www.nina.no

1774 NINA Report

Carbon storage in Norwegian ecosystems

Karbonlagring i norske økosystemer

Jesamine Bartlett, Graciela M. Rusch, Magni Olsen Kyrkjeeide, Hanno Sandvik & Jenni Nordén

NINA

Norwegian Institute for Nature Research

NINA Publications

NINA Report (NINA Rapport)
This is NINA's ordinary form of reporting completed research, monitoring or review work to clients. In addition, the series will include much of the institute's other reporting, for example from seminars and conferences, results of internal research and review work and literature studies, etc. NINA

NINA Special Report (NINA Temahefte)
Special reports are produced as required and the series ranges widely: from systematic identification keys to information on important problem areas in society. Usually given a popular scientific form with weight on illustrations.

NINA Factsheet (NINA Fakta)
Factsheets have as their goal to make NINA's research results quickly and easily accessible to the general public. Fact sheets give a short presentation of some of our most important research themes.

Other publishing.
In addition to reporting in NINA's own series, the institute's employees publish a large proportion of their research results in international scientific journals and in popular academic books and journals.

Carbon storage in Norwegian ecosystems

Karbonlagring i norske økosystemer

Jesamine Bartlett
Graciela M. Rusch
Magni Olsen Kyrkjeeide
Hanno Sandvik
Jenni Nordén

Norwegian Institute for Nature Research

Bartlett, J., Rusch, G.M., Kyrkjeeide, M.O., Sandvik, H. & Nordén, J. 2020. Carbon storage in Norwegian ecosystems. NINA Report 1774. Norwegian Institute for Nature Research.

Trondheim, March 2020

ISSN: 1504-3312
ISBN: 978-82-426-4531-9

AVAILABILITY
Open

PUBLICATION TYPE
Digital document (pdf)

QUALITY CONTROLLED BY
Research director Signe Nybø

SIGNATURE OF RESPONSIBLE PERSON
Research director Kristin Thorsrud Teien (sign.)

CLIENT(S)/SUBSCRIBER(S)
WWF Norway

CLIENT(S) REFERENCE(S)
3701

CLIENTS/SUBSCRIBER CONTACT PERSON(S)
Jon Bjartnes

COVER PICTURE
Permafrost tundra & mire – Varangerhalvøya, Finnmark © Jesamine Bartlett 2019

KEY WORDS
Carbon sequestration, Norway, greenhouse gases, soil, forest ecosystems, alpine ecosystems, cryosphere, peatland ecosystems, lowland ecosystems, aquatic ecosystems, oceanic ecosystems

NØKKELORD
Karbonopptak, Norge, klimagasser, jord, skogøkosystemer, alpine økosystemer, kryosfære, myrøkosystemer, åpent lavland, ferskvannøkosystemer, marine økosystemer

CONTACT DETAILS

NINA head office	NINA Oslo	NINA Tromsø	NINA Lillehammer	NINA Bergen:
P.O.Box 5685 Torgarden	Gaustadalléen 21	P.O.Box 6606 Langnes	Vormstuguvegen 40	Thormøhlens gate 55
NO-7485 Trondheim	NO-0349 Oslo	NO-9296 Tromsø	NO-2624 Lillehammer	NO-5006 Bergen.
Norway	Norway	Norway	Norway	Norway
P: +47 73 80 14 00	P: +47 73 80 14 00	P: +47 77 75 04 00	P: +47 73 80 14 00	P: +47 73 80 14 00

www.nina.no

Abstract

Bartlett, J., Rusch, G.M., Kyrkjeeide, M.O., Sandvik, H. & Nordén, J. 2020. Carbon storage in Norwegian ecosystems. NINA Report 1774. Norwegian Institute for Nature Research.

This report discusses approximate estimations of the carbon budgets within Norway's mainland ecosystems. It stands as an initial overview of the natural potential of carbon storage and sequestration in Norwegian ecosystems. We describe carbon cycling in five key ecosystem groups: forest, alpine and cryosphere, agriculture and grassland, wetland, and freshwater and near-shore ecosystems. We emphasise the vital ecosystem service that Norwegian landscapes and ecosystems provide in sequestering carbon, and how climate change and management practices may aggravate or mitigate this function. We find that the largest stores of carbon in Norway are in the forests (32%) which also cover 38% of the total land area. Wetlands and permafrost cover 9% and 3% of the total land mass respectively, yet are storing over 2.2 Pg C, 31% of the nation's carbon. These two ecosystems are the most carbon dense ecosystems per km^2, with 53 and 48 kg C m^{-2} for wetlands and permafrost respectively. The next densest storage of carbon can be found in freshwater lake sediments, with 45 kg C m^{-2}, amounting to 13% of all carbon stores. Forests and low-mid alpine zones sequester the most carbon on an annual basis (5.5 and 5.3 Tg C yr^{-1}, respectively), with soils in alpine heathlands contributing the most to alpine carbon stores. In considering the carbon stored in key ecosystems we find that Norway contains approximately 0.18% of all global carbon stocks, with a land mass that is 0.07% of the planet. This high carbon to area ratio is likely due to the large proportion of the country that is carbon rich peatlands (alpine and lowland) and boreal forest.

Since ratifying the Paris Agreement, Norway has pledged to become carbon neutral by 2050, yet is presently one of the highest CO_2/CO_2-e emitters per capita in Europe, and within the top 20% of emitters globally. Terrestrial ecosystems that are included in the emissions reporting system for Norway include productive forest and land-use changes therein, arable land and farm grazing land, infrastructure areas, and a small portion of the total area of mire. However, a large portion of the remaining land area in Norway is not included in the accounting, but whose carbon emissions and sink capacity can be significantly affected by management practices and/or conversion. Currently non-managed areas such as wetlands, alpine zones, mountain forest, freshwater sediments, habitats included in non-agricultural open lowland classes, and the cryosphere including permafrost, are not considered in carbon reporting, or accurate land cover estimates. These areas account for more than half of the land cover of Norway and could account for approximately 68% of the nation's carbon stores. Additionally, coastal ecosystems, such as kelp forests are also not included, yet play a key role in both carbon budgets and biodiversity measures.

The Intergovernmental Panel on Climate Change (IPCC) finds that the conservation and enhancement of carbon sinks, and natural carbon stores is one of the surest ways for us to combat the extremes of climate change. The most efficient and cost-effective process is by using existing ecosystems. Current national inventories do consider the changes in land use, and how this may impact carbon emissions. However, much of the regularly assessed land types are biased towards managed ecosystems and there is currently no framework for how to incorporate impacts on biodiversity. The loss of biodiversity is accelerating and that has negative consequences for populations, species, communities and ecosystems, and thus ecosystem services, including those underpinning the capacity for climate mitigation and adaptation. The recent reports from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES Global Assessment 2019), and Intergovernmental Panel for Climate Change (IPCC Land report 2019) both point to biodiversity and ecosystems as underpinning climate actions, and the necessity of developing mixes of instruments that make best use of synergistic opportunities that can motivate land-owners and other decision-makers to make decisions that both conserve biodiversity and ecosystem integrity, and deliver high levels of ecosystem services, including reduced greenhouse gas emissions and increased removals. Ensuring a diverse portfolio of healthy ecosystems, either through conservation of already

existing ones or by restoring degraded ones, will have the greatest value of ecosystem services and ensure the highest chance of adaptability to climate change pressures in the future. The ability of non-managed and seemingly unproductive ecosystems, such as alpine landscapes to sequester and store carbon is significant.

We suggest that in addition to a 'Klimakur' ("climate cure"), there is a need for a 'Naturkur' ("nature cure") to implement a strategy for biodiversity and ecosystem services following-up the findings and recommendations from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), the new international commitments under the Biodiversity Convention (CBD), and the national implementation of the Norwegian "Nature for Life" white paper (Meld. St. 14 (2015-2016), Ministry of Climate and Environment 2015). A Naturkur would emphasise the value of maintaining a diverse portfolio of ecosystems at a national level, ecosystems that are inextricably interlinked with carbon storage, sequestration capacity and biodiversity itself, and of finding solutions that can help achieve multiple objectives by proposing synergistic measures. Or rather, a harmonized Klima-Naturkur, where actions for climate mitigation and adaptation, and for biodiversity and ecosystem services conservation are not designed independently, but address societal challenges in a coordinated manner, are synergistic and reinforce each other to achieve multiple benefits.

Jesamine Bartlett (jesamine.bartlett@nina.no), Graciela M. Rusch (graciela.rusch@nina.no), Magni Olsen Kyrkjeeide (magni.kyrkjeeide@nina.no), Hanno Sandvik (hanno.sandvik@nina.no). Norwegian Institute for Nature Research (NINA), P.O. Box 5685 Torgarden, 7485 Trondheim. Jenni Nordén (jenni.norden@nina.no). Norwegian Institute for Nature Research (NINA), Gaustadalléen 21, 0349 Oslo.

Sammendrag

Bartlett, J., Rusch, G., Kyrkjeeide, M.O., Sandvik, H. & Nordén, J. 2020. Karbonlagring i norske økosystemer. NINA Rapport 1774. Norsk institutt for naturforskning.

Denne rapporten presenterer omtrentlige estimater av karbonbudsjettene i Fastlands-Norges økosystemer. Den gir en innledende oversikt over det naturlige potensialet for karbonopptak og -lagring i norske økosystemer. Vi beskriver karbonkretsløpet i fem viktige økosystemgrupper: skog, fjell (inkl. kryosfære), åpent lavland (inkl. jordbruksareal), våtmark og ferskvann/kyst. Vi fremhever den viktige økosystemtjenesten som norske landskap og økosystemer yter ved å lagre og binde karbon, og hvordan klimaendringer og forvaltningspraksis kan forverre eller dempe denne funksjonen. Våre estimater viser at det største karbonlagret i Norge ligger i skog (32 %), som også dekker 38 % av det totale landarealet. Våtmark og permafrost dekker henholdsvis 10 % og 3 % av den totale landmassen, men lagrer allikevel over 2,2 Pg C, som tilsvarer 31 % av landets karbon. Disse to økosystemene er de mest karbontette økosystemene per km^2, med henholdsvis 53 og 48 kg C m^{-2} for våtmarker og permafrost. I innsjøsedimenter finnes 45 kg C m^{-2}, som utgjør 13 % av all karbonlagring. Skog og lav- og mellomalpin sone tar opp mest karbon på årsbasis (henholdsvis 5,5 og 5,3 Tg C per år), med alpine lyngheier som naturtypen som bidrar mest i fjellets karbonlager. Våre estimater viser at Norge totalt har omtrent 0,18 % av de globale karbonlagrene, med en landmasse som tilsvarer 0,07 % av jordoverflaten. Dette skyldes sannsynligvis den høye dekningen av karbonrike myrer og boreale skoger.

Siden godkjenningen av Parisavtalen har Norge forpliktet seg til å bli karbonnøytral innen 2050, men har i dag et av de høyeste CO_2(-ekvivalent)-utslippene per innbygger i Europa, og er dermed blant de 20 % av verdens land med høyest utslipp. Utslippsrapporteringssystemet for Norge omfatter (bruksendringer i) produktiv skog, jordbruks- og beitemark, infrastrukturområder og en liten del av det totale myrområdet. En stor del av Norges øvrige arealer er imidlertid ikke omfattet av karbonregnskapet, selv om forvaltning og/eller bruksendringer har stor betydning for deres karbonutslipp og -opptaksevne. For øyeblikket tar f.eks. ikke karbonrapportering og arealstatistikk høyde for ikke-forvaltede arealer, slik som våtmarker, permafrost, alpine soner, fjellskog, ferskvannssedimenter eller åpent lavland utenom landbruksareal. Disse områdene utgjør mer enn halvparten av Norges areal og kan utgjøre omtrent 68 % av landets karbonlager. Heller ikke kystøkosystemer som tareskog er inkludert, selv om disse spiller en nøkkelrolle for både karbonbudsjetter og biologiske mangfold.

Ifølge FNs klimapanel (IPCC) er bevaring og forbedring av naturlige karbonfangere og karbonlagre en av de sikreste måtene å bekjempe de mest ekstreme klimaendringene på. Den mest kostnadseffektive måten er ved å bruke eksisterende økosystemer. Nåværende nasjonale karbonregnskap vurderer kun endringer i arealbruk og hvordan disse kan påvirke karbonutslipp. Ikke-forvaltede økosystemer er dermed sterkt underrepresentert, og deres betydning for naturmangfold blir heller ikke tatt høyde for. Tapet av biologisk mangfold er akselererende og har negative konsekvenser for bestander, arter, samfunn, økosystemer og dermed økosystemtjenester. Å sikre et mangfold av økosystemer med god tilstand, enten ved å bevare uberørte naturtyper eller ved å restaurere degradert natur, vil sikre den største verdien av økosystemtjenester og tilpasningsevnen til klimaendringer. Ikke-forvaltede og tilsynelatende uproduktive økosystemer, som alpine naturtyper og våtmarker, har en betydelig evne til å binde og lagre karbon.

Vi foreslår at det i tillegg til Klimakur, utredes en tilsvarende Naturkur. Målet bør være å implementere norsk handlingsplan for naturmangfold (Meld. St. 14 (2015-2016)), følge opp funn og anbefalinger fra det internasjonale naturpanelet (IPBES) og de nye globale målene som skal vedtas av Konvensjonen om biologisk mangfold (CBD) i oktober 2020. En Naturkur vil kunne bidra til at Norge opprettholder et mangfold av økosystemer i god økologisk tilstand, noe som er svært viktig for lagring og opptak av karbon. Naturkur bør blant annet inneholde oversikt over tiltak og løsninger som er bra for både naturmangfold og klima. Naturkur bør inneholde særskilte kapitler som kombinerer Klima- og Naturkur, hvor tiltak for klimatilpasning og bevaring av biologisk mangfold og økosystemtjenester ses i sammenheng, gir synergier og forsterker hverandre.

Jesamine Barlett (jesamine.bartlett), Graciela Rusch (graciela.rusch@nina.no), Magni Olsen Kyrkjeeide (magni.kyrkjeeide@nina.no), Hanno Sandvik (hanno.sandvik@nina.no). Norsk institutt for naturforskning (NINA), Postboks 5685 Torgarden, 7485 Trondheim. Jenni Nordén (jenni.norden@nina.no). Norsk institutt for naturforskning (NINA), Gaustadalléen 21, 0349 Oslo.

Contents

Foreword

In this report we summarize the knowledge on carbon storage and sequestration in Norwegian nature. We evaluate the potential of carbon storage and sequestration in Norwegian ecosystems, and the effects of anthropogenic and natural factors on the carbon cycle and storage. The report is a result of an assignment from WWF Norway who wished to make available a knowledge base on carbon storages in Norwegian ecosystems.

The project was small in extent which limited the detail we could include for this vast and complex topic. We aimed at covering the main ecosystem types, and the main factors and processes influencing carbon storage and sequestration in these. However, the details given are not exhaustive. We highlight the uncertainties related to carbon storage and sequestration and their potential in Norway. The effects of emission mitigation measures suggested in Klimakur on ecosystems and their carbon fluxes are discussed, and a Naturkur is proposed to implement the actions following the IPBES assessments.

All the authors contributed to all parts and critical assessment of the contents of the report. Jenni Nordén led the project.

We thank Erik Framstad for valuable comments on the whole report, and Marte Fandrem on the wetland section.

Jon Bjartnes has been our main communication partner for the project commissioners at WWF Norway. WWF has contributed to the selection of elements included in this report, but the contents and orientation of the work were decided upon by the authors. We thank WWF for a constructive and good communication.

Oslo, 2.3.2020

On behalf of the project team,

Jenni Nordén
Project leader

1 Introduction

1.1 Background

The natural carbon cycle exchanges carbon dynamically between the land, ocean, and atmosphere over years or even millennia. The carbon that enters and is sequestered, or leaves through respiration or export, is known as carbon flux, and the rate at which carbon flows through a biome, or habitat, is carbon turnover. Understanding the turnover in different ecosystems can highlight where carbon is vulnerable to release, ultimately as a greenhouse gas (GHG) to the atmosphere. GHGs most often associated with human activity are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) – these gases absorb infrared radiation in the atmosphere, leading to trapped heat and surface warming. Globally, land-use change drives 23% of anthropogenic GHG emissions that come from agriculture and forestry and other land uses (IPCC 2019). Management of ecosystems based on knowledge of their turnover is important in influencing the flux of carbon to and from ecosystems, and ultimately is a critical defence against further climate change.

Estimates of carbon stocks vary, and there is much uncertainty about how carbon storage changes with temperature, moisture and vegetation (Gonzalez-Domingues et al. 2019). Nevertheless, there is consensus in that the amount of carbon in soil represents most of the carbon found in terrestrial ecosystems. Ontl and Schulte (2012) estimate that nearly 80% (2.5 Eg C, see **Table 1** for units used) is found in soil and that the amount of carbon found in living plants and animals is comparatively small (0.56 Eg C) relative to that found in soil. Further, the global soil carbon pool is approximately 3 times larger than the atmospheric pool.

Crucial to terrestrial carbon balances will be the availability of key nutrients such as nitrogen. Nitrogen is often the nutrient limiting the growth of organisms in many terrestrial ecosystems, and nitrogen fertiliser is therefore commonly applied in forestry and agriculture. The long-term consequences of fertilisation and the changes in soil communities on nutrient cycling, soil productivity and climate regulation (GHG emissions) are insufficiently known (Li et al. 2019), especially in interaction with the effects of climate change. However, excessive nitrogen fertilisation, such as is seen in managed croplands, can inadvertently increase GHG release by altering the nitrogen cycling pathway resulting in an increase in N_2O gas, a GHG 300 times more powerful than CO_2 (Snyder et al. 2009, Aarrestad et al. 2013).

Emissions associated with human activity occur in addition to the natural carbon cycle. The burning of fossil fuels has increased the atmospheric CO_2 concentration by 32% (from 277 ppm in 1750 to 407 ppm in 2018; Friedlingstein et al. 2019). Greenhouse gas emissions as a result of human activity have already increased global temperatures by ca. 1°C since the pre-industrial era. Looking forward, it is likely to reach 1.5°C by 2030–2052, with the speed of change causing unprecedented levels of change on our weather systems, and subsequent stress within our ecosystems (IPCC 2018). In order to limit further warming to at least that 1.5°C level, as per the Paris Agreement which Norway ratified in 2016 (Government 2016), significant action needs to be taken at global, national and local levels, to retain existing carbon stocks, enhance sequestration, and limit further carbon release. Carbon release can be in the form of burning of fossil fuels, land-use change, such as deforestation, increased agriculture, and the loss of wetlands, the melting of the cryosphere, the transformation of the oceans from carbon sinks to carbon sources, or many other ways in which the CO_2 output of humans is overwhelming ecosystems (**Figure 1**).

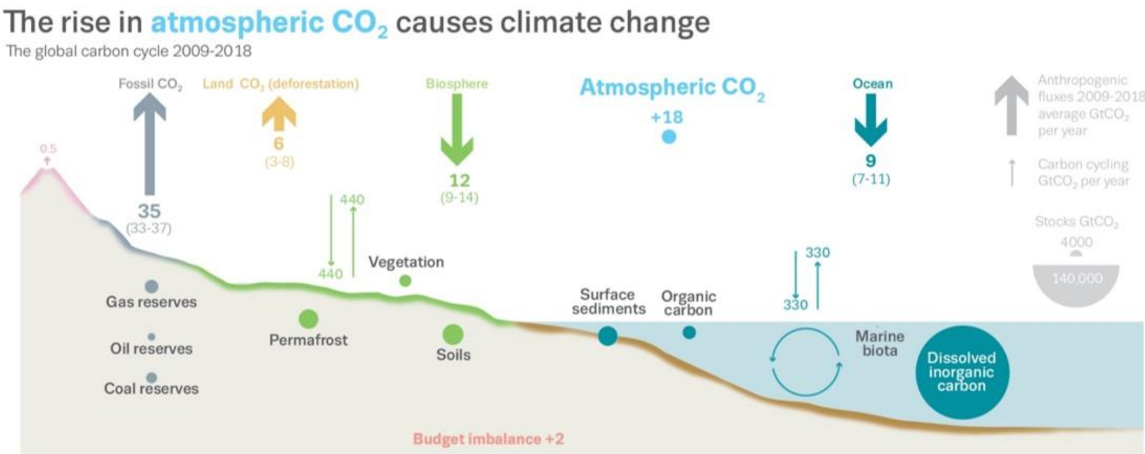

Figure 1. *The perturbation of the global carbon cycle caused by anthropogenic activities, averaged globally for the decade 2009–2018 (Pg CO_2 yr^{-1}). Extraction use of fossil fuels drive CO_2 emissions, with land use change the next biggest factor in affecting climate change. Produced by the Global Carbon Project based on Friedlingstein et al. (2019).*

The International Panel on Climate Change (IPCC 2018) finds that the conservation and enhancement of carbon sinks, and natural carbon stores is one of the surest ways for us to combat the extremes of climate change (UNFCC 2015). The most efficient and cost-effective process is by using existing ecosystems (Villa & Bernal 2017). The scale of using ecosystems as a mitigator for carbon emissions depends on the “pursued mitigation portfolio” of national governments (IPCC 2018): overall, limiting emissions would need to be coupled with carbon dioxide removal (CDR) from the atmosphere in the range of 0.1–1 Eg CO_2, for the 1.5°C limit to be met. For ecosystems, CDR methods can include both afforestation and reforestation, land restoration and conservation measures that encourage soil carbon sequestration and oceanic carbon burial.

