 cm in mineral soil and in litter) also significantly increased with basal area growth (Table 21).

Table 21. The outcome of regression analysis between tree basal area and C and N-stocks that were studied across all afforested sites (n=26 plots) in SW Iceland in Autumn 2017; P: ANOVA significance of regression; R^2 coefficient of determination, y^0: intercept of linear function and a: slope of liner or function.

Variables	R^2	P	y^0	a
Tree C	0.99	**<0.01**	-98.18	195.93
Bush C	0.01	0.41	4.73	-4.041
Vegetation C	0.20	**<0.01**	189.55	-4.91
Total above-ground C	0.99	**<0.01**	92.88	191.29
Soil organic C (0-30cm)	0.15	**<0.05**	11258.3	124.31
Litter C	0.39	**<0.01**	393.25	22.95
Root C	0.01	0.61	0.41	0.003
Total below-ground C	0.19	**<0.01**	11652	147.27
Soil N	0.08	**0.03**	716.31	6.11
Litter N	0.27	**<0.01**	8.99	0.437
Total N	0.09	**0.02**	725.31	6.55

Excluding control plots and the three mixed plots in Heiðmörk

3.6. C-sequestration rates in individual chronosequences

When the data were analysed separately for each forest type at each site (chronosequences), the regression analysis showed that in Heiðmörk the coniferous stands were significantly sequestering C in both top 30 cm of mineral soil and litter (76 g C m^{-2} $year^{-1}$) and in tree biomass (additional 101 g C m^{-2} $year^{-1}$), while neither the old growth birch or the mixed forest type had a significant change in their C-stocks with age (Table 22).

The young planted birch stands in Ölfusvatn significantly sequestered higher amount of C in soil (134 g C m^{-2} $year^{-1}$) than aboveground (only additional 4 g C m^{-2} $year^{-1}$), but the young conifer mostly sequestered C in tree biomass aboveground and in coarse roots (155 out of 160 g C m^{-2} $year^{-1}$; Table 22).

The two chronosequences in Nesjavellir were interesting since it was possible to compare C-sequestration by naturally regenerated or planted birch of similar age. For the top 30 cm in the mineral soil and in the litter combined the planted birch had a strong trend for C-sequestration (P=0.06; 112 g C m^{-2} $year^{-1}$, while the change under naturally regenerated birch (6 C m^{-2} $year^{-1}$) was far from significant. When the tree biomass (above-ground and in coarse roots) was included the planted birch had a significant C-sequestration rate of 132 g C m^{-2} $year^{-1}$, but the naturally regenerated had only a strong trend (P=0.06; 33 g C m^{-2} $year^{-1}$; Table 20). This difference between planted and naturally regenerated birch is interesting.

Table 22. The outcome of regression analysis between age (x: years) and ecosystem C-stock that were studied at individual chronosequence in Heiðmörk (conifers n = 18, mixed n = 6 and old birch n = 6) in Ölfusvatn (conifer n = 6, planted birch n = 5) and Nesjavellir (planted birch n = 6 and natural birch n = 5 plots); R^2 coefficient of determination, y^0: intercept of linear function and a: slope of liner or function. Ecosystem C-stock was denoted as; soil organic carbon within 0-20 cm depth plus litter carbon (SOC+Li) and soil organic carbon within 0-20 cm depth plus litter carbon and tree carbon was symbolised as (SOC+Li+Ct).

		Heiðmörk		
Variables*	R^2	P	y^0	*a*
Conifer				
SOC	0.19	0.08	10780	94
SOC+Li	0.32	**<0.05**	7699	76
SOC+Li+Ct	0.56	**<0.01**	7793	114
Mixed				
SOC	0.41	0.17	11198	94
SOC + Li	0.46	**0.14**	8294	96
SOC+Li+Ct	0.44	0.15	8629	127
Old birch				
SOC	0.05	0.68	10795	13.8
SOC +Li	0.11	0.47	8011	17
SOC +Li+Ct	0.17	0.36	8406	23
		Ölfusvatn		
Conifer				
SOC	0.16	**0.04**	12419	81
SOC +Li	0.003	0.91	8909	4.9
SOC +Li +Ct	0.66	0.05	9217	159
Planted birch				
SOC	0.51	0.11	12036	87
SOC +Li	0.78	**0.05**	8179	134
SOC+Li+Ct	0.78	**0.05**	8496	138
		Nesjavellir		
Planted birch				
SOC	0.46	0.14	9222	126
SOC+Li	0.62	0.06	7208	112
SOC+Li+Ct	0.69	**0.04**	7746	132
Natural birch				
SOC	0.09	0.63	9266	46
SOC+Li	0.002	0.94	7817	5.6
SOC+Li+Ct	0.06	0.70	7981	33