Carbon budgets measure the balance, or imbalance, of carbon emissions to carbon sequestration or storage. Globally, there are approximately 43.5 Eg of carbon stored in the planet’s ecosystems (**Figure 2**). In order to preserve that storage and the ability of ecosystems to continue to contribute to carbon uptake, we need to limit our emissions from these ecosystems to a total of 0.8 Eg CO_2 (UNFCC 2015). This is our best chance of limiting climate change to less than 2°C above the pre-industrial period. However, global emission levels are 0.04 Eg CO_2 yr^{-1} – this could mean that unless greater carbon sinks are created, or emissions are substantially reduced, our remaining global carbon budget of 0.8 Eg will be gone in just 20 years (CICERO 2017).

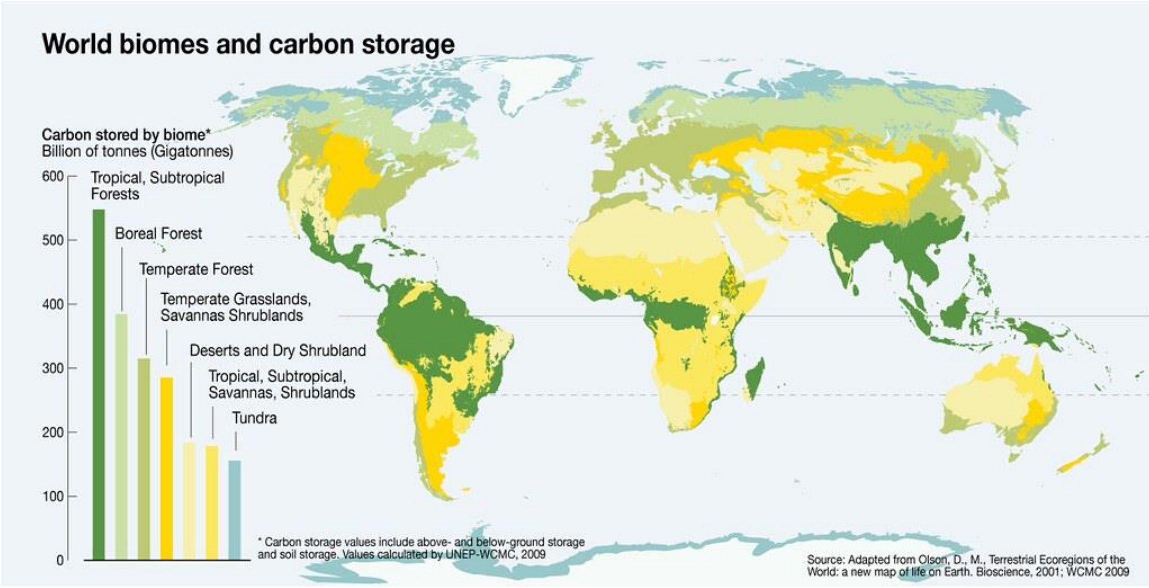

Figure 2. *Carbon storage in global biomes (excluding oceans; GRIDA 2015).*

1.2 The scope of the work and definitions

This report discusses approximate estimations of the carbon budgets within Norway's mainland ecosystems. We examine carbon cycling in each of the five key ecosystem groups. We emphasise the vital ecosystem service that Norwegian landscapes and ecosystems provide in sequestering carbon, and how climate change and management practices may aggravate or mitigate this function. This report is intended as a brief summary, rather than a detailed analysis from which definitive conclusions can be drawn. Rather, this report can be used to provide an overview of the potential of carbon storage within key Norwegian ecosystems and suggested ways to preserve or encourage the sequestration and storage within them. To give reference as to the state of biodiversity within a discussed ecosystem, we use the Norwegian Nature Index system: Nature Index values are rated between 1 (reference state) and 0 (very poor state), and have given insight into the changing biodiversity within an ecosystem over the last decade (Framstad 2015).

1.2.1 Ecosystems

Figure 3 shows the proportion of land types and key ecosystem groups that make up mainland Norway. For this report we have categorised ecosystems in Norway into five groups, with approximate areas based on land cover statistics from Statistics Norway (2019a), or from primary literature as mentioned, as follows: forest (121,000 km^2, Statistics Norway 2019a; Bryn et al. 2018[1], alpine and cryosphere (107,000 km^2, Norwegian Water Resources and Energy Directorate 2019; Bryn et al. 2018[1]), open lowlands, including heathlands, grasslands and croplands (18,000 km^2, Statistics Norway 2019a; Bryn et al. 2018[1]), and wetlands (28,000 km^2, Bryn et al. 2018[1] – although this could be an underestimation, e.g. peatland forest not included [ca. 13,000 km^2, Bryn et al. 2018[1]], see **Section 2.4**). Aquatic ecosystems are discussed as fresh water, coast and the seabed (20,000 km^2 for freshwater, Statistics Norway 2019a; Bryn et al. 2018[1]) (**Figure 3**). However, the diversity within Norwegian ecosystems is far greater, and in 2009 a national classification system identified 68 major types of habitat from coast to mountain top (Norwegian Biodiversity Information Centre 2018). Within each of these, carbon budgets will

[1] Inclusion of results from Bryn et al. (2018), in addition to other literature indicates that the two results concur within the reported margin of error.

differ, as will their response to climate change and land use with factors such as microclimate, temperature and moisture, which are some of the largest drivers of terrestrial carbon flux (e.g. Cahoon et al. 2012). Thus a broader scope is necessary at a national level. Our classification of 'alpine' ecosystems can be summarised as everything above the treeline (e.g. Austrheim et al. 2010) divided into the three alpine vegetation zones, plus the nival zone, exclusive of ice cover. The cryosphere includes mainland glacier cover and permafrost. Meanwhile, forests include boreal/sub-alpine and boreonemoral (mixed deciduous and evergreen) forests. Freshwater and near-shore aquatic ecosystems are discussed as freshwater lakes and rivers, and coastal habitats as kelp forests, intertidal algae, seagrass meadows, saltmarshes and intertidal mudflats. Agricultural and semi-natural grasslands are discussed alongside 'open lowlands', whilst wetlands/peatlands are those that cover both lowland and upland areas. This particular ecosystem type will overlap with tundra/permafrost mire in the alpine region analysis, and the results from each will need to be considered with this in mind – separation of the two is beyond the scope of this current report.

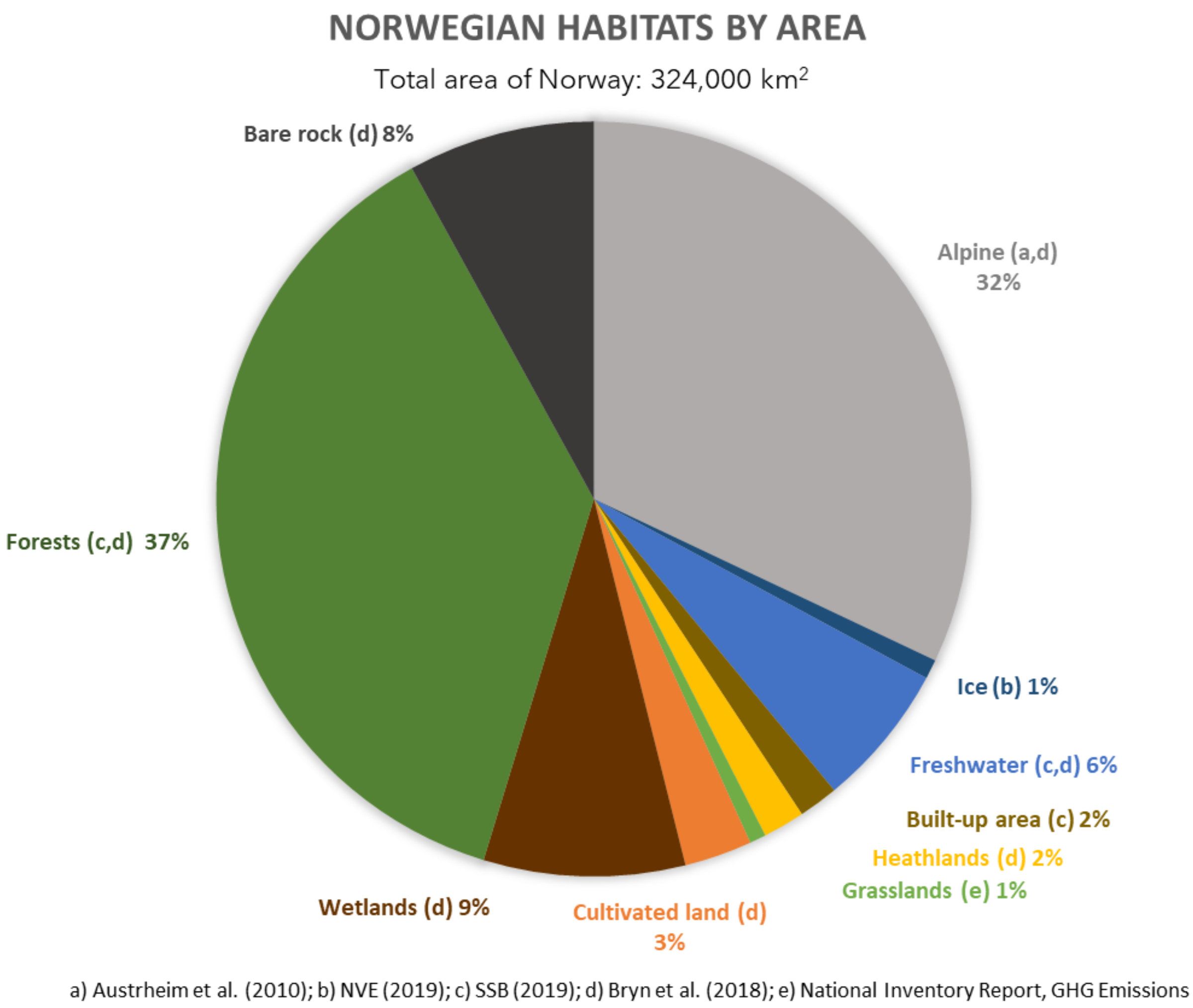

***Figure 3**. Key habitat types within mainland Norway and their approximate percentage land cover of total mainland surface area – where multiple sources agree within their own stated margins of error, both are listed, otherwise the most recent state sourced data is used (i.e. Norwegian Water Resources and Energy Directorate [NVE] or Statistics Norway [SSB]). Excludes coastal ecosystems.*

1.2.2 Carbon terminology and conversions

The **carbon cycle** is the complex series of reactions by which carbon passes through the Earth's atmosphere, biosphere, pedosphere, hydrosphere, lithosphere and cryosphere (the climate system). **Carbon removal** results from the capacity of plants to absorb and retain CO_2 from the atmosphere through the process of photosynthesis. **Emission** takes place for instance when plants die and decay, while **storage** takes place for example when organic material builds up in soils. **Carbon sequestration** is the uptake and long-term storage of carbon in a reservoir. It can refer to, for example, carbon reservoirs in the soil or dead wood. **Carbon Dioxide Capture and Storage (CCS)** is a process in which a relatively pure stream of carbon dioxide (CO_2) from industrial and energy-related sources is separated (captured), conditioned, compressed and transported to a storage location for long-term isolation from the atmosphere.

Carbon sink is any reservoir (for example ecosystem) that removes carbon released from some other part of the carbon cycle. **Carbon source** is any process, activity, or mechanism that releases carbon to another part of the carbon cycle. **Carbon stock** is the absolute quantity of substance of concern (for example, carbon or a greenhouse gas) held within a reservoir at a specified time. A reservoir is a component of the climate system, other than the atmosphere, which has the capacity to store, accumulate, or release a substance of concern (for example vegetation, soils, oceans).

Carbon dioxide equivalent (CO_2-e) is a measure used to compare the emissions from various greenhouse gases based upon their global warming potential. The carbon dioxide equivalent for a gas is derived by multiplying the mass of the gas by the associated global warming potential (relative to CO_2). Carbon may also be used as the reference, and other greenhouse gases may be converted to carbon equivalents. To convert carbon to carbon dioxide, the mass of carbon is multiplied by 44/12 (the ratio of the molecular weight of carbon dioxide to carbon).

All references to the mass of carbon or of CO_2-e will be expressed in multiples of gram as per **Table 1**. Please see the Glossary (**Appendix 7.1**) for further information on terms and expressions.

Table 1. *Conversion of different units to measure carbon stocks or fluxes. Measurements can be in multiples of grams (g) or metric tons (t). Note that for alternative units given in italics, conversion factors differ from 1 (bold in last column).*

Unit	Meaning	Alternative units	Conversion
Mg	megagram	[metric] ton	1 Mg = 10^6 g = 1 t
Gg	gigagram	Kiloton	1 Gg = 10^9 g = 10^3 t = 1 kt
Tg	teragram	Megaton	1 Tg = 10^{12} g = 10^6 t = 1 Mt
Pg	petagram	Gigaton	1 Pg = 10^{15} g = 10^9 t = 1 Gt
Eg	exagram	Teraton	1 Eg = 10^{18} g = 10^{12} t = 1 Tt
$g\,m^{-2}$	gram per square metre	*kilogram per hectare*	$1\,g\,m^{-2}$ = **10** $kg\,ha^{-1}$ = $1\,t\,km^{-2}$
$kg\,m^{-2}$	kilogram per square metre	*ton per hectare*	$1\,kg\,m^{-2}$ = **10** $t\,ha^{-1}$ = $1\,kt\,km^{-2}$
$Gg\,yr^{-1}$	gigagram per year	*ton per day, gram per second*	$1\,Gg\,yr^{-1}$ = **2.7** $t\,d^{-1}$ = **32** $g\,s^{-1}$
g C	gram of carbon	*gram of CO_2*	1 g C = **3.67** g CO_2
g CO_2	gram of CO_2	*gram of carbon*	1 g CO_2 = **0.273** g C

2 Habitat types and management practices

2.1 Forest

Forests in Norway make up approximately 38% of the total mainland area. 'Productive forest land', with an annual increment of > 1 m^3 ha^{-1}, covers 27% of the total mainland area. 'Poorly productive forest land', with an annual increment of < 1 m^3 ha^{-1} covers 11% (Storaunet & Rolstad 2015). Mountain birch forests are included in forest land in some cases, such as for 'Land Use, Land-Use Change and Forestry (LULUCF), then increasing the area considered as forest land to 44%. Forestry operates primarily in the productive forest land. Since the 1920's, Norway has tripled its standing timber stocks to the present day 900 million m^3, and the industry now supports around 25,000 people (Government 2014). However, forestry activities include the addition of roads, and fertilisation, thus the managed nature of much of forested land cover makes it one of the most influential ecosystems to both biodiversity and climate change, in the country. Sixty per cent of the 44,000 known species in Norway live in forests, and 48% (1122 species) of the threatened species in Norway are forest species (Henriksen & Hilmo 2015). However, despite the high species richness, forest ecosystems have a low Nature Index rating (NI) of 0.37 in 2014 (Framstad 2015). The low value is due to many indicators with values rather far from the reference value, which has been influenced by both forestry and large carnivore management.

2.1.1 Carbon cycle in forest ecosystems

Trees have a major role in the forest ecosystem carbon cycle. Trees accumulate, through photosynthesis, large amounts of carbon during their lifetime. The carbon stored in trees is later relocated in the forest soil, where it accumulates over long periods of time, eventually forming very large soil carbon storages (**Figure 4**). The dominant tree species in Norway, Norway spruce (*Picea abies, gran*) and Scots pine (*Pinus sylvestris, furu*) have long natural lifespans: up to 300–500 years in spruce and 500–700 years in pine. Silver birch (*Betula pendula, hengebjørk*) and downy birch (*B. pubescens, vanlig bjørk*) are also common, and reach ages of up to 150 years. The pedunculate oak (*Quercus robur, sommereik*) and the sessile oak (*Q. petraea, vintereik*) are able to form forests in southern Norway. Oaks may live hundreds of years, even up to a thousand years. The processes of tree death and decomposition may be very slow in the oaks and the pine; a decorticated kelo pine may stand dead for 500 years before falling down where it may remain for at least decades before decomposing and becoming incorporated into the soil (Niemelä et al. 2002). In the other tree species, the process of death is usually faster, but may take years. Storms, however, may kill healthy trees instantly. The complete decomposition of relatively small (diameter 11–16 cm) dead trees may take 40 years in birch and 85 years in pine and spruce (Mäkinen et al. 2006). The larger the dead tree, the longer the time needed for the decomposition process (Herrman et al. 2015), and consequently the decomposition of large logs (**Figure 5**) may take well over 100 years.

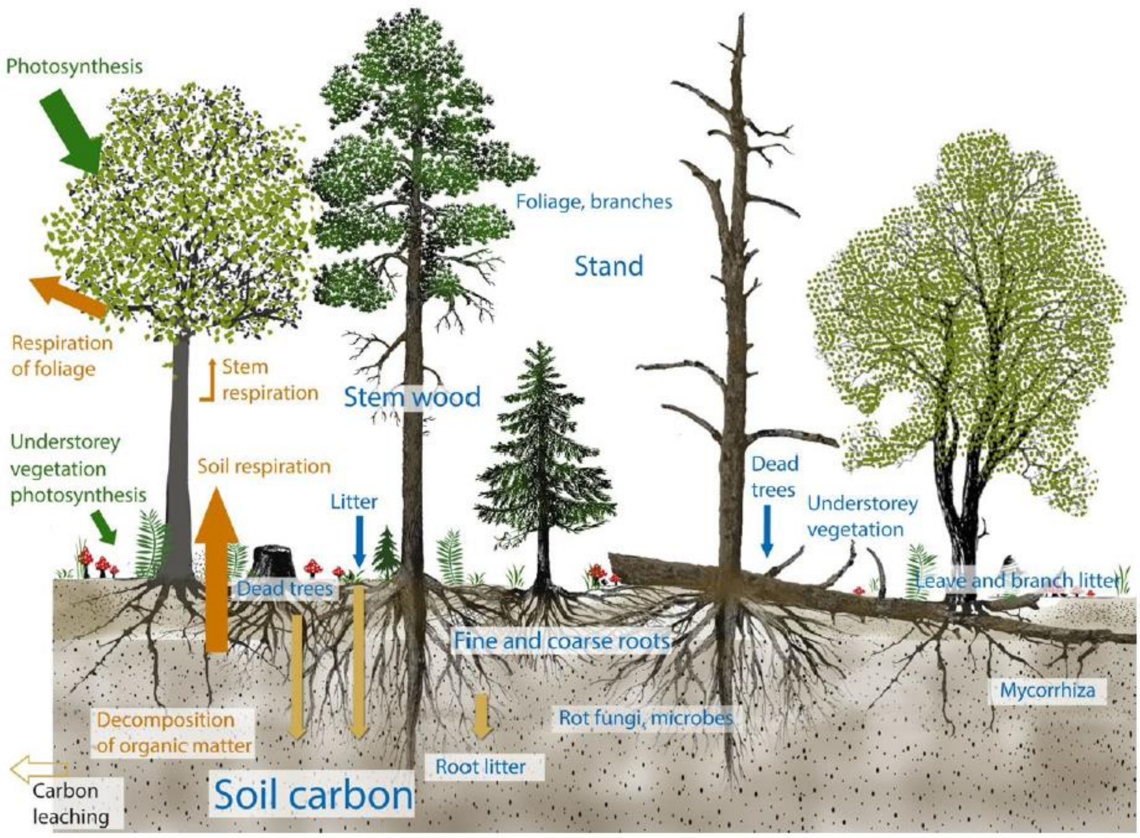

Figure 4. *The carbon cycle of the forest ecosystem. Source: Onarheim (2018).*

Dead organic material originating from trees and other plants (litter), fungi, animals and bacteria form the basis of soil organic matter and supply carbon to the soil. This carbon is partly lost into the atmosphere as CO_2 produced by heterotrophic respiration by soil organisms that govern the decomposition process. The remaining carbon persists in the soil and leads to the build-up of stable carbon in the process of assimilation. Assimilation and the closely related process of soil formation take at least decades, and result in major stocks of carbon in the forest soil (Gobin et al. 2011) that may become very old, for instance ca. 2500 years at 1 m depth (Clemmensen et al. 2013).

2.1.2 Carbon storage and sequestration in forest ecosystems

Figure 5. *Large dead trees sequester carbon for decades or even longer. Part of the carbon in dead wood joins the soil carbon stock in late stages of decomposition. The rest of the carbon is emitted to the atmosphere as CO_2 through respiration of saprotrophic organisms, in dead wood especially fungi and bacteria. Large dead trees host diverse and species rich communities of fungi, animals and bryophytes, many of which are red-listed. A single large dead tree may host tens to hundreds of species of fungi. This kind of logs also serve as nursery logs, providing a suitable seedbed and good conditions for the growth of tree seedlings. Large dead trees in advanced stages of decomposition are today mainly found in forest reserves. (Photo: Jenni Nordén)*

Boreal forest ecosystems hold the largest terrestrial carbon stocks globally and also in Norway (Rusch 2012, Bradshaw & Warkentin 2015). This is primarily because of trees that themselves are a major storage of carbon, and that create the large organic carbon storage of the forest soil both during their lifetime and after their deaths. During their life, trees transfer photosynthetic products below-ground into their roots and their mycorrhizal symbionts. Living trees also produce litter that is a carbon source for soil saprotrophic fungi and contributes to the soil organic matter. After tree death, the large carbon storage in the tree stem is sequestered for extended periods of time, until part of it joins the soil as organic matter.

The carbon storage of living trees in Norway is an estimated 0.5 Pg C (Søgaard et al. 2019). The carbon storage increases with living tree biomass, being the largest in the oldest trees. Therefore, old forests usually hold the largest carbon stocks (Framstad et al. 2013). However, forests older than 160 years cover only 2.5% of the productive forest land in Norway (Tomter & Dalen 2018). Forests consisting of two or more tree species tend to have higher carbon stocks than forests that consist of only one tree species (Rusch 2012, Liu et al. 2018). The carbon storage of living trees also increases with temperature and soil fertility (Grønlund et al. 2010). The yearly forest growth has more than doubled since the 1920s, and there are today 630 million m^3 more wood in Norwegian forests than 100 years ago (Dalen 2017). The increase in biomass and consequently also in carbon stock is due to silviculture, forestry planning, temperature increase and the depleted growing stock of Norwegian forests in the early 1900s, caused by overexploitation. The low timber stocks of forests in early 1900s prompted the start of the national forest stock monitoring (National Forest Inventory of Norway).

According to Pregitzer and Euskirchen (2004), soils of > 200 yr old boreal forests contain on average > 20 kg C m^{-2}, while younger boreal forests of age 71–120 yr, corresponding to the age range of final cutting in Norway, have an average carbon storage of ca. 10 kg C m^{-2}. Fennoscandian comparisons of carbon stocks in forests of different ages are few (see however Nord-Larsen et al. 2019). The Norwegian forest soil carbon storage is an estimated 1.83 Pg C within 1 m depth (Søgaard et al. 2019), which corresponds to 15 kg C m^{-2}. This is clearly more than reported by Pregitzer and Euskirchen (2004), especially when considering that 82% of

Norwegian productive forests are ≤ 120 yr old, while forests > 160 yr old cover only 2.5% of the productive forest land (Tomter & Dalen 2018). It is possible that northern European forests have higher soil carbon stocks than other boreal forests. Assuming that the global estimate of at least twice as large soil carbon stocks in boreal forests of age > 200 yr than in forests of age < 120 yr applies also for Norway, Norwegian forest soils probably contained 2.4–3.6 Pg C within 1 m depth before intensive human influence on forest soils. But now due to the extensive area covered by younger trees, this figure may be around 1.2 Pg C, which would be a lower estimation to the 1.83 Pg C estimated by Søgaard et al. (2019) (see also **Table 6**).

Norwegian forest soils currently hold 3–4 times more carbon than the biomass of the forest trees and understorey plants (Søgaard et al. 2019). This estimate is in line with a global estimate: according to Scharlemann et al. (2014), boreal forest soils possess up to 80% of the carbon storage of a forest ecosystem. Also fungal mycelia in dead wood and especially in soil hold large quantities of carbon and represent a below-ground input to the soil carbon storage (Clemmensen et al. 2015). Soil carbon storage of the boreal forest increases with age through accumulation. It also increases with temperature, probably because of higher input of organic material into the soil in productive southern forests, and potentially also because of reduced rate of decomposition in the south. Decomposition is expected to be faster in the warmer southern temperatures because of higher microbial activity, but it has been shown to be slower than expected, presumably because of greater deposition of nitrogen in the south than in the north (Stendahl et al. 2010, Framstad et al. 2013). Additional nitrogen in forest soils may slow down decomposition and consequently heterotrophic respiration (see section 2.1.3.3). In the north, production is lower and decomposition is slower because of lower temperature. The soil carbon stocks depend also on multiple physical site properties such as moisture: dry forests have a lower carbon storage than moist forests (Olsson et al. 2009).

Dead wood represents a carbon dense organic matter that has decreased considerably in Norway and most other areas in the world as a result of forestry and changing land use. For Norway, an average dead wood carbon stock of 500 g C m^{-2} was estimated (Norwegian Environment Agency 2019a), assuming a mean dead wood volume of 8.3 m^3 ha^{-1} on forest land. The dead wood volumes in natural forests in Fennoscandia range 60–120 m^3 ha^{-1} in the southern and middle boreal vegetation zones, 50–80 m^3 ha^{-1} in the northern boreal vegetation zone and ca. 20 m^3 ha^{-1} in the timberline (Siitonen 2001). Calculating the carbon stock of dead wood in a similar manner as the Norwegian Environment Agency (Norwegian Environment Agency 2019a), dead wood in natural forests can be estimated to hold 3.6–7.2 kg C m^{-2} in the southern and middle boreal, 3.0–4.8 kg C m^{-2} in the northern boreal and 1.2 kg C m^{-2} in the timberline in Fennoscandia. However, large dead wood carbon stocks currently occur almost exclusively in forest reserves which cover only 5% of the forest area in Norway (Norwegian Environment Agency 2019a). In support of the boreal Fennoscandian dead wood carbon estimates, a semi-natural beech forest reserve in Denmark was found to hold 2.1 kg C m^{-2} which corresponded to 6% of the forest ecosystem carbon (Vesterdal & Christensen 2007). Globally, an estimated 73 Pg of carbon is stored in dead wood, which makes up 8% of the current total carbon stock in the world's forests (Pan et al. 2011).

The carbon storage of living trees is relatively well known at the national level because of country-wide data on the distributions of ages, volumes and tree species of forest trees, provided by the National Forest Inventories, and the availability of direct measurements of the living wood carbon content. In contrast, there is considerable uncertainty about the size and dynamics of the soil carbon storage, as systematically collected representative data are not available (de Wit et al. 2015). Norway is, however, considering (Svendgård-Stokke et al. 2019) to collect country-wide soil carbon data as a contribution to the Global Soil Organic Carbon Map by the Food and Agriculture Organization of the United Nations (FAO). The size and variation in the dead wood carbon content is also based on coarse estimates instead of exact measurements (Norwegian Environment Agency 2019a).

Forests are estimated to account for 94% of the terrestrial carbon uptake in Norway (de Wit et al. 2015). This is not only because of the large proportion of land that the forests cover (38%), but also because the carbon uptake by unit area is much higher in forests (49 g C m^{-2} yr^{-1}) than for instance in peatlands (19 g C m^{-2} yr^{-1}) or any other ecosystem in Norway (de Wit et al. 2015). Living trees account for the majority of the carbon uptake, 40 g C m^{-2} per year, while the soil takes up 8.8 g C m^{-2} per year. The rate of carbon uptake is often assumed to be the highest in middle-aged trees and slowing down as the trees age because the net productivity of the tree decreases. However, several studies show that old trees and old-growth forest stands and soils still take up more carbon than they emit, and they therefore act as carbon sinks (Luyssaert et al. 2008, Gleixner et al. 2009, Wardle et al. 2012, see also Framstad et al. 2013). Stephenson et al. (2014) show for Norway spruce and several other tree species that mass growth rate increases continuously with tree size, making large old trees strong carbon sinks. Uptake of carbon in the soil is slower than in trees, but a process that continues hundreds or thousands of years or longer (Wardle et al. 2012). The formation time of fertile soil is therefore very long, beyond any management or policy-related time frame. Uptake of carbon in the forest stand may be halted at intervals of tens or hundreds of years or longer by stand-replacing disturbances such as a storm, fire or insect outbreak (Angelstam & Kuuluvainen 2004), or at present mainly by forestry operations. Rotation times in forestry (60–120 yr) are considerably shorter than intervals of natural stand-replacing disturbances (Kuuluvainen 2009).

2.1.3 Prevailing management practises and the use of forest biomass: effects on carbon cycle, storage and sequestration

2.1.3.1 Forestry

The majority (ca. 91%) of productive forest land in Norway is harvested by clear-cutting (with or without seed trees). Commercial thinning is done 1–2 times in young to middle-aged forests, and final felling is done when the age of the dominating trees is 60–120 years, depending on area, site fertility and tree species. Trees are therefore harvested much before their natural senescence. All or the majority of the harvested tree stems are removed from the forest. The effect of this practice on the forest ecosystem carbon cycle is considerable, as transporting biomass from the forest means removing a large stock of carbon (and nutrients) from the ecosystem. This leads to a substantial decrease in the input of carbon into the soil, especially with repeated logging cycles (Liski et al. 2003), in turn leading to smaller soil carbon stocks in production forests than in old-growth forests (Pregitzer & Euskirchen 2004). The clear-cut forest will be a carbon source for 10–20 years, as there is very little photosynthesis but the CO_2 fluxes from the soil are increased (Luyssaert et al. 2008, Alam et al. 2017).

Site preparation – soil scarification after harvesting to improve forest regeneration – has a negative effect on the soil carbon storage: it creates soil disturbance that is known to change the microclimate and stimulate the decomposition of litter, leading to increased CO_2 fluxes from the soil (Vanhala et al. 2013). Scarification may also cause higher leaching of nutrients to surface waters or groundwater (Rappe-George et al. 2017). The reduced availability of carbon and nutrients in the forest soil may have a negative influence in its future productivity in the long term (Vanhala et al. 2013).

Biological and environmental values in forests are taken into account through forest legislation and the PEFC and FSC certification systems that promote sustainable forestry. In practice this means for instance that living and dead retention trees are spared at the harvesting site. As a consequence of the practice of retention trees, together with the aging of the growing stock and increase in the area of set-asides, the amount of dead wood in Norwegian production forests has been increasing during the last decades (Dalen 2017). Storaunet & Rolstad (2015) give an average of 10.6 m^3 ha^{-1} of dead wood for productive forest land. Retention trees and increasing volumes of dead wood in production forests have certainly positive effects on the forest carbon cycle and lead to higher carbon storage. The carbon storage of the living stand, dead wood and soil of production forests is, however, inevitably still considerably lower than that of natural forests.

2.1.3.2 Increasing use of forest biomass

There is an increasing interest in the use of wood products and wood-based fuels, motivated by the substitution effect (Leskinen et al. 2018), i.e. reduced GHG emissions due to replacement of fossil-based products and fuels. This seems to result in reduced net carbon emissions in the long term (Berndes et al. 2016, Taeroe et al. 2017, Iordan et al. 2018), although e.g. Taeroe et al. (2017) emphasize the many uncertainties related to the modelling assumptions. Collecting logging residues and tree stumps from the forest after harvesting, for bioenergy, is a relatively new practice that aims to replace fossil fuels with forest-based energy that is considered renewable. The use of forest bioenergy as a climate mitigation tool has, however, been criticised for, at least initially, exacerbating rather than mitigating climate change (Norton et al. 2019). Intensive harvest of bioenergy has been shown to lead to losses of the soil organic carbon storage (Achat et al. 2015, Repo et al. 2015), especially if also the stumps are harvested. Removal of stumps does not only remove the carbon stored in stumps, but it also causes soil disturbance that increases the rate of carbon fluxes from the soil to the atmosphere (Vanhala et al. 2013). Forest bioenergy may therefore decrease or even neutralise the forest soil carbon sink. Also e.g. Soimakallio et al. (2018) highlight a significant trade-off between emission reduction through fossil fuel substitution with wood-based products and reduction in the forest carbon sink. Overall, increased use of wood products and wood-based fuels creates a carbon debt in the forest that is not compensated for if the life time of the wood products or fuels is shorter than the time it takes for the new forest to recreate its carbon stock (Seppälä et al. 2019). Persvingelen (2019) estimated the carbon payback time of increased harvest of stems and residue for bioenergy in Norway to be 89–362 years, depending on the amount of greenhouse gas emissions from fossil fuels that is estimated to be avoided by replacing fossil fuels with bioenergy from harvested wood. Therefore, even if wood-based products and fuels contribute to climate change mitigation in long time frames, they may not do so in short time frames (Repo et al. 2015, Taeroe et al. 2017) which are important because of the urgency of preserving and increasing carbon stocks and sequestration (IPCC 2018, 2019).

2.1.3.3 Nitrogen fertilisation

Nitrogen (N) is the main limiting nutrient in boreal forest ecosystems, and therefore commonly applied as a fertiliser in production forests as a means to promote tree growth and in that sense considered a climate change mitigation tool (Haugland et al. 2014). Airborne anthropogenic pollution increases N deposition in both production forests and set-asides. Increasing input of N seems to slow down decomposition and thus reduce the rate of heterotrophic respiration, probably because of shifts in saprotrophic community composition and potentially also because of an increase in the production of N-polyphenol complexes which inhibit decomposition (Deluca & Boisvenue 2012). This leads to an increase in the soil carbon storage, while simultaneously the above-ground production and litter input increases (Olsson et al. 2005). Nitrogen addition may, however, have negative effects on soil processes and lead to depletion of base cations and acidification of soil (Aarrestad et al. 2013) and consequently availability of nutrients in the soil, which may in turn have a negative effect on plant growth (Van Sundert et al. 2018). Nitrogen addition has also several strong negative effects on soil communities and their functions. For instance, Zhang et al. (2018) showed in a global meta-analysis that nitrogen addition reduces total microbial biomass, bacterial biomass, fungal biomass, biomass carbon and microbial respiration, and the effects increased with nitrogen application rate and duration. Nitrogen addition also leads to lower species richness and changing community composition in plants (Aarrestad et al. 2013, Midolo et al. 2019), with potential effects on carbon and nutrient cycles (Lange et al. 2015), e.g. through changed litter quality, replacement of perennials by annuals, and changing root:shoot ratios (Zeng et al. 2010). The long-term consequences of the changes in soil biota for the ecosystem processes, functions and condition, are poorly understood (Bardgett & van der Putten 2014). In addition to changing CO_2 fluxes, the fluxes of other GHG such as N_2O and CH_4, may change as a consequence of increasing use or deposition of nitrogen (Brumme & Beese 1992, Du et al. 2019). Pukkala (2017) suggests using more frequent but lower amounts of fertiliser application to reduce the negative effects of fertiliser while keeping the positive effects (improved tree growth). This may reduce N_2O production, but the cumulative effects of fertiliser application may still lead to changed biotic communities and their functions.

2.1.4 Potential effects of climate change on the forest carbon cycle, storage and sequestration

Warming climate and extended growing season are expected to increase tree growth, potentially leading to an increase in carbon sequestration in living trees by 75%, from 4 to 7 Tg C annually within 100 years, assuming a 2°C warming (Astrup et al. 2010). Atmospheric CO_2 fertilisation may contribute to increased carbon sequestration by living trees (Tagesson et al. 2020). However, a warmer climate may also mean faster tree turnover (reduced tree longevity) and consequently shorter carbon residence time, and therefore lower than expected sequestration despite higher productivity (Büntgen et al. 2019). Increasing tree growth rates and faster turnover of individual trees will lead to higher volumes of dead wood especially in reserves and other set-asides (Claesson et al. 2015). Precipitation is expected to increase in Norway, but this is partly attributable to increased frequency of heavy rain episodes while there may be considerable drought periods that increase the risk of forest fires (Hanssen-Bauer et al. 2017). Periods of heat and drought may make forest trees more vulnerable to pest species and their outbreaks (Jactel et al. 2019). Tree pathogenic fungi may extend their distributions and abundance in Norway (Solheim et al. 2011). These changes in abiotic and biotic stress factors and disturbance regimes may affect carbon sequestration and growing stocks, while potentially increasing the carbon stocks in dead wood and soil at least in the short term.

The carbon storage of dead organic material is vulnerable because of the susceptibility of the decomposition process to temperature and moisture changes (Rinne-Garmston et al. 2019). Microbial activity is expected to increase with temperature, which may mean increased rates of decomposition of dead wood and soil organic matter. The rate of heterotrophic respiration is consequently expected to increase, leading to higher CO_2 fluxes from the soil and dead wood to the atmosphere (Ågren & Hyvönen 2003), and potentially slower accumulation of the soil carbon storage. The future rates of decomposition and respiration are, however, difficult to predict as they depend on the community composition and species richness of the saprotrophic communities (van der Wal et al. 2015), both affected simultaneously by the warming climate, forestry, pollution and changing land use (Bradford et al. 2014, Nordén et al. 2013, 2018, Mosier et al. 2017). Related to this, the future changes in the storages and fluxes of other GHG, such as CH_4, are equally challenging to predict.

2.2 Alpine & cryospheric ecosystems

Norway is a mountainous high latitude country, meaning that much of the landscape is above the treeline. The climatic treeline, and subsequent shrub-line is highest in inland southern Norway, and lowest in the north where habitats are considered 'sub-Arctic' (CAFF 2001) (**Figure 6**). Actual treeline may be modified by human induced activity e.g. grazing by farm animals. These 'alpine' zones make up 33% of Norway's mainland area (Austrheim et al. 2010, Bryn et al. 2018), with ~1% of the mainland covered in permanent ice (ice sheets and glaciers; Norwegian Water Resources and Energy Directorate 2019). Despite the cold climates, these are not lifeless habitats and even glaciers will be contributing to biological carbon cycling. For this section, we shall examine these habitats above the treeline as discrete bioclimatic zones: low and mid alpine (typically 1000–1450 m a.s.l), and high alpine (> 1450 m), after Austrheim et al. (2015), using vegetation types as proxies (shrub, heath, meadow and nival). The nival habitat in the high alpine zone is characterised by permanent snow and ice of which the latter shall be discussed within the context of the 'cryosphere'. The cryosphere with classifications of: permanent snow and ice (grouped as glaciers) and permafrost (**Figure 6**). Overall, biodiversity in the mountains is generally good with a Nature Index value of 0.62 in 2014 (Framstad 2015). This is likely due to the lower levels of human interference, and does not account for the future threat of a changing climate, to which these habitats are acutely vulnerable.