*Including control plots for each chronosequence

4. Discussion

This study revealed that afforestation induced changes in ground vegetation and soil properties. It also revealed that it significantly increased the accumulation of C in the ecosystem, especially in standing tree biomass, litter C and SOC. I relate my discussion to what other comparable studies have found, to draw more general conclusions on the effects of growing trees in treeless landscapes on ground vegetation, ecosystem C-stocks and chemical and physical properties of the top layer of soil. Treeless control sites are first discussed in order to better set the baseline before changes following afforestation are discussed.

4.1. Comparison of treeless sites

The first step was to find out if there were variations in the treeless sites in terms of vegetation cover, soil properties and ecosystem C-stocks without the effect of afforestation. My investigation showed that, contrary to my expectation, there were no significant differences across all the three treeless sites, except in grass cover, fine roots and soil dry mass (Table 3). Even within the high-temperature geothermal area of Nesjavellir (Gunnarsson et al., 2015), the result showed no significant variations in C-stock, vegetation cover and soil properties compared to Ölfusvatn and Heiðmörk (Tables 3 & 4). This could be because the effects of geothermal activities on soil and vegetation are usually only found on a small spatial scale around geothermal vents or hot-spots where bedrock is warmed up by geothermal channels, but not across large areas (Sigurdsson et al., 2016). So even if geothermal vents exist at Nesjavellir (Gunnarsson et al., 2015), they are only rarely located within the afforested area and therefore did not affect the ground flora or soil conditions in my study.

The fact that there were almost no significant differences between these three treeless study areas, made it, however, possible to merge all the data from the three sites for some of the analyses. As I looked at the general trends of how the ground flora and soils responded to afforestation, also with different forest types and different time (age). I could then later compare such overall trends with the site-dependent data, using minimum number of measurement plots for comparisons (see later).

4.2. Changes in soil physical and chemical properties following afforestation

4.2.1. Bulk density

Andosol is the dominant soil type of Iceland and it is normally characterized by low bulk density (BD), high porosity and soil water retention (Oskarsson et al., 2004; Arnalds, 2008). The relatively low bulk density (Table 7) found within the present study sites fits well with some other studies in Iceland. Arnalds (2015) reported somewhat higher, or an average BD lower than 0.8 g cm^{-3} for Icelandic Andosol soils. However, similar BD values, ranging from 0.3 to 0.8, were observed in a birch forest and grassland in south Iceland (Hunkziker et al., 2019). The overall low BD of Andosols has been linked to their high C concentration (Arnalds, 2015). Following afforestation, there was a slight tendency for lower BD and increased C concentration of the top 30 cm in this study (Table 7), but those changes became clearer when only the topsoil layers were studied (Fig. 21). This implies that changing land use from treeless land to forests affected BD, soil structure, compaction and porosity. Ellert and Bettany (1995) and Murty et al., (2002) reported a reduction in BD and increased porosity in forests, while European Commission (2006) confirmed that degrading soils is one of the major causes of soil compaction which leads to increased BD.

Human induced soil compaction and changes in BD can, however, also occur in forests. It has been shown that the two most important human activities responsible for soil compaction in Europe are agriculture and forestry due to ground pressure from heavy machinery (both) or animal management in agricultural land (Virto et al., 2014; Solgi et al., 2018). At my afforested study sites, there was no evidence of soil compaction, but instead there was some soil expansion taking place, especially in the top 10 cm (Fig. 21). This was very likely attributed to the minimum forest soil management practices that had taken place at the sites (limited use of heavy machineries during land preparation). It is, however, important to keep it in mind that this may change in Icelandic forests when and if the use of heavy machinery increases.

4.2.2. Soil mass correction

The soil mass correction factor (SCF) method, which accounts for the differences in soil mass among treatments, is being increasingly employed when SOC stocks are measured (Weismeier et al., 2015). The reason behind using the SCF method in this study was to estimate changes in SOC by eliminating the differences in soil mass caused by differing land use practices. Similar methods have previously been used for Icelandic data by Ritter (2007) and Barcena et al. (2014). The correction factors were always < 1.4 for individual forest soil layers (Fig. 21). Although relatively low SCF values were found (Fig. 21), correcting soil mass turned out to be very important for accurately calculating SOC stocks in the 0-30 cm depth. Using the SCFs increased SOC-stocks in the top 30 cm of soil by 1,875 g C m^{-2} or by of 0.8% in afforested compared to treeless sites (Table 8). The SOC stock gained with this method is in agreement with Mikha et al., (2013) who detected a loss of 890 g C m^{-2} when they used equivalent soil mass (ESM) and compared to traditional method. It is crucial that soil mass correction is done to accurately assess SOC stock between different land uses in Icelandic studies. Otherwise, the variations in soil mass, which mainly occur in surface soils (0-10 cm), may obscure changes in the profile SOC stock when estimated to a fixed depth only.

4.2.3. Soil pH

Changes in soil pH following afforestation with different tree species have been documented (Johnson et al., 1990). A study in Sweden compared a 55-year old stands of Norway spruce with European beech and found a strong acidification in the top soil layer (Tamm and Hallbäcken, 1986). In this study, regression with age of individual chronosequences from the three forest types showed a significant reduction of pH under conifer over time but not for the other two forest types (Table 18). This change was, however, minor or only 0.01 pH per year, and interestingly the intercept of this relationship (initial conditions) for the conifer stands was higher (less acid) than for the other two forest types. This explains why no acidification was detected when average conditions were compared in an earlier analysis (Table15). Such mild acidification could be caused by acids produced in decomposing litter from the more productive conifer stands (Nykvist, 1959; Binkley and Richter, 1987). Other soil types than Andosols may be more reactive, e.g. soil pH declined from 4.0 to between 3.7 to 3.5 when pine/mixed hardwood forests were studied in northern USA (Montagnini et al., 1986). This study concurs with the view that soil pH appears to be influenced by forest types, typically by a few tenths of units over some decades and in some cases, it changes even faster.

4.2.4. Soil and litter C:N ratio

Upon establishment of forests, surface litter had a significantly higher C:N ratio compared to mineral soils (Table 7). The forest types did not have different litter C:N ratio (Table 15) but especially the conifers which had the highest growth rates had also most accumulation in litter C-stock (Table 16). This was consistent with (Kirschbaum et al., 2008). The higher C:N ratio in the forest litter was attributed to the transfer process of C from dead plant materials on the surface to the mineral soils. Paul et al. (2002) and Davis et al. (2002) in their review, confirmed that while there was no consistent changes in soil C a few decades after afforestation, there was typically an increase in C-stock in the litter layer (O-layer).

4.3 Changes in vegetation

4.3.1. Cover and composition

Afforestation is known to influence ground vegetation composition especially after canopy closes (Lortie et al., 2004; Liancourt et al., 2005). In this study, the dwarf bush drastically changed with a significant mean reduction of 30% (Table 6). This confirmed the hypothesised changes. Such reductions in dwarf bush composition may not only be linked to more competition for soil water and nutrients (Fahey, 2001), but most importantly due to decreased irradiation at the soil surface (Sigurdsson et al., 2005). Light is a key resource for plant growth, so, tree canopies which are light-absorbing objects can efficiently reduce light availability beneath the forest canopies (Sigurdsson et al., 2005).

4.3.2. Forest type effects on ground vegetation biomass

The ground vegetation biomass was significantly lower in coniferous and mixed forests compared to the deciduous in the present study (Table 14). This can be partly related to the temperate deciduous forest trees dropping their leaves in autumn allowing high seasonal variation in available light for the ground vegetation; while evergreen conifer trees keep their leaves (needles) year round and thereby reducing the light availability in all seasons as observed by Mestre et al. (2017). However, the three forest types investigated in this study also differed in mean forest structural characteristics, like basal area (measure of stand density) and shade tolerance of each stand which might have influenced the light conditions below the tree canopies.