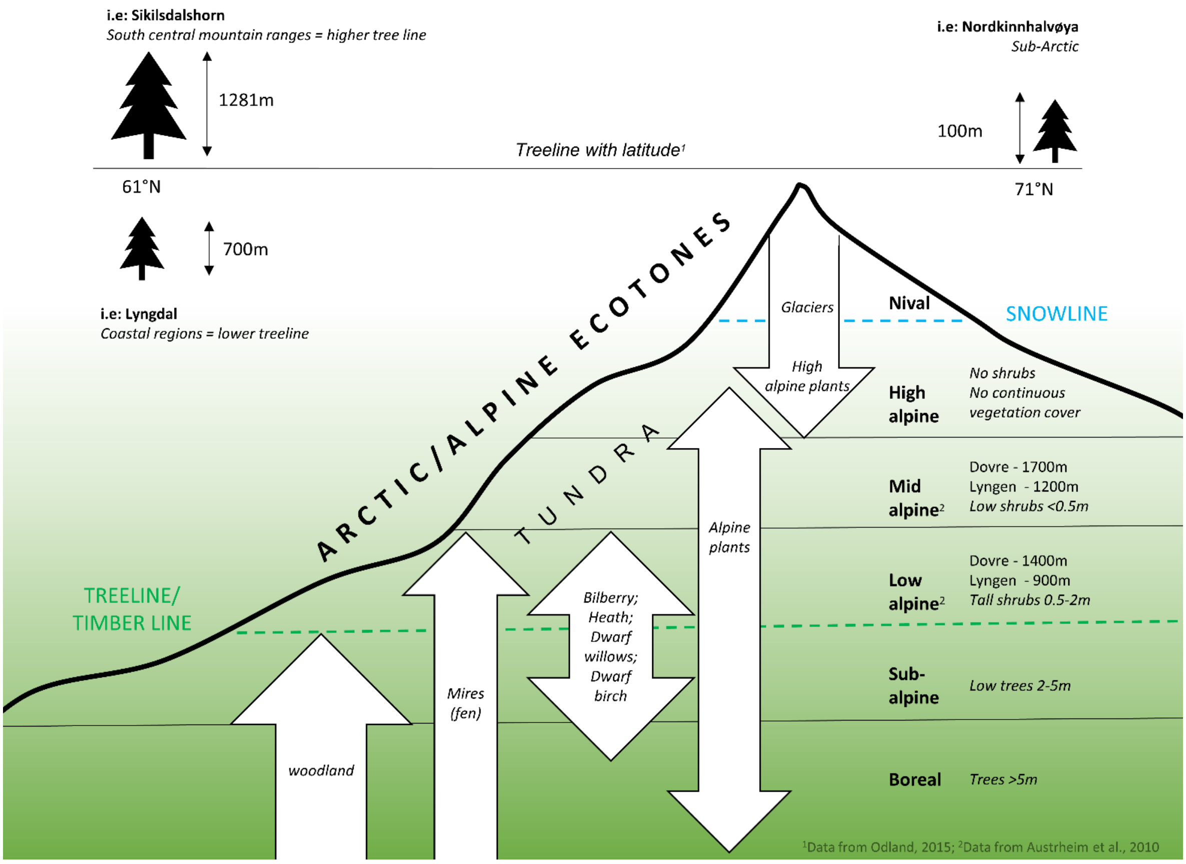

Figure 6. *Norway above the treeline: Alpine and arctic zones shown with approximate boundaries of division (metres above sea level). Treeline with latitude data taken from Odland (2015); treeline with altitude taken from Austrheim et al. (2015).*

One of the leading causes of polar and alpine climate feedbacks, is the change in surface albedo as the result of decreased snow and ice cover, and increasing vegetation (Hall 2020). Snow and ice reflects solar radiation and have a high albedo, whilst bare ground and vegetation absorb it and have a low albedo. As temperatures warm, and the cryosphere melts, the surface will heat faster, further accelerating melt and/or vegetation growth. Different vegetation types have different albedo properties, with models suggesting that the displacement of tundra by forest will decrease albedo, enhancing the rate of climate warming (Zhang et al. 2013).

2.2.1 Carbon cycle in alpine ecosystems

Globally, alpine and particularly tundra ecosystems, are thought to have one of the highest terrestrial soil organic carbon (SOC) values on the planet of ca. 22 kg C m^{-2} (Post et al. 1982), largely due to the storage capacity of permafrost mires. In calculating the area of Norway that is made up of alpine zones, we have taken our lead from area cover estimates after Bryn et al. (2018), with dwarf shrub heath (39,000 km^2); alpine heath (38,000 km^2), alpine meadows (8300 km^2) and then the snow bed, or nival zone (19,521 km^2). We substitute known glacial coverage from the nival calculations to produce a nival zone result only applicable to ice-free land (which then becomes negligible) (**Table 2**).

Table 2. *Approximate surface area of Norwegian alpine areas and ecotones. *Permafrost underlies much of the alpine regions, particularly those of high alpine and nival zones (calculated by Gisnås et al. 2016 excl. glaciers). Glaciers are considered a more dominant influence in high-nival zones than low-mid, thus their surface area is subtracted from high-nival.*

Ecotone/habitat	Approximate surface area of Norway (km^2)	Source
Total Alpine	104,000	Bryn et al. (2018)
Low & mid alpine zones (shrub & heathlands)	77,000	Bryn et al. (2018)
High alpine & nival zone (excl. glaciers) (meadows and nival)	27,000	Bryn et al. (2018)
Glacier	2,700	Norwegian Water Resources and Energy Directorate (2019)
Permafrost (excl. glaciers)*	13,000	Gisnås et al. (2016)

Within the zones above the treeline, SOC does not follow a strictly linear trend with altitude/latitude. For example, studies from the Tibetan Plateau find that SOC in low-alpine zones were relatively low (2.6 kg C m^{-2}), but increased with altitude to peak in the high alpine zone (13.7 kg C m^{-2}), before decreasing with altitude in the nival zone down to 1 kg C m^{-2} (Ohtsuka et al. 2008). Similar patterns of carbon storage are also found globally, with alpine meadows tending to have larger carbon stocks than heath and shrub dominated habitats, despite higher primary productivity in the lower alpine areas (Körner 2003). However, few other studies explicitly examine the variation between low, mid, high, nival carbon budgets. Carbon budgets of high alpine and nival zone ecosystems are little studied, with any examinations at these altitudes focussing more on nutrient availability over SOC and carbon flux (Ohtsuka et al. 2008). Due to the thin soils, low temperature and low vegetation coverage within the ice-free nival zones, we consider the value of 1 kg C m^{-2} from Ohtsuka et al. (2008) to be a reasonable assumption to make for comparable Norwegian land areas (**Table 3**). However, considering the potential of some nival zones to have vegetated snow beds, those areas with vascular vegetation are more likely to behave more in line with alpine meadows than vegetation free nival areas. Therefore, these areas are measured as alpine meadows in nival calculations, using carbon flux and storage data for alpine meadows after Sørensen et al. (2017) and the area for 'sedge and grass snow bed' (7525 km^2) after Bryn et al. (2018).

For our calculations of vegetated alpine carbon budgets, we consider the Norwegian alpine study by Sørensen et al. (2017) to be the most pertinent, not only geographically, but because it is one of few studies that includes detailed values for alpine vegetation primary productivity (PP), respiration (R), as well as above- and below-ground carbon storage (for example, only SOC considered in Ohtsuka et al. 2008). However, based on global literature, we find that the communities of 'shrub', 'heath' and 'meadow' examined in the Sørensen et al. (2017) paper most directly reflect values representative of potential carbon in 'low', 'mid' and 'high' alpine zones, respectively, considering the aforementioned non-linear gradient of SOC with altitudes, and so could be used as a proxy for these ecotones (**Table 3** and **Figure 6**). Calculations represent only available growing days (140) after Wagner et al. (2009). Variability between outputs calculated from Ohtsuka et al. (2008) and the aforementioned Norwegian study is discussed in **Table 6**.

***Table 3**. Approximate levels of annual gross primary production (PP), annual gross respiration (R), annual net carbon flux (NET), carbon stored in alpine ecosystems.*

	PP (Gg C yr^{-1})	R (Gg C yr^{-1})	NET (Gg C yr^{-1})	STORAGE (Gg C)
SHRUB	5,500	3,100	2,400	256,000
HEATH	3,000	650	2,350	351,000
MEADOW	925	500	425	101,000
NIVAL	>0.8	>0.6	>0.2	0.01 – 90,000

2.2.2 Carbon cycle in cryospheric ecosystems

Only in recent decades have scientists begun to realise the carbon potential in the cryosphere, as climate change speeds the melt and thaw of the planets long-term ice and the implications of the losses are becoming known. As such, estimations of the carbon content of glaciers and permafrost are still in their early stages and the cascading impacts of warming still poorly understood (Turetsky et al. 2019).

2.2.2.1 Glaciers

Glaciers cover a small percentage of Norwegian alpine areas, and currently total ca. 2700 km^2, with much of the ice contained within the ice-caps of Jostedalsbreen, Hardangerjøkulen and Folgefonna in the south and in Saltfjellet-Svartisen in the north (Norwegian Water Resources and Energy Directorate 2019, see https://gis3.nve.no/link/?link=breatlas, for maps). The carbon budgets of glaciers can be estimated using global average data for the particulate organic (POC) and dissolved organic carbon (DOC) content of mountain glaciers (MGL) calculated by Hood et al. (2015). Norway's mainland is just 0.02% of global glacial surface area, thus the values for global POC and DOC storage and runoff can be applied to give approximate values of the current carbon storage and contribution of Norwegian glaciers (**Table 4**).

***Table 4.** Calculations of approximate stored carbon, and carbon export through runoff of Norwegian mainland glaciers (calculations after Hood et al. 2015).*

	Global avg MGL stored C (Pg)	Avg MGL C runoff (Tg C yr^{-1})	Stored C Norway (Gg C)	C Runoff Norway (Gg C yr^{-1})
POC	0.06	0.70	11	0.14
DOC	0.07	0.58	14	0.12

Within calculations for stored DOC, we also need to consider that glaciers are biologically active and capable of continually sequestering carbon through primary production. After Anesio et al. (2009), we can conservatively estimate the carbon flux of Norway's glaciers based on cryoconite activity on the glacial surface: typically, only 2% of a glacier's surface is covered with biologically active cryoconite (based on European and Arctic studies), which amounts to an area covering 53.84 km^2 of Norwegian glaciers. This would equate to a carbon sequestration through primary productivity of 0.19 Gg C yr^{-1} and annual respiration of 0.05 Gg C yr^{-1}, resulting in a net carbon gain of 0.13 Gg C yr^{-1} (**Table 4**).

2.2.2.2 Permafrost

It is considered that in light of climate change, permafrost is no longer a carbon sink, but rather a carbon stock that has become a carbon source worldwide. As permafrost thaws out and the active layer deepens, respiration increases with the decomposition of previously frozen organic matter (OM). This varies depending on: the length of time into the thaw process, with initial thaws releasing more carbon; the depth of the permafrost layer, with higher 'active' layers respiring more than deeper layers; and the organic and microbial content of the permafrost (e.g. Monteux et al. 2018).

The permafrost regions of Norway have been modelled and mapped through NORPERM and associated publications and researchers (with open source access available through http://geo.ngu.no/kart/permafrost/), which combined with estimates of the carbon stored at various depths, can provide a broad value for carbon stored in Norwegian mainland permafrost. Hugelius et al. (2014) estimate that Scandinavian permafrost at a depth of 0–3 m contains a median of 57.5 kg C m^{-2}. This would equate to approximately 750 Tg of carbon being stored in the top 3 m of Norwegian mainland permafrost. A conservative median value of Norwegian permafrost depth is 42.5 m, however, the majority of SOC is within the top few metres of permafrost, which is also the most susceptible to warming, so the level at which this value would increase is debatable and outside the scope of this study.

Respiration rates associated with permafrost thaw are being considered throughout the Arctic regions, given that their thaw and release of CO_2 and especially CH_4 could have a significant positive feedback into climate change processes. We find that the most applicable studies to Norway, from nearby Abisko in Sweden, estimate a permafrost landscape respiration rate of 3.3 g C m^{-2} yr^{-1}, measured in July. Extrapolated across the year and over the permafrost area of Norway, this equates to respiration of nearly 16 Tg C yr^{-1} (Hicks et al. 2015). Care needs to be taken interpreting this result as winter respiration rates will be lower, although year-round rates will accelerate under climate change.

2.2.3 Prevailing management practices and effect on carbon balance

Much of Norwegian policy regarding protected areas, has focussed on the mountains and developing protections for native species, such as wild reindeer reserves (Kaltenborn 2014). Approximately 30% of the country's alpine area fall under national park protection.

2.2.3.1 Tourism and recreation

It is a national aim to "increase the sustainable use of mountain areas for tourism as a basis for local development" (Skjeggedal et al. 2015). Arguably the largest risk to the alpine zones is the growth in holiday cabins/second homes, with 25% more construction on the timber-alpine boundary from 1985–2005 as a result of this form of development and recreation (Kosmo et al. 2007). More than half of the Norwegian population own, or have access to a private second home, spending 10–30% of the year at them (Rye et al. 2011). The development of the treeline and low alpine regions of Norway has already been identified as potentially negatively impacting the alpine environment by the Storting in Document 3:11 from the Office of the Auditor General, with the same report highlighting a lack of national overview of the consequences of such land management (Kosmo et al. 2007).

Impacts on the actual carbon stocks as a result of the physical disturbance is likely to turn the low alpine area occupied by second homes from a carbon sink to a carbon source. For example, the average size of a Norwegian cabin in 2003 was 79 m^2, and the number of cabins is estimated to be in the region of 460,000 (Ericsson & Overvåg 2009). Using our calculations we estimate that recreational cabins displace approximately 574 Gg of carbon stored from low-alpine soils, with a further loss through primary production as a result of vegetation removal of at least 80 Gg C yr^{-1}. This is caveated with the assumption that all second home cabins are built in this alpine zone.

2.2.3.2 Mountain agriculture and grazing

Large herbivores can alter carbon stocks by changing the plant communities, for example, by reducing the number of trees and woody shrubs, which as well as affecting carbon stocks, will alter local albedo. Low-mid alpine zones are the most affected by livestock grazing: For example, even at low density (e.g. 25 sheep km^{-2}), it has been found that grazing significantly reduces the recruitment of subalpine trees, and where grazing is excluded, the birch and shrub line can extend in altitude (Mysterud & Austrheim 2008). As a result, where sheep are excluded, carbon stocks *above ground* increase (Speed et al. 2014). It is estimated that cessation of sheep grazing alone would increase the carbon storage in Norway's alpine ecosystems by 4.2 Tg C, largely as a result of the expansion of the tree-line (Speed et al. 2014). However, colonisation of mid-alpine/tundra heath by trees and shrubs may have further effects on the carbon cycling as it could decrease the amount of carbon that is stored *below ground* in alpine meadows for example (Sørensen et al. 2017). Thus grazing is one confounding variable on 'shrubification', which is examined in the following section **2.2.4.1**.

2.2.4 Potential effects of climate change on carbon cycle in alpine and cryospheric ecosystems

With a third of Norway's mainland land mass above the treeline, and at the same time, one of the habitats most vulnerable to climate change, alpine habitats will be significant influencers on the country's carbon budget.

2.2.4.1 Shrubification and permafrost feedbacks

One of the most significant impacts of climate change on alpine and arctic carbon budgets, is the 'shrubification' of above-ground vegetation. Broadly, the warming of higher altitudes and latitudes expands the range potential of shrubs by extending their growing season, facilitating their move from low alpine into mid-alpine zones, and upwards in latitude. Overall, it has been found that by 2100 tundra ecotones will increase primary production by 244 g C m^{-2}, and respiration will increase by 139 g C m^{-2}, resulting in the tundra becoming a larger net carbon sink (106 g C m^{-2}) (Mekonnen et al. 2018). However, some models forecast that the permafrost carbon sink will not persist in tundra ecotones beyond 2100 due to warmer autumns, and may in fact completely offset the carbon gains of increased woody plants during spring and summer (Piao et al. 2008, IPCC SROCC 2019). Shrubification is expected to expand by 24–52% of tundra by 2050 (IPCC SROCC 2019), the median area value of which would mean an increase in shrubs found in the low-mid alpine range, and a decrease of mid-high alpine habitats. As a result of greater ecosystem primary production, very crude estimates would suggest a rise in net carbon sequestration of 100 Tg C between 2020 and 2050 in Norway. This would be likely offset by the slow release of the 700 Tg C estimated to be stored in Norwegian permafrost. It is also worth considering that encroachment of shrubs may reduce the amount of SOC belowground, as alpine meadows in particular have significantly larger below-ground carbon stores than shrub habitats, and that this could also offset gains made in above-ground carbon accumulation as a result of greater woody mass and PP as a result of longer, and warmer growing seasons (e.g. Sørensen et al. 2017). Furthermore, a change in surface albedo as a result of shrubification is another complicating factor, with shrubs and treeline expansions likely to decrease local surface albedo, and increase regional warming (e.g. Miller & Smith 2012).

2.2.4.2 Glaciers

Since 2000, many of Norway's glaciers have shrunk considerably. The increase of export, decrease in primary production active surface area, and diminishing storage capacity as the volume of the ice decreases, will mean a continuing loss of carbon and carbon potential from glaciers over time. Whilst the loss of glaciers from Norway's mainland landscape will be culturally detrimental, carbon stocks here are comparatively low compared to that of the tundra and associated permafrost. However, rising downstream effects including increased landslides and flooding, will release carbon, whilst the succession of vegetation into newly ice-free areas may result in increased carbon fixation and storage in the long-term, as with shrubification. However, the loss of ice cover will also release sub-glacial permafrost to the effects of a warming planet.

2.3 Open lowlands

The cultural landscapes formed by agriculture/croplands and grassland areas in Norway, are all classified as open lowland areas below the treeline, including natural and semi-natural vegetation habitats. These habitats are often formed via historic disturbance activities such as grazing and farming, or clearance of forest vegetation (Framstad et al. 2015). Habitats considered “open lowland” are incompletely mapped (Venter et al. 2019), but estimates of 2.1% for “non-forested dry land below the tree-line” - which includes heathlands -, and 3.8% for cultivated land and pastures are given in Bryn et al. (2018). Other national inventories consider other grasslands to amount to 1% of land cover (Statistics Norway 2019). Open lowland habitats have a very high potential for biodiversity conservation, including old meadows as habitats for pollinators, pastures for livestock, and recreation opportunities for people. These habitats occur across Norway, and their composition varies along gradients of temperature, soil nutrient content and humidity, as well as current management and its history (Framstad 2015). Open lowlands, i.e. the semi-natural ecosystems, have the nature index value 0.47, with the most important factor affecting the value being land-use change (Framstad 2015).

2.3.1 Carbon cycle in grasslands, croplands and heathlands

Grasslands and heathlands are systems of treeless vegetation where ecological disturbances shaped by grazing, browsing, mowing and/or fire are key ecological factors. Despite that a portion of the above-ground biomass is removed by herbivores or by management (mowing/hay harvest), most of the atmospheric CO_2 captured in these systems by plant photosynthesis is stored in the soil as soil organic matter. Soil organic matter contains ca. 50% of soil organic carbon, it is primarily determined by the root biomass of plants (Ontl & Schulte 2012), and is particularly high in grasslands, because grasses have most of their biomass (50–80%) in long, thin and short-lived roots that build large amounts of soil organic matter.

A survey of United Kingdom semi-natural grasslands found that acidic grasslands in particular had the highest carbon stocks of any UK broad habitats and were capable of sequestering carbon at a rate higher than that of slow growing forests in the country (Bullock et al. 2018). Similar trends are found in Falloon et al. (1998), where plant inputs to soil organic carbon were higher in grasslands than in forest (433 vs. 381 $g\,C\,m^{-2}\,yr^{-1}$). In some assessments, soil organic carbon in grasslands may be underestimated since grasslands can store carbon at greater depth than forest, and carbon estimates are often limited to the soil surface layers. Soil organic carbon (27 cm depth) in old grasslands (> 100 yr) managed with mowing and different fertilization treatments, ranged between 7 and 13 $kg\,C\,m^{-2}$ in Hopkins et al. (2009). These values are in the range for temperate grasslands in Europe (30 cm depth, 9–10 $kg\,C\,m^{-2}$) (Eaton et al. 2008). National inventories for Norwegian greenhouse gas emissions (Norwegian Environment Agency 2019a) calculate potential grassland storage to be 9.8 $kg\,C\,m^{-2}$, which is in line with others, and shall be the unit we use in this report.