4.3.3. Age related changes in ground vegetation cover, biomass and C-stock

Regression relationship showed a significant annual reduction in ground vegetation biomass and vegetation aboveground C-stock under conifer stands, but not in deciduous and mixed ones (Table 18 and 19). Sigurdsson et al. (2005) have previously shown that ground vegetation may increase in the first decade(s) following afforestation in Siberian larch, while it generally decreased in middle-aged planted Siberian larch forests after canopy closure and old-growth downy birch forests, and at a similar annual rates as found in the present study. In the Sigurdsson et al. (2005) study the ground vegetation C-stock increased again in 50-year-old Siberian larch stands following thinning. I.e. the annual change in ground vegetation C-stock may not always be as linear with time as was found in the present study. MacLean and Wein (1997) showed a similar non-linear change in pine and mixed stands of 7 to 57 years of age, where ground vegetation biomass production decreased with age, but later stabilized at an older age.

4.4 Changes in litter

4.4.1. Litter biomass and C-stock

It has been showed that afforestation can lead to increased litter production and accumulation underneath the forest canopy (Cao et al., 2019). I also found that afforested sites had significantly higher amount of litter dry mass (necromass) i.e., 760 g DM m^{-2} more compared to treeless sites (Table 7). This is due to higher quantity of detritus/dead plant materials (leaves, bark, needles, twigs and cladodes) which were periodically added to top soil layer (O-horizon) after thinning or pruning. A literature review and long term observation revealed a strong relationship between litter amount and thinning in Norway spruce, Scots pine (*Pinus sylvestris*) and European larch (*Larix decidua*) in central Europe (Kacálek et al., 2018).

Connected to amount was litter depth which was found to be significantly deeper or thicker in afforested sites (ca. 5.9 cm) compared to treeless plots (ca. 4.2 cm; Table. 7) in the present study. A similar litter depth ranging from 5-30 cm was recorded under conifer stand in European forests (Kacálek et al., 2018). In the current study, I also found a significantly higher amount of litter C

(34%), litter N (46%; Table 8) and significantly higher necromass (35%: Table 7) in afforested-compared to treeless sites. Similar differences in litter C stock and biomass of 36% and 32% was also reported in north and central Europe, respectively (Neuman et al., 2018). On average, the increase may even approach 50% in boreal forest (Liski et al., 2006).

Another reason for the differences in litter between treeless and afforested sites could be due to the variations in nutrient concentrations in the litterfall. Plants that grow in areas with poor soil (low nutrients) tend to produce litter with low nutrient concentrations, while fertile soils produce nutrient-rich leaves which are returned to the soils by leaching or relatively faster mineralisation (Gallardo and Merino, 1992). The specific focus of this study was therefore not to evaluate how or why litter C respond to afforestation, but rather to show the importance of including litter when estimating forest C stocks and changes over time. IPCC Guidelines (2003, 2006) show that on average conifer species contain 220 g C m^{-2} of litter and deciduous have 130 C m^{-2}. In this study we recorded 924 g C m^{-2} for conifer and 491 g C m^{-2} for deciduous trees (Table 14). At global scale, litter accounts for 5% of all forest ecosystem C stocks (Domke et al., 2016). This therefore means that changes in litter C pool have important implications for national and global C budgets and C emissions reduction targets and negotiations.

4.4.2. Effect of forest type on litter C

I hypothesized that coniferous trees would accumulate more C in the litter layer compared to broadleaves and mixed forest. My observation supported this hypothesis and showed that conifer forests contained almost twice as large stocks of litter C compared to deciduous and mixed forests (Table. 16). This conformed well with findings of a Pan-European forest monitoring network which reported larger litter C-stock of 52.3% more in conifer than deciduous (Neumann et al., 2018). The variation in litter C among the forest types could be a result of an inherent difference in litter composition and other ecological processes, such as decomposition, formation of humus and nutrient cycling. Krishna and Mohan, (2017) showed that litter C-stock can differ substantially between forest types depending on litter composition and this may partly be explained by differences in cell wall components, such as lignin and cellulose that influence the litter decomposition and nutrient release. In the birch forests (broadleaf) that I studied, green plants beneath the tree canopy (dwarf shrubs, mosses, herbs and grasses) probably produced the majority of the litterfall, while deadwood was the main component of litter layer in conifer forests. This could explain higher litter decomposition rate in the birch forest and, thus explaining part of the difference in accumulated sources of litter C in each forest type. However, the process of decomposition was not studied in this research. Thus, understanding the determinants of litter production and quality and rates of litter decomposition are of paramount importance for better understanding the apparent differences in litter C accumulation in different forest types in Iceland.

4.4.3. Effect of age on litter C and N

Regression relationship showed that litter C and N-stocks increased significantly with age across all the forest types at the rate of 7.6 g C m^{-2} yr^{-1} and 0.2 g N m^{-2} yr^{-1} (Table 20). The rate was even higher when this was broken down to individual forest types with the mean annual sequestration rate of about 21.9 g C m^{-2} yr^{-1} and 0.25 g N m^{-2} yr^{-1} for conifer and 8.9 g C m^{-2} yr^{-1} and 0.25 g N m^{-2} yr^{-1} for mixed forests (Table 19). Birch forests did not have significant regression between litter C and N-stocks amount and age when studied separately; but their annual trend amounted to 4.9 g C m^{-2} $year^{-1}$. The annual litter accumulation rates estimated in the current study could probably be because of the forests management practices and stand ages which are still young and growing vigorously. An assessment of the successional development in a Canadian White pine (*Pinus strobus*) plantation stands aged 2, 15, 30 and 56 years old indicated an increase in litter C from 8.0, 75, 54 and 121 g C m^{-2} yr^{-1}, respectively (Peichl and Arian. 2006). So such age-dependent differences are to be expected.

4.5. Changes is soil organic carbon (SOC)

4.5.1. Effects of afforestation on SOC

Comparing the relative share of C among the forest C-pools in Table 8, SOC in the top 30 cm of soil was demonstrated as the largest forest ecosystem C-stock. This supports the hypothesis of Ritter (2007) that SOC is the biggest forest C pool in Iceland. Studies elsewhere also have found a high proportion of the total ecosystem C stock to be SOC in natural or planted forest ecosystems (Forest Europe and FAO, 2011; Gundersen et al., 2014; Poeplau et al., 2017). The reason for the SOC forming the major part of forest C stocks is attributed to their balance between the processes of photosynthesis and autotrophic respiration on one hand and biomass production, litter fall, decomposition and associated heterotrophic respiration on the other (e.g. Vesterdal et al., 2013; Gougoulias et al., 2014). Also the relative amount and spatial distribution of litter production from roots, compared to aboveground litterfall, in forests maintains their higher SOC stocks compared to other vegetation types (Vogt et al., 1986; Kleja et al., 2008; Finér et al., 2007; Rasse et al., 2005; Crow et al., 2009). More detailed studies are needed on those processes in Iceland to better understand which of those are the most important for the observed changes in the present study.

Sigurdsson, (2014) recently showed that following afforestation in Iceland, the belowground SOC-sequestration is relatively more important during the first couple of decades than the aboveground woody C-sequestration; albeit both processes are lower during the initial years. Sigurdsson (2014) also found that this reversed when planted forests become >30-40 years old and the biomass C-stocks then start to increase more rapidly than the SOC stocks. Bearing in mind the relatively low average age of the forest stands in the present study, my finding supports this observation. The finding of Sigurdsson (2014) and the present study seems, however, to be in some contrast to a statement made by Vesterdal et al., (2013), who claimed that at initial stages of growth trees have very little impact on SOC stock, but as forest develop, input of C from litterfall increases and stabilises at approximately 20-30 years. However, Sigurdsson (2014) also found that the SOC sequestration rates increased with age, but also that they were still relatively higher than the C-sequestration rates in aboveground biomass in the early years following afforestation.

4.5.2. Effects of forest types on SOC

The current study further explored whether SOC differed under different forests types (conifer, deciduous and mixed) and the result showed that while average litter C stock was significantly different, there was no significant variation in the average SOC stock across the three forest types (Table 16). This was consistent with Barcena et al. (2014) who looked at paired stand studies in the Nordic and UK region and found that forest types were not different in SOC stocks except when SOC and litter C were considered together. They found that planted coniferous forests stored relatively more C as litter than as SOC. However, Vesterdal et al. (2013), who reviewed and synthesised larger set of papers on the effect of tree species on litter and SOC in mineral soil in temperate and boreal region of Europe and North America, concluded that forest type could have major effect on SOC storage. They also found that the C-storage is larger in the litter layer of coniferous forests than in deciduous forests. This study agrees therefore more with the conclusion of Barcena et al. (2014) that forest types only influenced SOC stock differently when the litter layer was also included. This shows the importance of carefully assessing the litter layer stocks when C-sequestration is evaluated in different forest types. This observation further implies that if I had excluded changes in litter stock with forest type, I would possibly have misinterpreted the difference in surface-soil C sequestration under different forest types.