Carbon stocks and cycling in cropland ecosystems is highly dependent on both soil type (organic or mineral) and the prevailing management practices (Maljanen et al. 2010, Grønlund et al. 2008). The majority of all croplands in Norway are fully cultivated grasslands, with agriculturally managed croplands accounting for 65% of all agricultural land in Norway: the remainder used for cereal, oil seed and root crops (Rognstad & Steinset 2012). Because of the low amount, and transient nature of live biomass, carbon fluxes from cultivated ecosystems are difficult to establish at a national level. However, carbon emissions from the soils are estimated to be between 300–860 $g\,C\,m^{-2}\,yr^{-1}$, based on boreal inland organic grassland soils (IPCC Wetlands Supplement 2014). Grønlund et al. (2008), estimate average soil carbon densities of 15–56 $kg\,C\,m^{-2}$ giving an approximate soil carbon store of 390 Tg C between mineral and organic agricultural soils in Norway. In contrast, the national inventory report for greenhouse gas emissions reports values of 8.3 $kg\,C\,m^{-2}$, which amounts to 91 Tg C in total (Norwegian Environment Agency 2019a).

Carbon stores in Norwegian heathlands is little explored in lowland areas but has been studied in alpine habitats which suggests that in alpine regions heathlands have large carbon soil stores (e.g. Strimbeck et al. 2019, Sørensen et al. 2017; see also early work from FunCaB projects at the University of Bergen: https://app.cristin.no/results/show.jsf?id=1750872). Lowland heaths classed as 'coastal', '*Calluna* spp. (heather, Norwegian *lyng*)', and 'damp' heath by Bryn et al. (2018) cover approximately 6800 km^2 in Norway (2% of total land cover). Compared to alpine heaths, these lowland heaths are likely drier and thus soils thinner, culminating in a lower carbon store than higher altitude/latitude heathlands. The large portion of the carbon stock in heathland is in the soil, e.g. rough estimates from England indicated an average of 8.8 kg C m^{-2} in soil and 200 g C m^{-2} in the vegetation (Alonso et al. 2012). Milne and Brown (1997) suggested that podsols which are common in lowland heaths in the United Kingdom, contain approximately 17.5–21.1 kg C m^{-2}, and sandy heathland soils found on the coasts, contain approximately 9.3 kg C m^{-2}. If applying these crudely to Norway we estimate that costal heathland contain 24.4 Tg C, and the remaining *Calluna* and damp heath contain 79.8 Tg C. These three habitats make up the bulk of all non-agricultural open lowland areas.

2.3.2 Prevailing management practices and effects on carbon balance, storage and sequestration in open lowlands

The area of these habitats has been strongly reduced in the 1900's due to the abandonment of traditional and extensive management practices, in favour of intensive forms of food production. The main driver of habitat change of open lowland habitats has been the large-scale cessation of management practices, e.g. grazing, mowing and/or fire, that hinder the development of the vegetation into shrubland and forest (NOU 2013). Encroachment is also the main driver of change for coastal heathland in Norway, in addition to infrastructure development, conversion to arable land and afforestation with exotic tree species (Norderhaug & Johansen 2011).

Encroachment of grassland and heathland with shrubs and trees is not necessarily associated with gains in total carbon stocks. Despite the larger accumulation of above-ground biomass in shrubs and trees (Speed et al. 2014), there is evidence that carbon stocks in shrubland and forest soils are lower than in grassland soils, and that total ecosystem carbon stocks could potentially be larger in meadows and heathlands compared to shrublands in Norwegian mountains (Sørensen et al. 2017). International studies also show higher soil carbon stocks in grasslands compared to woodland and wooded meadows (Upson et al. 2016).

Grazing intensity is likely to change soil carbon stocks, because it often changes the quality of the plant material, and may affect soil aeration, both factors influencing SOM decomposition rates. When grazing pressure is too high, trampling by livestock can result in low vegetation cover and soil erosion, leading to carbon stock loss. Salt meadows are particularly vulnerable to soil erosion by trampling (Evju et al. 2015), due to the exposure to the action of tides and waves. Evju et al. (2015) report a large proportion (41%) of the salt meadows in their study showing signs of damage by trampling. However, if managed with adequate grazing pressure, grassland and heathland are ecosystems that provide the strongest protection of SOC, due to the continuous vegetation cover, and that the soil is not disturbed.

Further, cultivation and drainage of open lowland habitats have increased the pressure to apply mineral fertilizers and drainage works. This also applies to salt meadows, where all these practices have been applied (Evju et al. 2015). In the material analysed in Evju (2015), 14% of the sites in the study showed signs of eutrophication. Cultivation is considered a major intervention leading to the loss of SOC in time. Other kind of physical destruction by the construction of infrastructure are also common in salt meadows.

2.3.2.1 Maintainance by grazing, mowing and/or fire in the lowlands

In Norway, non-agricultural open lowlands encompass a group of semi-natural habitats dominated by grassland and heathland vegetation which have been shaped by long-term management with grazing, mowing and in some cases, fire, and which require management to maintain their characteristic qualities.

Coastal heathland is a vegetation type generally dominated by heather (*Calluna* spp.), although other low shrubs can be important, that used to cover extensive areas along the Atlantic coast of Europe, from Portugal to northern Norway, the majority of which has derived from former woodlands (Måren & Nilsen 2008). Heathlands used to be managed for grazing and harvesting of forage, especially to provide fodder for the winter, and small areas of heath were burned at intervals of 25 to 40 years in western Norway to keep the vegetation young and rich in nutrients (Nilsen 2004).

It is estimated that 80% of the heathland in Europe has been lost, and that in Norway, only 10% of the original area remained in 1990 (Framstad 2015). Abandoned heathlands are characterized by old heather (*Calluna vulgaris*) shrubs and encroachment by shrubs and trees. A recent study shows that one third of the studied coastal heathlands were in a late encroachment stadium with trees (Johansen et al. 2015).

Species-rich semi-natural grasslands within open lowlands include hay-meadows which are managed mainly with mowing, and semi-natural grasslands used as grazing land. These habitats were main sources of fodder for livestock before the introduction of inorganic fertilizers for fodder production and the use of other feed sources for livestock in Norway. The area of these habitats is not well known, but it is estimated that there remains approximately 1% of the hay meadows in Europe, but Norway is unique in terms of the richness of managed hay meadows in the past and at present (Norderhaug & Svalheim 2009). There are 2500 registered habitats used for hay harvest – which includes a broader range of habitats used to collect winter fodder in Norway – of which 600 have received payments to be managed (Norwegian Environmental Agency, Environmental Status[2]). The total area of habitats for hay production in Norway has been estimated between 500 and 2000 ha (Norderhaug & Svalheim 2009). These habitats must be mowed regularly to hinder encroachment by shrubs and trees.

Salt meadows are open grass- and herb-dominated vegetation types in the littoral zone where a short grassland vegetation is maintained by grazing. The area estimated in Norway in 2012 based on field surveys was 204 km^2 (Evju et al. 2015). The majority of the areas (63%) are located within the middle boreal eco-zone, but salt meadows occur in most ecological regions in Norway ranging from boreonemoral to the north-boreal zones. Many salt meadows have been used for grazing for a long time, and livestock grazing and/or mowing has been necessary to maintain the low grassy vegetation. When grazing and mowing cease, salt meadows develop into reed and/or black alder dominated vegetation, a process which changes the distribution and amount of carbon stocks.

2.3.3 Potential effects of climate change on carbon cycle in open lowlands

Grasslands which build large soil carbon stocks over living biomass are more resilient as carbon sinks, especially in fire-prone and drier areas (Dass et al. 2018). Measurements of long-term soil organic carbon stocks in grasslands (30–40 years) show no detectable changes of stocks despite significant warming of the soil and air, indicating that projected climate change impacts on soil carbon stocks may be lower than earlier assumed (Hopkins et al. 2009). However, heathlands may be affected by increased mean temperatures which increase the risk of drought in wetter heathland sites, reducing potential carbon stocks and altering community composition and in drier heaths will increase the risk of wildfires (Alonso et al. 2012). In heathlands dominated by

[2] https://www.environment.no/no/Tema/Naturmangfold/Kulturlandskap/

dwarf shrubs such as *Calluna* species, it is possible that longer growing seasons will benefit grasses, potentially resulting in a decrease in above-ground carbon stocks towards a grassland system with a tendency for larger below-ground stores (Wessel et al. 2004).

2.4 Wetlands

Wetlands are areas characterised by permanent or periodic water saturation that support emergent plants adapted to a life in wet conditions, and can include mires and peatlands, swamps, floodplains, marshes and springs. Approximately 10% of the mainland of Norway is covered by wetlands. Wetlands have a wide range of unique species, often specialized to wetland habitats, including amphibians, bryophytes, vascular plants, and birds. The Nature Index for mires and wetlands varies greatly throughout Norway but has been on a downward trend since 1990, and was last estimated to a national average of a moderate 0.55 in 2014 (Framstad 2015).

2.4.1 Carbon cycle in wetlands

Wetlands can be highly productive ecosystems and hold the highest density of carbon in the soil of terrestrial ecosystems (Villa & Bernal 2018). That makes them efficient and cost-effective options to sequester atmospheric carbon and important in long term storage of carbon (Were et al. 2019). Wetlands are the transition between land and water. This makes the carbon cycle in this ecosystem complex and it is difficult to know exactly where carbon is lost in the system.

In wetlands, atmospheric CO_2 is taken up by photosynthesis and released by respiration of plants, but most of the carbon assimilated goes into the soil as soil organic carbon (SOC) or soil organic matter (SOM). Carbon is sequestered in the system as SOM which accumulates over time. Carbon goes in and out of wetlands systems also as POC, PIC, DOC, and DIC from upstream ecosystems and then exit to downstream ecosystems. In addition, carbon is released as methane (CH_4) through methanogenesis, that is the anaerobic metabolism of organisms that grow in habitats with no access to oxygen. Globally, as much as 24% of CH_4 emissions to the atmosphere is from wetlands (Villa & Bernal 2018). Because of CH_4 emissions, it may vary to what extent wetland sites are net carbon sinks, but the ecosystems are in any case holding a large carbon pool below ground (Villa & Bernal 2018).

The water table is the most important ecological factor affecting the carbon cycle (Minkkinen et al. 2002, Evans et al. 2016). Lowering the water table allows oxygen to enter the system and decomposition of organic material begins. The release of methane ceases, but the release of carbon through CO_2 turns drained wetlands into carbon sources. Evans et al. (2016) showed that peatland sites are roughly GHG neutral when the mean water table level is in the range of 0 to 10 cm below the surface, but that peatlands become large net emission sources when sites are drained (up to 3 kg CO_2-e m^{-2} yr^{-1} at the most deeply drained sites) or when inundated (over 1 kg CO_2-e m^{-2} yr^{-1} at the most waterlogged site).

Peatlands are the wetland type with highest coverage in Norway (ca. 9% land cover; Magnussen et al. 2018). Peatlands are peat-forming ecosystems, and are usually defined as having a peat layer of 30 cm or deeper (Moen et al. 2011). A peatland where peat is actively accumulating through its vegetation and waterlogged conditions is generally called a mire. The peat mosses (*Sphagnum, torvmoser*) are important peat forming species in mires. Peat mosses are ecosystem engineers and form their own habitats, as they have a high capacity of holding water, acidify their environment and build peat. Peat (organic soil) is partially decomposed plant material that accumulates because decomposition is extremely slow as the water table is high and oxygen levels low (**Figure 7**). At higher latitudes, for example the boreal region, the low temperatures further reduce the rate of decomposition. Thus, carbon is taken up from the atmosphere and is stored in peat, and as much as 50% of peat consists of carbon.

Peatlands grow and develop over time (**Figure 7**). Most peatlands have developed over the last 10,000 years, since the last glacial maximum. It takes about 1000 years to build one metre of peat (Moen et al. 2011). The deepest mires in Norway are up to 10 metres. Accumulation and degradation are slow processes that are dependent on climate and environmental factors, but anthropogenic disturbance, such as ditching, interrupt the water balance in peatlands instantly.

The time frame which intact wetlands can store carbon is still uncertain, but the ecosystem can likely store carbon up to several hundreds of years to millennia (Were et al. 2019). It is estimated that as much as 500 Pg carbon are tied up in peat mosses globally, dead and alive (Yu et al. 2010).

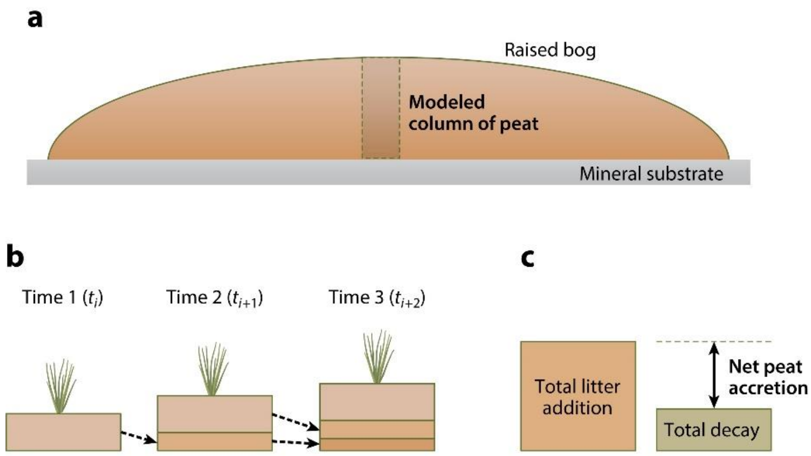

Figure 7. *a) Cross section of a mire (raised bog) showing the peat column, b) peat accumulation in time, and c) net gain of peat showing that decomposition is much lower than the plant material (litter) that are added. Illustration from Page & Baird (2016).*

2.4.2 Carbon storage and sequestration in wetlands

Carbon sequestration in wetlands is obtained from the difference in carbon input and output (Were et al. 2019). When production is higher than decomposition, carbon is sequestered in the organic soil (Swindels et al. 2019). There is a huge difference in productivity of wetlands. Villa and Bernal (2018) give an overview of mean carbon sequestration of different wetland types (Ramsar wetland classification system) based on 110 studies. The type freshwater tree-dominated has the highest capacity followed by permanent freshwater marsh, intertidal marsh and intertidal forested (122.6 g C m^{-2} yr^{-1}, 113.2 g C m^{-2} yr^{-1}, 102.7 g C m^{-2} yr^{-1}, respectively), while non-forested peatland only sequester 26.1 g C m^{-2} yr^{-1}. De Wit et al. (2015) estimated a lower net carbon uptake for peatlands in Norway; 19 ± 15 g C m^{-2} yr^{-1}, but this is within the range of other estimates (ranging from 11 to 32 g C m^{-2} yr^{-1}, see references in de Wit et al. 2015, Turunen et al. 2002). De Wit et al. (2015) further calculated the annual change in carbon pool of undisturbed boreal peatland in Norway to be 0.29 ± 0.22 Tg C yr^{-1}, while disturbed boreal peatland has a net loss of carbon (-0.05 ± 0.04 Tg C yr^{-1}).

Very roughly, the peat mosses in Nordic mires are generally considered to have a growth rate of 1–4 cm per year, with only approx. 1 mm of the organic material being added to the lower peat layers annually (Aaby & Tauber 1975). The peat accumulation will also have local and regional variation (van der Linden 2014, Turunen et al. 2002), be dependent on climatic, hydrologic and hydrochemical conditions (Stivrins et al. 2017, Belyea & Clymo 2001) and will vary from year to year (Roulet et al. 2007, Alm et al. 1999).

More important than annual sequestration of carbon, is the huge amount of carbon stored in wetlands (Villa & Bernal 2018). Of the terrestrial ecosystems, wetlands cover the smallest area globally, but hold the largest belowground carbon storage. Boreal peatlands and coastal wetlands (e.g. salt marshes) seem to be the largest carbon sinks (Villa & Bernal 2018).

On a global scale, peatlands cover only 3% of the global landmass, but contain more than 20% of carbon stored in soil. Estimates vary because of uncertainties about depth of peat deposits, but as much as 500 Pg carbon are likely tied up in peat (Yu et al. 2010). The amount of carbon stored at individual sites varies with the size of the area, the depth of the peat layer and the density of carbon. As an example of variation between sites, the carbon stocks in different fen sites in England can range from 137 to 282 kg C m^{-2} (Evans et al. 2016). In Finland, 30% of land (ca. 90,000 km^2) is covered by peatland, and as much as 2/3 of the carbon stored in Finnish ecosystems is in peat (Minkkinen et al. 2002). Turunen (2002) estimated the total carbon pool of undrained Finnish mires to be 2257 Tg. England's peatlands cover 11% of the land area (ca. 14,000 km^2) and are estimated to contain 584 Tg C (Alonso et al. 2012).

Grønlund et al. (2010) reported that 943–1035 Tg C (with high uncertainty in the estimate) are stored in peatlands in Norway, based on an area of 18,800 to 21,700 km^2, and thus 50 kg C m^{-2}. However, the land cover of open mires are estimated to 28,300 km^2, covering 9% of Norway's area (Bryn et al. 2018). The areas covered by peatlands (mires and swamp forest) prior to land-use changes (150–200 years ago) was likely around 44,700 km^2, but today 37,719 km^2 (including 9400 km^2 of swamp forest) are still intact (Joosten et al. 2015). Of this area only 17,341 km^2 is reported through the Norwegian land use statistics (Statistics Norway 2019a). The potential amount of carbon stored in peatlands in Norway is therefore likely a lot higher than reported.

Peatlands in Norway have been affected by land-use change, mostly by drainage (Joosten et al. 2015). Most of the areas have been turned into agricultural lands by ditching, draining the areas to make them suitable for forestry or as grassland and cropland. The carbon release of damaged peatlands has been estimated to be 5.55 Tg CO_2-e annually (about 10% of Norway's total release in 2013; Joosten et al 2015). These numbers are based on an area of 3618 km^2, but most likely the area of peatlands affected by land-use change is closer to 7000 km^2 (Joosten et al. 2015), making the carbon loss estimate a minimum. In addition, some areas have been or are still exploited for excavation of peat to be used as for example growing media for agricultural use (Øien et al. 2017). A mean of 220,000 m^3 peat has been extracted yearly since 1990 in Norway (Øien et al. 2017). Excavated peat will become more or less completely decomposed, releasing all carbon stored.

2.4.3 Prevailing management practices and effects on carbon balance

Direct human interventions are the biggest threat to wetland habitats. Drainage for agriculture and forestry is the main threat to peatlands (e.g. Lyngstad et al. 2018), but also peat cutting, overgrazing, draining for infrastructure, as well as the construction of forest roads and off-road traffic, accelerating in recent decades (Bjerke et al. 2010, Tømmervik et al. 2012).

As wetlands have been under anthropogenic pressure worldwide, the Ramsar convention (the convention on wetlands of international importance) was developed in 1971, and today 169 countries are part of it. The goal is to protect and secure responsible use of wetlands. In Norway, 63 areas are included as Ramsar sites. These are protected areas and altogether they cover 1200 km^2, two thirds being marine wetlands. Norway has developed an action plan for restoring wetlands – Wetland restoration plan in Norway (2016–2020) (Norwegian Environment Agency 2016). One of the goals in the action plan is to reduce carbon emissions from drained wetlands. Drainage of peatlands for forestry was forbidden in 2007, but maintaining already existing ditches is still allowed. In 2019, it was decided to prohibit cultivation of undisturbed peatlands for agriculture, but exceptions may occur.

Restoration of peatland is mostly done by re-establishing the water table level to the ground level (rewetting). This will prevent further carbon emissions from drained peatland sites (Joosten et al. 2015), as the areas again will be saturated with water leading to low oxygen levels and decomposition. Järveoja et al. (2016) studied the effect of restoration by comparing unrestored sites with restored sites three years after restoration. They found that the restored sites had half of the emissions of the unrestored sites (Järveoja et al. 2016). This study therefore suggests that restoration may serve as an effective method to mitigate the negative climate impacts of drained peatland areas.

Joosten et al. (2015) estimated the effects rewetting will have on drained peatland in Norway, and found that the reduction in carbon emissions is highest in sites used as croplands (26.4–33.1 Mg CO_2-e), between 6.0 and 26.4 Mg CO_2-e in grassland and 1.2–11.2 Mg CO_2-e for forest and areas where peat has been extracted. They conclude that the most effective approach to reduce carbon emissions by restoration is to rewet cropland.

Land use change of wetlands often leads to net carbon loss, because change in hydrology shifts the carbon cycling and turns the ecosystems from sinks to sources of carbon. Bárcena et al. (2016) estimated how much carbon emission can be reduced by not turning intact peatlands into agricultural land. The estimates show that that carbon release can be reduced by 25 Gg CO_2-e per km^2 of peatland that is kept intact. They conclude that draining mires for agricultural use does not give a socioeconomical profit, because the emission of CO_2 will be incredibly high. Because of that, wetland protection is preferable to restoration, as the net loss is not compensated when sites are restored even though carbon sequestration is re-established. Despite this, restoration is needed to improve the state of existing, degraded wetlands. Not only to restore the carbon cycle, but also to conserve biodiversity.

2.4.4 Potential effects of climate change on carbon cycle in wetlands

In parts of Norway, climate change will likely result in higher summer temperatures and increased frequency of drought events (Wong et al. 2011), thus, wetlands may be affected negatively. Higher temperatures can turn peatlands from carbon sinks to carbon sources because of drying (Gallego-Sala et al. 2018). Swindles et al. (2019) analysed long term changes in hydrology in European peatlands. They used testate amoebae to reconstruct past water table depths from peat profiles and found that peatlands in general have shifted to drier conditions the past 300 years (69% of sites are now drier, while only 7% have wetter conditions). They found significant drying in Scandinavia the last 400 years (78% sites, 22% unchanged, no sites from Norway), with a transition in year 1777 and a larger transition in year 1990. This correlates with higher summer temperatures and less precipitation in the summer. On the other hand, higher temperatures in combination with high stable water levels may increase peatmoss growth and thus, increase sequestration of carbon (Bengtsson 2019).

Climate-driven drying has been accelerated by human impact the last centuries, and it is impossible to separate the effect of climate change and human impact (Swindles et al. 2019). Swindles et al. (2019) conclude that future climate scenarios will likely lead to further decrease in water tables, which again will lead to loss of carbon stocks following higher aerobic decomposition. Restoration can mitigate loss of soil carbon, but as peatlands seems to be in a state of transition, climate should be taken into account in management strategies, as even undisturbed peatlands seem to be drier. This indicates that we might already have passed the tipping point where peatlands are carbon sinks, but are becoming carbon sources. However, large amounts of carbon is still stored in peat.

2.5 Fresh water, coast and seabed

Some of Norway's healthiest ecosystems are the freshwater rivers and lakes taking approximately 5% of Norway's area. An estimated 5000 animal and plant taxa live in freshwater habitats in Norway, from invertebrates, 42 species of fish and six mammals to over 80 bird species reliant on these habitats (Schartau et al. 2010). Freshwater ecosystems have an overall good state of biodiversity with the Nature Index in 2014 of 0.75 (range 0.71–0.78). However, acid rainfall, increased eutrophication and habitat destruction, particularly by hydropower development, threaten the biodiversity of these ecosystems (Schartau et al. 2015).

Coastal waters are considered as the areas one nautical mile off the shoreline, and Norway's coastline is over 100,000 km long, making it the second longest in the world after Canada. It hosts rich ecosystems from kelp forests on the seabed through to the herring and plankton rich water, that since 2010 has been in decline as human activity both on the coasts and the adjacent land make these ecosystems vulnerable (Gundersen et al. 2015): the Nature Index values for coastal ecosystems have been relatively high, viz. 0.62 (range 0.56–0.68). Approximately 80% of Norway's population live within 10 km of the coasts, and key marine pressures include aquaculture and fisheries, oil and gas production, and the run-off of nutrients and waste from land, through rivers to the ocean (Norwegian Environment Agency 2018).

2.5.1 Carbon cycle in marine and freshwater ecosystems

The oceans contain about 93% of the global carbon stocks (ca. 41 Eg C; Ciais et al. 2013). Most of this is stored as dissolved CO_2 in the deep sea (37 Eg C), followed by ocean floor surface sediments (1.8 Eg C), dissolved CO_2 in the surface ocean (0.9 Eg C), dissolved organic carbon (0.7 Eg C) and marine biota (3 Pg C; Ciais et al. 2013). There is continuous gas exchange between the atmosphere and the ocean. Pre-industrially, the net carbon flux was 0.7 Pg C per year from ocean to atmosphere. Due to human CO_2 emissions, the carbon flux has reversed and now amounts to 2.3 ± 0.7 Pg C per year from atmosphere to ocean (Ciais et al. 2013).

As a consequence of the reversed carbon flux, approximately one third of the total anthropogenic CO_2 emissions since the industrial revolution has been absorbed by the oceans, mainly because of the "solubility pump", i.e. transfer of CO_2 to oceans due to the undersaturation of ocean water. The second process is the "biological pump", which consists of the photosynthetic activity of phytoplankton that extracts CO_2 from the atmosphere, whereas the excess of primary production sinks into the deep sea. The world oceans have thus buffered the climate system by reducing the anthropocentric greenhouse effect (a process that results in ocean acidification). With increasing temperatures, however, this buffering effect will level off, because warmer water masses can store less CO_2.

The organic carbon burial in ocean floor surface sediments, which permanently removes carbon from the carbon cycle (and, in geological time scales, contributes to building up fossil carbon stocks), amounts to roughly 0.2 Pg C per year (Ciais et al. 2013). An estimated 50–70% of this organic carbon burial in oceans occurs in coastal vegetated habitats, even though these habitats only occupy ca. 0.3% of the oceanic area (Nellemann et al. 2009).

As regards freshwater systems, rivers globally transport 2.9 Pg C per year, which are washed out from soils or originate from rock weathering. Lakes lose 1.4 Pg C per year to the atmosphere due to outgassing, while 0.6 Pg C per year are buried in lake sediments. The remaining 0.9 Pg C per year run off into oceans (Kirschbaum et al. 2019).

2.5.2 Carbon sequestration and storage in marine and freshwater ecosystems

Globally, roughly half of all "green carbon" (i.e. carbon stored in living biomass) is captured by marine and coastal organisms and has therefore been referred to as "blue carbon" (Nellemann et al. 2009). The most important ecosystems in this regard are mangrove forests, tidal saltmarshes and seagrass meadows (Laffoley & Grimsditch 2009).

Of these ecosystems, only saltmarshes and seagrass meadows exist in Norway (**Table 5**). Complicating matters, estimates from other countries are not necessarily transferable to Norwegian conditions even for these ecosystems. For instance, saltmarshes are indicated to have extremely high rates of carbon burial ($210\,g\,C\,m^{-2}\,yr^{-1}$; Laffoley & Grimsditch 2009), but it is doubtful whether the Norwegian saltmarshes are comparable in productivity and carbon sequestration to the saltmarshes on which these estimates are based (Gulf of Mexico, Wadden Sea, Mediterranean). **Table 5** uses estimates from Great Britain (Chmura et al. 2003) and assumes that these are applicable only to a small part of the relevant nature types in Norway, viz. saline foreshores (T11, *saltanrikingsmark i fjæresonen*) and tidal meadows (T12, *strandeng*) according to Nature in Norway (Norwegian Biodiversity Information Centre 2018).

While seagrass meadows occur in Norway, their area is rather restricted: the area of occupancy of "sea meadows" (M7, *marine undervannsenger*) is estimated at $93\,km^2$ (Gundersen et al. 2018a). In addition, the global sequestration estimates are based on subtropical seagrass species that do not occur in Norway. Norwegian seagrass meadows are dominated by eelgrass (*Zostera marina, ålegress*), for which estimates are presented in **Table 5**.

Other marine ecosystems include intertidal mudflats, seabeds and coral reefs. Coral reefs do not act as carbon sinks (Laffoley & Grimsditch 2009) and do not occupy large areas in Norwegian waters, and they are therefore not considered here. Based on data in France, intertidal mudflats may have a gross primary production and respiration of approximately 245 and $110\,g\,C\,m^{-2}\,yr^{-1}$, respectively (Spilmont et al. 2006), whereas the burial rate of $16\,g\,C\,m^{-2}\,yr^{-1}$ is taken from British estimates (**Table 5**).

Stable (i.e. rocky) seabeds are relevant as the substrate for kelp forests and intertidal algae. Gundersen et al. (2011) have estimated the current and potential area occupied by these ecosystems, their biomass and carbon sequestration (summarised in **Table 5**). The primary production and carbon sequestration of kelp is in the order of its own biomass per year, i.e. roughly $3.6\,Tg\,C\,yr^{-1}$. As brown algae grow directly on the rocky seabeds, kelp forests do not have sediments, and therefore no carbon burial takes place in kelp forests. However, an unknown fraction of the dead kelp biomass will eventually become deposited elsewhere. The estimate in **Table 5** is based on Gundersen et al.’s (2011) assumption that this fraction is in the order of 3%.

***Table 5.** Estimates of standing carbon stocks (storage) and annual burial in marine and freshwater ecosystems. Estimates are provided for relative rates (i.e. per square metre), the area covered by the ecosystem in Norway, and the accumulated totals for Norway. Rates are partly based on estimates from other countries, some areas are just "educated guesses". References are indicated by footnotes.*

	Relative estimates			Totals for Norway			
	Standing stock / g C m^{-2}		Burial / g C m^{-2} yr^{-1}	Area / km^2	Standing stock / Gg C		Burial / Gg C yr^{-1}
Ecosystem	Biomass	Sediment			Biomass	Sediment	
Kelp forests[a]	450	0	30	8000	3600	0	250
Intertidal algae[a]	225	0	20	180	40	0	4
Seagrass meadows[b]	80	4900	20	93	7	460	2
Saltmarshes[c]	500	20,000	100	100	50	2000	10
Intertidal mudflats[d]	20	2000	16	1000	20	2000	16
Freshwater lakes[e]	3	50,000	5	18,000	50	900,000	90

[a] All figures from Gundersen et al. (2011)

[b] Stock from Röhr et al. (2018) for Skagerrak; burial from Röhr et al. (2016) for Limfjorden; area from Gundersen et al. (2018a)

[c] Sediment and burial from Chmura et al. (2003) for Great Britain

[d] Burial from Alonso et al. (2012)

[e] Biomass from Cyr and Peters (1996); sediment stock is a rough extrapolation based on sedimentation since the last glaciation, compatible with estimates from North America (Alin & Johnson 2007, Munroe & Brencher 2019); burial from Algesten et al. (2003) for Sweden; area from Norwegian Water Resources and Energy Directorate (2018) for all Norwegian lakes ≥ 0.0025 km^2

Since the 1950s, the area occupied by kelp forests has been reduced by approximately 55% (Gundersen et al. 2011), mainly due to grazing by the green sea urchin (*Strongylocentrotus droebachiensis*, *drøbaksjøpinnsvin*). The reasons for this change are still poorly understood, but may include the reduction in the population size of predators such as Arctic cod (*Gadus morhua*, *torsk*). In recent years, kelp has started to re-establish along parts of the coast (Gundersen et al. 2018b). If the entire loss of kelp biomass could be reversed, the standing carbon stock of Norwegian kelp forests would increase by 4.5 Tg C, a process that would take roughly 30 years (Gundersen et al. 2011). This would correspond to an additional carbon sequestration of 150 Gg C yr^{-1} for the duration of this re-growth period.

Based on detailed figures estimated in Sweden (Algesten et al. 2003) and some additional estimates from Norway (Larsen et al. 2011), the total carbon flow in Norwegian freshwater ecosystems can be summarised as follows: Rivers take up 1.7 ± 0.9 Tg C per year from terrestrial ecosystems. Of this amount, 91 ± 75 Gg C per year are emitted from rivers to the atmosphere, 730 ± 720 Gg C per year are emitted from lakes to the atmosphere, 94 ± 57 Gg C per year are buried in lake sediments, and the remaining 810 ± 400 Tg C per year are transported to the sea. The emission from lakes to the atmosphere is the net rate, meaning that lakes are net sources of carbon. The primary production (and thus carbon sequestration) in Norwegian lakes amounts to 26 ± 20 Gg C per year, which is roughly 30 times less than the carbon emission by lakes.

In addition to CO_2, lakes emit methane. The amount of methane emissions is approximately two orders of magnitude smaller than the amount of CO_2 emissions from lakes (Ciais et al. 2013). However, due to the higher global warming potential of methane, the warming effects of methane and CO_2 emitted from lakes are in the same order of magnitude. In contrast, the amount of methane emitted from marine ecosystems is negligible (Alonso et al. 2012).

2.5.3 Prevailing management practices and effect on carbon balance

Freshwater ecosystems are not managed in any way comparable to most terrestrial systems. If considering the building of reservoirs as a kind of freshwater management, this practice increases both carbon sedimentation and methane production (Dean & Gorham 1998, Battin et al. 2009). Hydroelectric reservoirs can substantially alter both the biodiversity and the carbon budgets of freshwater ecosystems, from initial flooding which increases GHG emissions (Mendoca et al. 2012a), to long-term changes in sedimentation which may increase carbon burial through greater sedimentation: globally, organic carbon burial in freshwater reservoirs is estimates to be 1.5 kg CO_2-e m^{-2} yr^{-1}, which is twice their estimated CO_2-e emission rate (Mendoca et al. 2012b). Thus, management of these reservoirs in Norway, and in particular sediment accumulation, dredging and removal can be expected to have an impact on the overall carbon budgets of these particular freshwater bodies.

In marine ecosystems, kelp harvest and fish farming may be considered as management. Kelp harvest amounts to approximately 10 Gg per year, which is a negligible quantity compared to the natural changes in kelp. Fish farming concerns mainly Atlantic salmon (*Salmo salar, laks*). Given that Norway currently produces 1.3 Tg farmed salmon per year, this activity entails that 730 Gg C per year are added to the coastal ecosystem in the form of fish feed. Of this amount, 220 Gg C per year are converted into fish biomass and subsequently harvested, 350 Gg C per year are respired by the fish (i.e. returned to the atmosphere as CO_2), whereas 160 Gg C per year are added to seawater as POC (Wang et al. 2012).

A couple of geo-engineering proposals have been made that might increase the carbon uptake of marine ecosystems, such as ocean fertilisation, ocean liming or CO_2 injection to deep water layers (Nellemann et al. 2009). However, none of these has been applied on a large scale, some of these have highly uncertain effects, and all of these can be expected to have negative side effects on biodiversity.

The most efficient management practice in marine systems is to prevent the degradation of coastal ecosystems such as seagrass meadows, kelp forests and saltmarshes. The two former are at the same time extremely important spawning areas for many species. The main human impacts that may lead to degradation of these ecosystems are constructions (e.g. of marinas) and eutrophication (e.g. due to salmon farming; see Gundersen et al. 2018a, 2018b).

2.5.4 Potential effects of climate change on carbon cycle

Seagrass meadows can only grow under conditions of weak to moderate wave exposure, so seagrass will be negatively affected by increasing storm intensity. Seagrass requires high light availability and thus responds negatively to decreasing water clarity; however, heavy precipitation events increase the turbidity of coastal waters due to rivers transporting suspended sediments to the sea. Finally, seagrass would profit from continuing ocean acidification, as this increases their competitive ability against macroalgae. The overall effects of climate change are thus uncertain, but most likely negative.

Kelp forests tolerate more wave exposure, but may also be negatively affected if storm intensity increases too much. Rising sea temperature would likewise be negative for kelp, although it seems that the green sea urchin is even more sensitive to warming, so that rising temperatures may temporarily lead to improved conditions for northern kelp forests (Gundersen et al. 2018b).

In freshwater systems, it has been shown that carbon sedimentation rates have been increasing during the last century (Heathcote et al. 2015), a pattern that can at least partially be attributed to climate change (de Wit et al. 2016, Rantala et al. 2016). Also the amount of TOC in lakes and the carbon export to the sea are predicted to increase due to climate change by 65% and 28%, respectively (Larsen et al. 2011).

2.6 Norway's contribution to carbon storage and sinks

In considering the carbon stored in key ecosystem types, we find that Norway contains approximately 0.18% of all global carbon stocks, with a land mass that is 0.07% of the planet. A third of all of Norway's carbon is stored in the forests, followed by the alpine zones, wetlands and then sediments in freshwater lakes (**Figure 8**). Of all habitat types, it is the living forests and lower alpine zones of shrub vegetation that sequester the most carbon on an annual basis (5.5 and an average of 5.3 Tg C yr^{-1}, respectively) (**Table 6**). When corrected for area, it is freshwater lake sediments, wetlands and the permafrost in the cryosphere that store the most carbon per km^2 (**Figure 9**).

Norway emitted 52 Tg CO_2-e (population 5,337,962) in 2018 (Statistics Norway 2019b). Compared to other Scandinavian countries this is high, and whilst overall emission rank Norway 61st out of 213 countries, the emissions per person in Norway for 2018 rank the country at 39th highest: results that put per capita emissions higher than that of the UK, China and India[3], with currently implemented policies considered "highly insufficient"[4]. Considering that 56% of all of the tundra permafrost in the Scandinavian peninsular is within Norwegian boundaries, Norway has a unique responsibility to reducing carbon emissions, and initiating programmes that seek to mitigate further losses from the landscape.