4.5.3. Annual sequestration rate of SOC

The present study observed significant changes in SOC when compared across all the afforestation sites (Table 8) which had a mean age of only 35 years. Regression analysis revealed annual changes in SOC sequestration rate of 41.5 g C m^{-2} yr^{-1}, as an average for all the afforested sites but also for 30 cm thick soil layer instead of 10 cm (Table 20). This is similar to Icelandic overall SOC sequestration rate of 52 g C m^{-2} yr^{-1} given in the national inventory (Hellsing et al., 2016). The observations made in this study and the national inventory indicated that changes in SOC stocks with age depend on land use prior to afforestation which, in Iceland, was mainly farming. Studies elsewhere showed that changing land use from agriculture to forest would require more than 30 years before significant change in the amount of SOC can be detected (Six et al., 2002; Laganiere et al., 2010; Nave et al., 2013; Barcena et al., 2014). This is because of the relatively large size of the initial SOC stock and the inherent spatial heterogeneity which may make it difficult to significantly detect relatively small changes in SOC in the initial years following afforestation. It was therefore important for the present study to include the Heiðmörk site, where older planted forests were also found.

It can be emphasised that the annual SOC sequestration rate was determined mainly by a combination of factors such as age, forest growth conditions and other environmental factors and this is supported by chronosequence observations of the different forest types.

4.6. Assessing ecosystem changes using basal area or stem volume instead of age

It was clear that age was not very accurate predictor for ecosystem changes when all the sites were analysed together, because of the large differences between forest types (Fig. 23). Therefore, I tested if basal area (BA) or standing stem volume could be better descriptors for ecosystem changes than age (Tables 10, 11 and 12). Unsurprisingly these variables were much better to describe tree biomass C-stocks across the whole dataset, as they are basically measures of the tree C-stock. More interestingly they were also much better to describe changes in litter and ground vegetation C-stocks than age (increased R^2 from 0.23 to 0.48 and 0.56 for litter C and from 0.09 to 0.35 and 0.32 for the reduction in ground vegetation C stock, respectively). However, neither BA nor standing stem volume could significantly predict any change in SOC-stocks across the whole dataset, as age could. Therefore, the general conclusion is that it would be preferable to use age-relationships found separately for different forest types. However, in absence of such forest type information or in mixed forests in another area in SW Iceland, the relationships found with BA or standing stem volume could be used.

4.7 C-sequestration dynamics in different chronosequences

4.7.1. Old growth, pure and mixed coniferous stands

I used a chronosequence approach to understand how respective forest type at each site accumulated C in above- and below-ground biomass just like Martin et al. (2005) did when they were studying C-stocks in northern boreal mixed forest chronosequences in Canada. One of the important motivating aims was to address whether the C-stocks and sequestration rates of old growth, pure and mixed coniferous stands differed in this study. I found that there was a greater C sequestration rate when trees were classed into individual forest types (chronosequences). For example, conifer had a sequestration rate of 75.6 g C m^{-2} yr^{-1} and birch was 16.8 g C m^{-2} yr^{-1} for above and below-ground compartments (SOC+Li+Ct and SOC+Li; Table 22). This finding is greater than the rate for Iceland as reported by Hellsing et al. (2016) but was consistent with Martin et al. (2005) who established a greater C-rates in soil and litter in deciduous stands when they stratified trees into mixed and deciduous. In their study, Martin et al. (2005) explained that the higher C-rates was attributed to multi-layered canopy in deciduous stands that supported foliage mass. In this study, the possible justification for higher C-rates in coniferous stands could be related to thinning management. The description of stand density and other forest characteristics (Table 5) showed that coniferous stands were thinned, thus facilitating higher deadwood and litter accumulation (Table 22). Seely et al. (2002) studied thinned forest stands in northern British Columbia and revealed that SOC-stocks reduced while C-rates increased, implying that the result of my study is consistent with others which confirmed that C-rates might not necessarily equate to C-stocks. I also suspect that the rates of C sequestration given in the Icelandic national inventory is based on unstratified forest study.