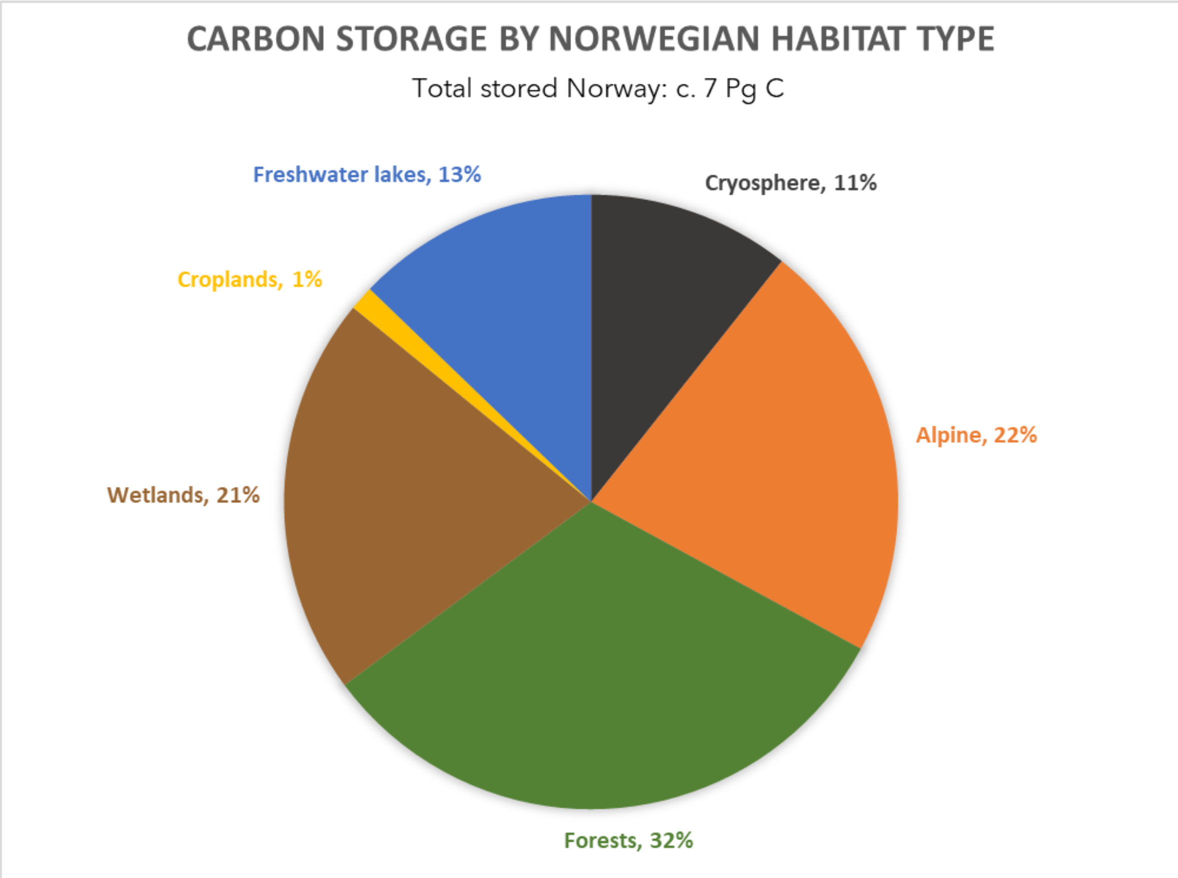

Figure 8. *Approximate amount of carbon stored in Norwegian habitat types, as a proportion of the total carbon stored. Where ranges of carbon storage are reported, the average between high and low is taken. Carbon in freshwater is largely from deep lake sediment. (See Table 6).*

[3] Global Carbon Atlas 2018, carbon dioxide emissions rankings. Available from: http://www.globalcarbonatlas.org/en/CO2-emissions

[4] Climate Action Tracker 2019 country summary. Available from: https://climateactiontracker.org/countries/norway/

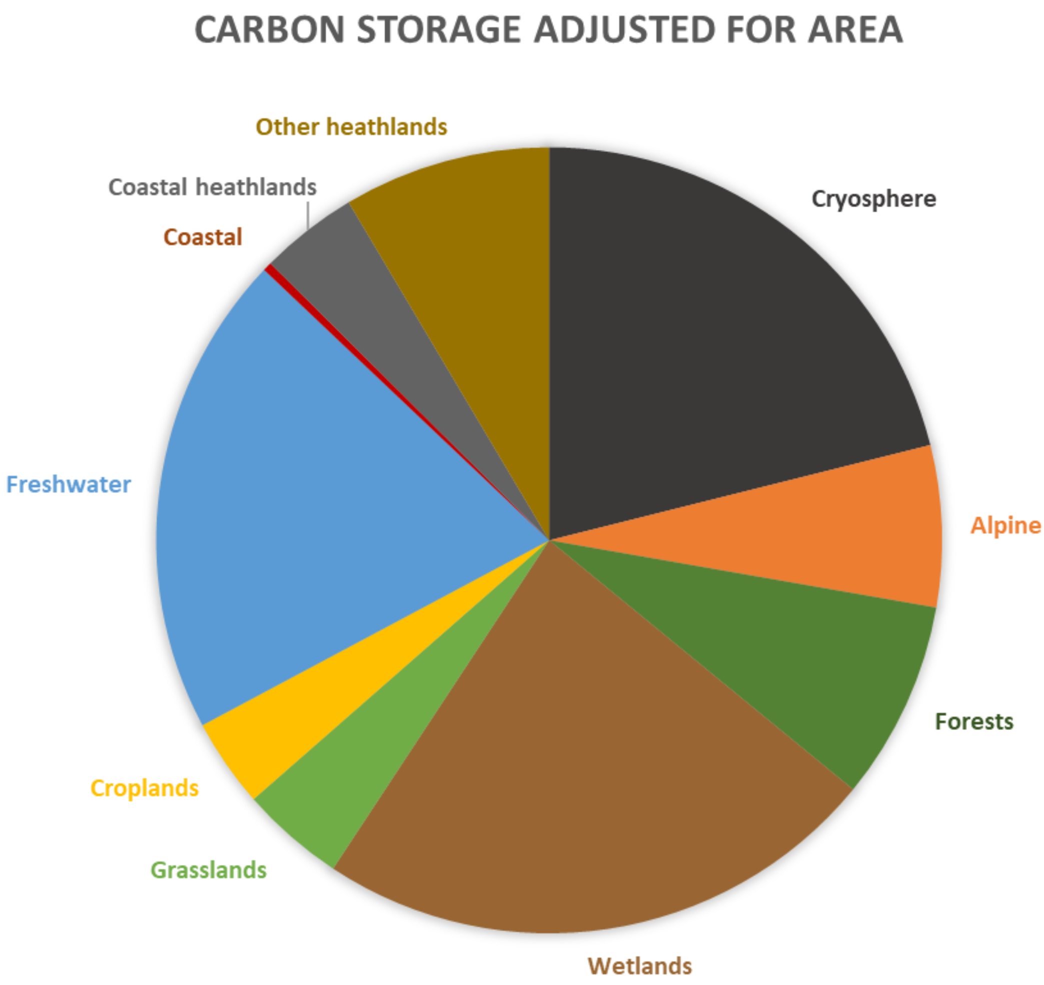

Figure 9. *Approximate amount of carbon stored in Norwegian habitats corrected for the area of assessed habitat. Proportions calculated as total carbon (Gg) per km^2 of area land cover based on assessments in Section 2 (therefore Gg C km^{-2}). Carbon in freshwater is largely from lake sediments. See Table 6 for details and uncertainty.*

3 Climate mitigation and adaptation, and the protection of biodiversity and ecosystem services

3.1 Norway's efforts to comply with the Paris Agreement under the UNFCCC

Following the commitments with the Paris Agreement under the United Nations Framework Convention on Climate Change (UNFCCC), the Norwegian Government has put in place key measures to achieve climate mitigation targets for 2030, in line with those set by the European Commission. The measures consist of three pillars: (1) the EU emissions trading system (EU ETS), (2) measures for those sectors not included under the EU ETS, which are regulated by the Effort Sharing Regulation (ESR, see below), and (3) measures addressing 'Land Use, Land-Use Change and Forestry' (LULUCF) as defined in the Kyoto Protocol (UN 1998) (see **Appendix 7.2**).

Norway, in line with the EU, has set ambitious goals of emissions reduction, which will necessarily imply profound changes in the society due to the magnitude of their scale, the need for lifestyle changes, the technological innovation requirements and the adaptation of productive systems. The transition to a low-emissions society will also imply changes with potentially high impacts on Norwegian ecosystems and their capacity to remove GHG and to generate the many other benefits society derives from them.

The recent report Klimakur 2030 ("climate cure"; Norwegian Environmental Agency 2020) presents a suite of 60 potential physical measures that could be implemented under Pillars 2 and 3 of the climate mitigation strategy. It is a comprehensive report with projections of GHG emissions by the different sectors from 2021 until 2030, including an analysis of the contribution of these measures to reduce emissions, and an estimation of their costs.

3.2 Potential impacts of Klimakur 2030 on Norwegian ecosystems

3.2.1 Carbon emissions accounts and reports from land systems

The capacity of land systems to mitigate GHG emissions is essentially about the capacity of plants and soils to absorb and retain CO_2 from the atmosphere through the process of photosynthesis, and by the biological processes that build stable soil organic matter. However, the carbon cycle in forests and other ecosystems is complex, including spatial and temporal dynamics, interactions among carbon pools, and natural and human influences (Janowiak et al. 2017), and the evidence to support current carbon emissions calculations is still limited. The IPCC Guidelines to estimate and report emissions (IPCC 2006) indicate three levels of detail of the data supporting LULUCF calculations, and acknowledge, for instance, big knowledge gaps about the dynamics of dead organic matter. New data on these will help to identify, quantify and reduce uncertainties in the years to come. Although coarse-level information is useful for raising awareness, data-intensive approaches are necessary for decision-making instances when important environmental and economic consequences are at stake (Gómez-Baggethun & Barton 2013). This is highlighted by Brown (2020), who state that "over reliance on simple symbolic target indicators… is likely to be ineffective and misleading for referencing climate change policy progress…". Thus, better knowledge of carbon dynamics and storage within all ecosystems (including soils), combined with important area coverage and responses to ecological variability and management interventions, are needed (IPCC 2006).

One limitation in using the emissions accounting system in the LULUCF to support land-based emissions mitigation measures, is that the land-systems in LULUCF include only those relevant for the ‘land-use sectors’, e.g. forest under forest management, and conversion from forest and natural systems to agricultural land and vice versa. Unmanaged forests and other ecosystems are not considered as anthropogenic GHG sources or sinks, and are excluded from LULUCF inventory calculations (IPCC 2006), which means that human interventions (negative and positive) on carbon pools, such as infrastructure construction, urbanization and ecosystem restoration are not considered in ecosystems outside the ‘land-use sector’. The Norwegian LULUCF classes include managed areas of: forest, arable land, pasture (not used as arable land), water and mire, and other open areas (i.e. mainly alpine heathland and ice). Further, since the most detailed information on carbon sequestration and storage is that available in the national forest inventory data, the main focus of carbon removals and emissions accounts has been on the role of trees, while there are important knowledge gaps on ecosystem level carbon budgets for forests and other terrestrial ecosystems, which are arguable equally, if not more important for national carbon budgets (see **Figure 9**).

3.2.2 Emissions mitigation measures under Pillar 2 and their impacts on ecosystems, carbon pools and GHG emissions

The transport sector, under Pillar 2 (see Appendix 6.2), has the largest emission sources and hence, the highest potential to contribute to a reduction of GHG emissions. The measures proposed for the sector are mainly based on the replacement of fossil fuel-driven vehicles by electricity-driven vehicles, both on land and at sea. The extensive electrification of the transport sector will require high investments in renewable energy projects, an extensive network of powerlines and charging stations, and the expansion of the railway network (Klimakur 2030). This means that large impacts on terrestrial ecosystems, and their carbon pools and emissions are to be expected. Part of these impacts are evident in recent LULUCF reports, showing that infrastructure development and urbanization have been the main causes of deforestation in the past years, with significant consequences for GHG emissions (Klimakur 2030).

Another measure to reduce CO_2 emissions from the transport sector is to include 10% of biofuels in current fuels, to be used in the remaining non-electrified vehicles. This 10% is intended to be ‘advanced’ biofuels, which means that biofuel sources should come either from waste or from other biomass, such as that coming from forest management practices (thinning) and residues after harvest. The use of advanced biofuels can have important consequences on forest carbon pools and nutrients, but there are few studies clearly documenting the effects of management (IPCC 2006) on litter carbon. The IPCC guidelines (2006) recommend countries experiencing significant disturbance or management regimes in their forests to develop higher detail level data to quantify the impacts from these changes and to report the resulting stock changes and non-CO_2 emissions. However, and despite this recommendation, the use of forest products for bioenergy is uncritically considered a carbon neutral energy source. Research has shown that the time needed to reabsorb the extra carbon released in bioenergy products from forest can be very long, so that current policies risk achieving the reverse of that intended – initially exacerbating rather than mitigating climate change (Norton et al. 2019). The review by Norton et al. (2019) points to possible reforms of current policies that would reduce negative impacts on climate by explicitly requiring biofuel sources with short payback periods.

Significant infrastructure developments disturb whole ecosystems from ancient soil profiles to wetlands and river systems. The largest disturbances in alpine areas, which cover just over a 3rd of the country, and provide 22% of its carbon stores (**Figure 8**), come from hydroelectricity development and the associated access roads, dams, flooding and forest clearance (Edenhofer et al. 2011). Norway has 1660 hydropower plants and 1000 storage reservoirs, most of which are in the mountains of South Central Norway, or in the sub-Arctic North[5]. The loss to alpine wilderness varies from ~2 to ~12 km^2 of land to every GWh of electricity produced, but the effect this will have on ecosystem carbon budgets is difficult to assess due to the land-use change at both

[5] https://energifaktanorge.no/en/norsk-energiforsyning/kraftproduksjon/

a terrestrial and aquatic level, and reduced national CO_2 consumption in the long-term as a result of renewable energy use (Bakken et al. 2017). When considering a boreal landscape flooded, dammed and a hydroelectrical plant installed and generating electrical output over decades, Teodoru et al. (2012) found that the land-use change resulted in overall carbon emissions compared to the previous landscape, having turned the landscape from a carbon sink to a carbon source. However, the long-term benefits ultimately reduce CO_2-e emission to 40% of that of a fossil fuel equivalent.

3.2.3 Emissions mitigation measures under Pillar 3 and their impacts on ecosystems, carbon pools and GHG emissions

Based on the LULUCF land-uses, the proposed emissions reduction measures include increasing the area of forest cover by afforestation, expanding forest management practices that increase tree growth (i.e. carbon accumulation in tree stems), and reducing deforestation and conversion of natural systems to agricultural land. In line with the IPCC recommendations to cut emissions from the LULUCF sector, Klimakur 2030 proposes three key measures: increasing tree planting density; fertiliser application in forestry; and expanding the forest area (afforestation). These management practices have important consequences on carbon stocks and dynamics, and detailed data are recommended to guide management decisions since there are large uncertainties about the carbon dynamics and flows involved (IPCC 2006, Brown 2020).

High density tree plantations have little under-storey vegetation which results in impacts on carbon removals. For example, under-storey vegetation contributes to a large portion of carbon sequestration and storage, as well as biodiversity, in old-growth forests (Wardle et. al. 2012). If estimates of carbon removal capacity are only based on tree growth (i.e. increase in timber volume, IPCC 2006), significant parts of the carbon sinks in forests with rich under-storey vegetation will remain unaccounted. Further, densification measures are likely to be implemented in highly productive forests, which harbour the highest species diversity of different taxonomic groups in Norway. It is estimated that 60% of Norway's estimated 40,000 species are associated with forests, with the cycle of dead and living wood supporting a wide variety of organisms (Henriksen & Hilmo 2015).

In addition to the impacts of high-density plantations on biodiversity, this kind of forestry management may present high risks for Norwegian forestry in the face of climate change. A recent report from Sweden (Skogsstyrelsen 2020) estimates potentially large losses in growing stock and environmental and societal values in high-density forests due to the higher risk of storms, and pathogenic fungi and insect outbreaks as a consequence of climate change. To reduce the risk of damage, the Swedish forest authorities recommend a higher diversity in forest management practices, and the avoidance of monocultures with high density of trees and particularly the fast-growing Sitka spruce (*Picea sitchensis*). Other recommendations are to avoid large areas of clear-cutting to reduce the risk of soil erosion, a major cause of loss of organic soil carbon, and land movements. After the storm Gudrun in 2005, which caused wind-falls in 100,000 ha of forest in Sweden, the forest authorities also recommend to avoid planting spruce, since spruce seems to be a species particularly sensitive to wind-falls, and other climate related damages. In Scotland, the majority of new woodland planting is still with Sitka spruce which whilst fast-growing, requires the drainage of the wet ground it is often planted on, leading to soil carbon loss, reduced biodiversity and transformation of the land from a carbon sink to a carbon source up to 30 years after planting, and decreasing the adaptability of the landscape to climate change (e.g. Vanguelova et al. 2019, Brown 2020).

Forest fertilization is another proposed practice with the potential to highly impact the environment through: pollution/eutrophication; reductions in plant and fungal diversity; changes in bacterial diversity; and in changes in the level of GHG emissions. Nitrogen fertilization increases N_2O emissions, which is a potent GHG (300 times more powerful than CO_2), and is at odds with

Norway's compliance to the Gothenburg Protocol[6] which aims to reduce the emissions of NO_x, by limiting the release of nitrogen in nature (Aarrestad et al. 2013, p. 40, 41–48, and 53, respectively). The impacts of fertilization are strongest on species adapted to chronically nutrient-poor environments (Aarrestad et al. 2013), and include effects on red-listed habitat types such as species-rich hay-meadows and semi-natural grasslands, as well as nutrient-poor alpine habitats. The risks of nitrogen leakages from fertilized forests or from fertilizer applications into these habitats need to be evaluated and given serious attention. Nitrogen fertilization can also increase the sensitivity to drought and pest attacks, which are expected to increase in Norway during the summers (Wong et al. 2011). One of the mechanisms of increasing susceptibility to pest attacks is the fertilizer-induced changes in the chemical defense of conifer needles (Nybakken et al. 2018).

Afforestation, or the expansion of forest area, can potentially have important negative effects on biodiversity, especially in the case of open semi-natural habitats. These are important areas for the conservation of light-demanding plant species and the organisms that use them, which have been commonly used in the agro-pastoral landscape in Norway, but which have decreased significantly due to shrub and tree encroachment. As indicated in earlier sections of this report, grasslands, heathlands and wetlands have a very high potential to store soil carbon, a characteristic that has been largely underestimated, so the plantation of trees in these areas may not render the expected carbon removal levels.

Facilitating the adaptation of Norwegian ecosystems to climate change will be a crucial consideration in managing the nation's carbon budget. The melt of permafrost is unlikely to be halted at this stage, and as the risk of landslides and floods in mountain regions will increase as a result of increased ground instability, adaptation to long term changes are recommended by the IPCC. However, the natural shrubification of alpine ecosystems in response to warming may counteract permafrost respiration (i.e. release of carbon) in these regions, although there are several confounding variables. Initiatives that investigate grazing pressures, development and infrastructure in alpine and particularly tundra regions, may lead to advances in understanding how we can enhance these ecosystems' ability to adapt to climate change. Furthermore, a decrease in geographical barriers, ecosystem degradation, and habitat fragmentation will increase the potential for species to shift their ranges upwards, and northwards naturally. Likewise, initiatives that promote landscape restoration, particularly for peatlands, and ecosystem management focussed on increasing ecosystem productivity, can be effective (IPCC SROCC 2019).

Since the beginning of the IPCC, LULUCF reporting and accounting systems have involved long processes of international negotiations and by necessity, compromises. Thus, it is understandable that it carries several limitations, particularly regarding the limited evidence about the temporal and spatial variation in carbon pools (Janowiak et al. 2017, Norton et al. 2019). Under these circumstances, and considering that large portions of Norwegian land-systems are not included in the accounting, the **precautionary principle** would be applicable whereby decision-makers are enabled to adopt precautionary measures when scientific evidence about an environmental or human health hazard is uncertain and the stakes are high[7]. Decisions would consider that well-functioning ecosystems, including soil, at their highest biological capacity have the potential to deliver highest levels of ecosystem services associated with climate mitigation functions. For instance, a recent study of soil biological networks shows that nature restoration on abandoned arable land tightened belowground biological networks, which led to enhanced efficiency of carbon uptake (Morriën et al. 2017). Also, Buzhdygan et al. (2020) show that higher diversity of plants and other trophic groups above- and below-ground resulted in

[6] Gothenburg Protocol. 1999 Protocol to Abate Acidification, Eutrophication and Ground-level Ozone. https://www.unece.org/environmental-policy/conventions/envlrtapwelcome/guidance-documents/gothenburg-protocol.html

[7] European Parliament, Think Tank. https://www.europarl.europa.eu/thinktank/en/document.html?reference=EPRS_IDA(2015)573876

more energy stored, greater energy flow and higher community energy-use efficiency across the entire plant-soil biota trophic network.

3.2.4 Options to increase GHG removals and reduce emissions from land-systems while enhancing co-benefits

Options based on **the precautionary principle** could more effectively account for the overall carbon removal and storage capacity of Norwegian ecosystems and address the risks of increased emissions under land-use and climate change. It would provide room to evaluate options that not only maximize carbon uptake and, for example, timber production, but also can identify and reinforce synergies that optimize the capacity of ecosystems to generate multiple benefits. For instance, several 'natural climate solutions' – conservation, restoration, and/or improved land management actions – have been proposed to increase carbon storage and reduce emissions in ecosystems (Griscom et al. 2017). Such solutions would be in line with those proposed in the recent climate change and land report (IPCC 2019) and the IPBES Global Assessment (IPBES 2019a), where important trade-offs are identified between climate mitigation actions and the protection and sustainable use of biodiversity and ecosystems, and where cross-sectoral concerted actions are necessary to best solve them.