4.7.2. C-sequestration in planted versus naturally regenerating birch stands

At Nesjavellir I compared two birch forest strata, one which was planted and another which was naturally regenerated. I hypothesized that the planted downy birch forest would influence C-sequestration differently compared to naturally regenerating birch. All the measured control plots for both natural and planted forests, had a similar vegetative cover and soil properties (Table 3) indicating that they were comparable before the afforestation. It was interesting that the ecosystem C-rates (SOC+Li+Ct) were significant for the planted birch forest (132 g C m^{-2} yr^{-1}), but not for the naturally regenerated one (33 g C m^{-2} yr^{-1}; Table 22). There are no other Icelandic data available that compared C sequestration in planted and seeded birch forests with similar vegetation cover and soil properties prior to afforestation, as far as I know. However, naturally regenerated birch can have a significant C-sequestration rates in the top 30 cm of soil and in aboveground stocks as shown by Snorrason et al. (2002) and Hunkziker et al. (2018), but in both cases the naturally regenerated stands were older than in the present study. The reason for these apparent differences seem to be related to much higher annual biomass growth of the planted birch material, compared to the naturally regenerated one in the present study (Table 5). That might both be related to different genetic properties of Icelandic birch (Thorsson et al., 2010) that could affect photosynthetic yields (Unwin and Kriedemann, 2000) or due to the enhanced growth of the nursery pre-cultivation of the planted material. What was the reason for the different biomass growth rates of those two different plant material is difficult to say, since it may be affected by many variables including the baseline biomass C content, photosynthetic yields, microbial and other respiratory activity and root allocation and turn-over (Smith 2006; Kell 2011).

4.8. Methodological issues

4.8.1. The choice of method to measure C-stock

There are three types of forest carbon accounting methods that have been developed: i) stock accounting, ii) emissions accounting and iii) project emission reduction accounting (IGBP

Terrestrial Carbon Working Group, 1998), but this study uses stock accounting which estimate C stock changes in five pools as described by IPCC (2003, 2006). The IPCC has classified the C accounting methodological approaches into three Tiers. Tier 1 employs the gain-loss method as described in the IPCC Guidelines and the default emission- or sequestration factors and other parameters provided by the IPCC for a specific region (e.g. N-Europe). Tier 2 generally uses the same methodological approach as Tier 1, but applies emission- or sequestration factors and other parameters which are specific to a region, a country or specific forest type. Some of the findings of the present study may supply others with valuable Tier 2 relationships that may be used to estimate C-stock changes with age, basal area or standing stem volume in other forests growing in SW-Iceland. In this study, I however adapted Tier 3 methodology, which involves direct measurements of all the relevant forest C-stocks (full C accounting), but with limited input data for each site. Normally more plots would be used to estimate the aboveground C-stock changes, but another thesis (Gústaf Jarl Viðarsson, unpublished) will address those C-stocks in more detail.

The IPCC (2003, 2006) recognises commonly used stratification variables as forest types, age, soil type, slope and elevation. In the current study, I divided each site into as many as five homogenous units (strata) using forest type, regeneration method and height (as a surrogate for age). Although three old-growth birch plots were identified in the course of the study in Heiðmörk, the initial stratification was important as it increased the accuracy and precision of accounting by reducing the field data variability when only limited number of plots is to be measured. Andersson et al. (2009) observed that appropriate stratification can reduce the cost of accounting by diminishing sampling effort while maintaining the same level of statistics confidence and can, therefore, lead to more efficient implementation of the field measurements. Gathering field measurements for C accounting requires sampling of a subset of areas as complete enumerations are neither practical nor efficient (Smith et al. 2005). By definition, sampling infers information about an entire population by observing a fraction of it. In order to confidently scale up this data from plot scale to whole afforested level, I properly designed randomised sampling. MacDicken, (1997) agrees with this that for C accounting, stratified random sampling yields more precise estimates.

The most comprehensive method of tree C estimation is destructive sampling (Brown, 1997). However, in this study, single tree stem volume and biomass functions for the specific tree species were used to calculate stem volume and biomass (Snorrason and Einarsson 2006; Snorrason 2010; Jónsson and Snorrason 2018). When permanent plots are used, a destructive sampling cannot be used, as it would change how the plots will develop into the future.

4.8.2. Number of plots measure

Was the measurement effort enough to detect significant changes in SOC at all sites? I was especially interested to test if a minimum number of measurement plots (n=3) could be used to validate soil C-sequestration for individual forest owners. When the data were analysed separately for each forest type, significant changes were observed only in Heiðmörk coniferous stands where the number of plots were more than three for afforested plots, but forest types that had three plots, there was no significant change except when different variables (SOC+Li) were put together (Table 22). This observation reveals that minimum number of plots (n=3) can only be accepted at sites with relatively low variability in stands. Otherwise, if Tier 3 of the IPCC (2003) Good Practice Guide is to be adopted by individual forest owners in Iceland, more plots (n>9) are needed especially in stands with high variability. This study also confirms the importance of establishing permanent inventory plots which can facilitate effective field measurements and basic statistical computations that helps to quantify SOC and develop appropriate management plan to increase SOC stocks in their forest ecosystem.

4.8.3. Future estimates of C-stocks

Permanent plots were established according to procedure used by the European National Inventories (Gschwantner et al., 2016), purposely to repeat the measurements in the future. In a future inventory only the plots within the forested areas would be re-measured, since ecosystem changes would then be derived by comparison to past conditions. In terms of the soil inventory, this is likely to increase the accuracy of the C-stock change estimates since the estimate would not be based on differences between treeless and afforested plots.

Conclusion

The present study aimed to evaluate changes in ground vegetation cover, soil properties and ecosystem C-stocks at three afforestation sites in SW Iceland. For this, I assessed C stock and sequestration rates for different forest types and compared them to the adjacent treeless sites. The study reveals that, not only does afforestation lead to changes in vegetation cover and soil properties, but most importantly, it significantly sequesters substantial amounts of C from the atmosphere and stores it in both above-ground (standing tree biomass and bush) and below-ground C pools (litter and SOC).

- The result indicated that among the soil physical properties studied, bulk density would have had a very strong influence on SOC stock calculations if I had not sampled complete soil layers. An error in determining BD values can therefore strongly contribute to SOC uncertainty. I therefore recommend that studies focusing on SOC should ensure precise and accurate determination of BD by coring soil from the deepest section within the plot, avoid compaction during coring and careful handling soil in the laboratory.
- Another critical observation connected to BD was the soil mass correction. In the current study, there was a decrease of 4% in the sampled soil mass (fine fraction) of the whole 30 cm soil layer, but the changes were larger in topsoil that led to relatively large changes in the total SOC changes in response to afforestation. It is therefore recommended that such soil mass corrections be done in all future studies on the effects of afforestation on SOC stocks.
- Afforestation by different tree species affected ground vegetation cover, biomass and vegetation C stock negatively. I conclude that it is therefore important to account for less ground vegetation C-stocks when ecosystem changes are estimated, albeit this stock was relatively small.
- The biggest relative change following afforestation, apart from the tree biomass, was witnessed in the litter C stock. I therefore suggest that models used to estimate forests ecosystem C stocks in Iceland should carefully consider this pool in their national C budgets.

- This study found a significantly larger total SOC stocks in afforested sites compared to treeless sites. The average changes (afforested-treeless) was 2,736 g C m^{-2} or 12% with an annual incremental rate of 41 g C m^{-2} yr^{-1} since afforestation started. These differences were obtained mainly from top soil at depth of 0-10 cm. Thus, the hypothesis that soil C-stock would respond positively following afforestation is supported.
- SOC stocks under different forest types (conifer, deciduous and mixed) were, on average, similar, however, there was some difference when conifer stands were compared with deciduous, but litter C accumulation is directly linked to the differences in soil C balance between the forest types. Thus, the hypothesis that there would be variations in SOC stocks under different forest types was not confirmed. The reason to explain the similarity in SOC stocks under different forest types in unclear but are confirmed by various studies from higher latitudes.

Forest inventory is a cornerstone of forest planning that gives land owners information about changes in ground vegetation, soil properties, forests productivity and crucially, changes in ecosystem C stocks. In order to validate ecosystem C-stocks, it is suggested that individual forest owners in SW Iceland should group (stratify) their forests according to species, height, age and then place at least nine permanent measurement plots per strata. This study was based on single estimate between forest and control plots, so three plots were not enough to estimate changes in SOC, but if the whole ecosystem stocks is being compared, then it may be enough. It is therefore suggested that repeated surveys (at least five-years interval) may lead to stronger estimates.

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