Therefore, cross-sectoral, pro-active and innovative solutions will support the transformation needs indicated in the IPBES report (IPBES 2019a) to address the challenges of biodiversity loss and ecosystem services degradation to be identified. This could include alternative measures to densification and fertilization in forestry as well as afforestation, such as ecosystem restoration, which has a large potential to produce co-benefits to climate mitigation measures (IPCC 2018). Restoration would also enable the implementation of Norway's international commitments under the Convention on Biological Diversity (CBD), e.g. the Aichi Biodiversity Target 15: "*By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks have been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation* …" In particular, the **restoration of forests and mires** can combine enhanced GHG removals, reduced emissions, gains in biodiversity conservation and delivery of other ecosystem services, including regulating services for climate change adaptation. Efforts to maintain forest cover help to maintain the capacity of the land to remove atmospheric carbon, by preventing emissions and by increasing the potential for additional sequestration. Compared to afforestation, which takes very long time to recover forest carbon stocks, restoration of mature forest stands would provide considerable benefits due to the high volume of tree biomass as well as soil carbon (Janowiak et al. 2017; Wardle et al. 2012). Compared to afforestation, the **restoration and maintenance of open semi-natural habitats**, including the harvest of exotic invasive species, would have a huge value for biodiversity conservation, while being an important source of biofuel material. In addition, open semi-natural habitats are key resources for pollinators (Norwegian Ministries 2018). The strong reduction in area of open semi-natural habitats due to abandonment or lack of management in recent decades is considered one of the major factors associated with the decline of wild pollinator populations in Norway and globally. Further, old-growth forests in the proximity of open semi-natural habitats significantly predict the occurrence of wild pollinators in Norway (Markus Sydenham, unpublished data) because old-growth forests provide nesting resources for many bee species. Hence, the restoration of semi-natural open habitats combined with areas of old-growth forest would contribute considerably to the implementation of Norway's national pollinator strategy (Norwegian Ministries 2018), thereby maintaining important carbon stocks and sinks while rendering significant co-benefits with other national priorities.

If deployed at large scale, restoration measures would require governance systems that enable sustainable land management to conserve and protect land carbon stocks and other ecosystem functions and services (IPCC 2018). However, although there are currently gaps in policies supporting ecosystem restoration (Rønningen & Follo, pers. comm.), instruments such as direct payments similar to other environmental schemes in agriculture and forestry could be implemented in the short-term to support land-owners to initiate and maintain restoration practices. In the longer term, synergistic policy mixes could be developed (Barton et al. 2017). Direct payments schemes (e.g. performance-based to improve effectiveness) could be financed through modalities of habitat off-setting (e.g. Bernasconi et al. 2014), potentially in combination with enhanced Environmental Impact Assessment standards, for instance, in connection with infrastructure development. Part of these policy mixes could include the development of new products and creation of markets for products from restored ecosystems, and promotion of practices and capacity building on practical restoration techniques of various types. Examples of options would also include the business and finance sectors, which can be drivers of positive change while addressing biodiversity impacts across the entire value chain (Mace et al. 2018).

Longer rotation times in forestry has been suggested as a way to reduce the carbon emissions (Framstad et al. 2013). Longer rotation times indeed lead to a greater ecosystem carbon stock (Liski et al. 2001, Lundmark et al. 2013). Lundmark et al. (2013) showed that prolonging the rotation period by > 10 yr led to decreased carbon storage in forest products and decreased substitution effects, but the net climate benefit was maximized at the longest rotation period examined (+30 yr). Also the importance of under-storey vegetation in carbon removals significantly increases in old compared to young forests (Wardle et al. 2012). At the same time, longer rotation times increase the spatiotemporal continuity of forest cover, thereby alleviating dispersal limitation and allowing more time for dispersal and colonisation of rarer forest species (Nordén et al. 2018). Longer rotation times also enable the forest stands to develop qualities similar to those of old-growth forest (which currently remain largely unprotected; Sverdrup-Thygeson et al. 2014); hence providing habitats for species dependent on them.

Continuous-cover forestry (CCF) has also been shown to improve forest carbon storage and sequestration (Peura et al. 2017), but Lundmark et al. (2016) conclude that the carbon balances of clear-cut and CCF sites are not necessarily different, since the carbon balance depends more on biomass growth and extraction than silvicultural management. As in the case of rotation times, the gains in terms of creating habitats for forest species can be higher in CCF systems compared to the clear-cutting practice.

Reduced harvesting of forests has been called for (Norton et al. 2019, see also Ter-Mikaelian et al. 2015) to create substantial cuts in carbon emissions quickly, as requested by IPCC (2018, 2019). Increasing the area of **protected forests** and **restoring formerly managed forests** towards a more natural state have been suggested both for the purposes of biodiversity protection and improved carbon storages (Bradshaw et al. 2009). Forest restoration is being done in Northern Europe (Similä & Junninen 2012, Halme et al. 2013), but not yet in Norway, apart from recent pilot projects such as TRANSFOREST (NINA-led project) and another project run by the nature conservation association NOA. Forest restoration can have very high impact on the diversity of many forest species from different taxonomic groups (Nordén & Olsen 2017).

The target locations of emission mitigation measures (e.g. afforestation, restoration) is critical for the effectiveness of the implementation, specifically because the biophysical and the socio-economic context determines the outcomes of these measures. For instance, Brown (2020) analysed actual locations for recent afforestation and peatland restoration, and showed that the areas targeted constrain net carbon gains. Hence, **territorial planning** is a crucial tool to achieve effectiveness in the implementation of measures. Further gains in effectiveness can be achieved, when multiple objectives are considered simultaneously in a spatial context, such as the avoidance of areas where damage to species and ecosystems by infrastructure development, forest habitat destruction and eutrophication can be significant. Several tools have been developed in the past decades in the field of systematic conservation planning that could support spatial planning (e.g. Schröter et al. 2014, and Hanssen et al.[8]) by helping to identify areas for actions where benefits can be maximized and conflicts and costs minimized. Spatial planning where multiple objectives are considered simultaneously will optimize climate mitigation and ecosystem adaptation benefits. For example: avoidance of unnecessary damage to ecosystems (e.g. forest habitat destruction and eutrophication), will mitigate soil erosion, pest outbreaks, and maintain biodiversity and ecological adaptations.

Life Cycle Assessments provide a useful tool to establish the impact of policies or specific measures on the environment, usually through measures including the land-use impact category 'land use and land use change' (LULUC). However, it is often not used since there is no uniform methodology on how to incorporate impacts on biodiversity (Lillesand et al. 2017 and references therein). Comparing land use impacts of hydropower plants show large variation of impact indicating the importance of including LULUC in such project (Lillesand et al. 2017).

A tool with elements of Life Cycle Assessment methodology has been developed in Scotland, viz. the carbon calculator, responding to assessment needs about the impacts of the development of large-scale wind farms. The carbon calculator aims to minimize the damage to biodiversity and carbon emissions following land use change (Nayak et al. 2008). The reason is that vast areas of peatlands may be affected by wind farms, impacted peatlands that may release more carbon than is actually saved by switching from fossil fuel to electricity. The carbon calculator includes both carbon removals and carbon emissions from all parts of the wind farm development. Carbon calculations are based on the following: (1) reduced carbon emissions from different power sources, (2) emission of carbon due to production, transportation, operation and decommissioning of the wind farm, (3) emission of carbon from backup power generation, (4) loss of carbon-fixating potential of peatland, (5) release and/or uptake of carbon stored in peatland by peat removal or changes in drainage, (6) carbon storage due to improvement of habitat, and (7) increase or decrease of carbon-fixating potential as a result of forestry clearance[9]. The carbon calculator is available online[10].

[8] ConSite. Consensus-based siting. https://www.nina.no/consite

[9] https://www.gov.scot/publications/carbon-calculator-technical-guidance/

[10] https://informatics.sepa.org.uk/CarbonCalculator/index.jsp

4 Naturkur ("nature cure")

Since the ratification of the Kyoto Protocol in 1998, progress has been made by the global community to address the challenges of human-induced climate change, and the need to stabilize the Earth's climate. A large number of assessments have been produced in the past two decades accumulating evidence and proposing measures and mechanisms to reduce the emissions from fossil energy sources, and from changes in land-use and land-management practices.

Paralleling the IPCC process, in 2012, 94 governments established the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) as an independent body established by states to strengthen the science–policy interface for the conservation and sustainable use of biodiversity, long-term human well-being and sustainable development. In the past two years, the IPBES has produced a series of assessment reports about the status and trends of nature, of the services it generates to society, and it made projections about the consequences of current trends of societal development on the life-supporting systems on which societies build.

One key message in the IPBES Global Assessment (2019a) is that land-use change, i.e. conversion of use and land-use intensification, is the main driver of biodiversity loss in terrestrial ecosystems. Land-use intensification results in the overall biological homogenization of landscapes, leading to the loss of genetic diversity and species with local adaptations, and in biological assemblages with stronger dominance of generalist species.

The IPBES Global Assessment report also points to the challenges of implementing measures to remove atmospheric carbon and reduce emissions from fossil energy sources which can aggravate current land-use change pressures on biodiversity and ecosystems. Several of the measures in Klimakuren's Pillar 3 point in this direction as well. The IPBES report points to alternative pathways that can help find solutions to multiple societal challenges, where stabilizing the climate is one of them.

The IPBES Global Assessment is recent (2019a), but timely, since it will inform the preparation of the biodiversity conservation and sustainable use strategy 2021–2030 that will be adopted by the 15th Conference of the Parties (COP) of the Convention on Biological Diversity (CBD) in October 2020. Similarly to the IPCC reports on the need to stabilize the Earth's climate system, the IPBES assessments indicate the urgency to halt the loss of biodiversity and the declines in the multiple ecosystem services generated by nature. Further, the IPBES assessments show that the declining trends of biodiversity will most likely hamper the achievement of the Sustainable Development Goals (SDG) of the Agenda 2030.

Some key messages point to the need of policy interventions that are harmonized across sectors, i.e. that explicitly search for synergies and avoid detrimental conflicts, helping to optimally resolve trade-offs, including those that may occur between climate mitigation and biodiversity conservation actions. The suggested policy options and measures need to be operationalized into measures at the national and international levels, similarly to the Klimakur 2030. Therefore, a *Naturkur* ("nature cure") would aim at implementing the Norwegian "Nature for Life" white paper (Meld. St. 14 (2015-2016), Ministry of Climate and Environment 2015), a would also be a natural follow-up of the IPBES assessments, a process which hopefully will start after the 15 COP CBD in October this year. Or rather, a harmonized *Klima-naturkur*, where actions for climate mitigation and adaptation, and for biodiversity and ecosystem services conservation are not designed independently, but address societal challenges in a coordinated manner, are synergistic and reinforce each other to achieve multiple benefits.

5 Uncertainties and perspectives

Considerable gaps remain in our understanding of carbon fluxes and storages within Norwegian ecosystems, and the data compiled in this report has relied on the limited literature on carbon within broad ecosystems. Here, we highlight some topics where uncertainties should be reduced and knowledge increased:

- A large reason for the gaps in our knowledge come from the **lack of accurate maps of habitat types** in Norway**,** which hampers land cover estimations and effective targeting of measures to either protect them, restore them and/or avoid harm, especially with regards to non-agricultural lowland ecosystems and disparities in the reporting of wetland area. The 'Trua natur' report from NINA highlights these gaps and the challenges of dealing with multiple categorizations of habitat types for effective conservation of threatened habitats (Kyrkjeeide et al. 2018). Likewise, Jakobsson et al. (2020) state that of 23 land cover datasets available, only two are "ready for direct use for categorisation of nature". Also, Venter et al. (2019) could only produce accurate maps of two categories of 'open lowland' habitats. To date, the most comprehensive – i.e. with complete area coverage – and consistent assessment of land cover is based on vegetation types (Bryn et al. 2018), and largely supports the broad classifications of habitats as reported by Statistics Norway (2019a). What Bryn et al. (2018) highlight is the variety of classes within habitat groups, and they emphasise the disparities in theirs and previous estimations (including topographical maps) of wetland cover for the country. Norway already produces annual data on land cover (Statistics Norway 2019a), but this is prioritised to represent managed ecosystems (e.g. productive forests and agriculture) and built-up areas; which most likely underestimates wetland cover by as much as 20,000 km^2 (see Statistics Norway 2019 vs. Bryn et al. 2018); and classifies all alpine and non-agricultural lowland (i.e. grasslands and heaths) under the umbrella of 'open firm ground'. As we demonstrate, nuances within ecosystems, such as the carbon difference between low and high alpine, mean that such broad classifications of land types are not reflective of the true portfolio of Norwegian habitats (**Table 6**): regular true land cover inventory is required for a more accurate evaluation of Norway's ecosystem carbon capital, as well as for the design of measures to maintain and enhance it that are aligned with biodiversity and ecosystem services conservation priorities.

- The **carbon stocks above- and below-ground** in this report are mainly based on studies done outside of Norway. Also, some habitat types within the ecosystems are not reported at all, including peatland forest (covers ca. 13,000 km^2, Bryn et al. 2018), nival alpine zones and the majority of open lowland habitats such as grasslands, heaths and meadows (see **Table 6**).

- **Carbon fluxes** in the assessed ecosystems are particularly variable for those ecosystems that are well studied (i.e. forests), and uncertain in others with few studies available for alpine and cryospheric habitats, for example, whereas storage estimates are more consistent in their coverage of land types. Research into the **carbon flux and storage** of Norwegian ecosystems is at best in its early stages, with active projects currently in the process of gathering essential information on grasslands (NIBIO: 'Carbon storage in long and short term grasslands'), alpine fjordlands (University of Bergen: 'FunCaB') and forests (NINA: 'ForBioFunCtioN') for example. In the meantime, little data exists on Norwegian examples of carbon sediments in lakes, saltmarshes and fjords. However, the comparability of some of these ecosystems to Norwegian counterparts is unclear. Similarly, data for peatlands and permafrost is disputable and at times is based on measurements taken three decades ago in temperate systems (drained peatland estimates from Armentano & Menges 1986, in de Wit et al. 2015). Furthermore, due to the variance in area estimates and variable depths of both peatlands and permafrost, our calculations are likely an underestimation of actual carbon stocks.

- The consequences for **carbon dynamics of various types of land management** and land-use change, together with other drivers of ecological change, need to be further elucidated in order to target actions that will be effective in reducing GHG emissions and enhancing removals at the same time that they can deliver several co-benefits, hence helping achieving multiple goals, as those formulated under the Sustainable Development Goals (UN, Agenda 2030). The potentially high carbon storage capacity in Norwegian ecosystems should be considered when implementing actions mitigating carbon emissions (e.g. including the impact resulting from habitat destruction of wind farms and other infrastructure on carbon emissions; Nayak et al. 2008).

- The interactions between the **carbon and nitrogen cycles** are not considered and may play a large role in emissions and land use change scenarios. Agricultural land use changes may result in an increase of N_2O release (a greenhouse gas). When excess nitrogen fertilisation pollutes waterways, land use changes may also disrupt ecosystems and change nutrient flows (e.g. Conley et al. 2009). Such effects and interactions between the nitrogen and carbon cycles are little explored outside of agricultural ecosystems (e.g. Ergon et al. 2016, Russenes et al. 2019). The same concerns CH_4. Few studies examine the release of CH_4 from systems in Norway, which is likely a significant source of atmospheric methane, given its latitude and abundance of wetlands and permafrost. This has been examined in detail in sub-Arctic Sweden, where permafrost coverage is diminishing under climate change. Likewise, this is observed in palsa mire ecosystems in Norway (e.g. Hofgaard & Myklebost 2019). Studies find that permafrost degradation has led to as much as a 66% increase in CH_4 emissions since 1970, and dramatic changes to the peatland ecosystems (Christensen et al. 2004). Furthermore, N_2O release from thawing permafrost, and changes to microbial communities, particularly in wetlands and marshes under climate change, will result in further positive feedbacks to climate change (Eberling et al. 2010). This area of study will be of particular importance to Norway, given that so much land mass is taken by either tundra, which will be susceptible to N_2O, CO_2 and CH_4 release under climate change, or managed ecosystems such as forestry lands that greatly alter natural biogeochemical flows.

- Major gaps remain regarding **the role of biodiversity in mediating carbon stocks and fluxes**, and how changes in species composition, species diversity and trophic networks affect carbon inputs and storage in both above- and below-ground biomass, and in the soil. Recent evidence underscores the significance of biodiversity in maintaining ecosystem functioning, and thereby, highlights the importance of considering with caution the poorly understood consequences of ecosystem homogenization and simplification. There is evidence that e.g. forest understorey can have very significant impacts on carbon dynamics and pools (Wardle et al. 2012) and that also the diversity and structure of soil organism networks are critical in determining carbon fluxes and stocks (Morriën et al. 2017). A recent study applying methods from ecosystem ecology on data from a large grassland biodiversity experiment shows that higher diversity of plants and other trophic groups above- and below-ground resulted in more energy stored, greater energy flow and higher community-energy-use efficiency across the entire trophic network, and conclude that trophic levels jointly increased the performance of the community, indicating ecosystem-wide multitrophic complementarity, which is potentially an important prerequisite for the provisioning of ecosystem services (Buzhdygan et al. 2020). Despite the lack of studies for Norway, the importance of biodiversity in these processes cannot be neglected because the outcome of interventions aimed to enhance atmospheric carbon removals may not have the intended effects or, in the worst case, have opposite effects than those intended.

- **Table 6** highlights where the **information is currently lacking in terms of carbon budgets**, and what we have been able to estimate: For example, literature can help us to identify the carbon storage capacity of forest soils, and their primary production, but we lack the data to estimate respiration. Where more evidence is available (e.g. alpine zones), it relies on local studies of discrete habitats patches which are then extrapolated to a national scale (e.g. Sørensen et al. 2017, Strimbeck et al. 2019).

Table 6. *Estimates of ecosystem areas and carbon values for: annual primary production; respiration; net flux; burial; export; standing stocks/storage and the variability in estimations where data sources differ.*

Habitat type	Area estimation used (km^2)	Area estimation alternatives (km^2)	Primary production/ assimilation (Gg C yr^{-1})	Respiration/ emission (Gg C yr^{-1})	Net flux (Gg C yr^{-1})	Burial (Gg C yr^{-1})	Export (Gg C yr^{-1})	Storage (Gg C)
CRYO-SPHERE	**15,700**							**750,025**
Glaciers	2700 [a]		0.19 [b]	0.05 [b]	0.13 [b]		0.25 [b]	25 [b]
Permafrost	13,000 [c]		*NA*	16,000 [d]				750,000 [e]
ALPINE	**104,000** [f]	110,000[g]						**708,000 – 2,420,000 [h,i]**
Nival	19,500 [f]		> 0.8 [h]	> 0.6 [h]	> 0.2 [h]			0.01 – 90,000 [h,i]
Shrub	39,000 [f]		5500 [h]	3100 [h]	2400 [h]			256,000 [h]
Heath	38,000 [f]		3000 [h]	650 [h]	2350 [h]			351,000 [h]
Meadow	8300 [f]		925 [h]	500 [h]	425 [h]			101,000 [h]
FORESTS	**121,000** [k]	142,560 [l]						**1,655,750 – 2,829,000**
Forest soil			1064 – 1862 [k,l,m]	173 [n]				1,240,250 – 1,830,000 [k,o,p,q]
Forest dead			367 [n]					60,500 – 499,000 [l,r,s]
Forests living			4840 – 5702 [k,l,m]					355,000 – 500,000 [o,p,q]
WETLANDS	**28,000** [f]	17,341 – 41,655 [k,t]						**890,002 – 2,109,582**
Undisturbed			329 – 791 [m,k,t]		200 [m]			869,817 – 2,089,397 [f,k,u]
Disturbed					−5500 [m]			20,185 [u]

a) Norwegian Water Resources and Energy Directorate (2019)
b) Anesio et al. (2009)
c) Gisnås et al. (2016)
d) Hicks et al. (2015)
e) Hugelius et al. (2014)
f) Bryn et al. (2018)
g) Austrheim et al. (2010)
h) Sørensen et al. (2017)
i) Post et al. (1982)
j) Ohtsuka et al. (2018)
k) Statistics Norway (2019a)
l) Storaunet & Framstad (2015)
m) De Wit et al. (2015)
n) Norwegian Environment Agency (2019a)
o) Pregitzer & Euskirchen (2004)
p) Tomter & Dalen (2018)
q) Søgaard et al. (2019)
r) Siitonen (2001)
s) Norwegian Environment Agency (2019b)
t) Bryn et al. (2018), including forest peatland areas
u) Grønland et al. (2010)

Habitat type	Area estimation used (km^2)	Area estimation alternatives (km^2)	Primary production/ assimilation ($Gg\ C\ yr^{-1}$)	Respiration/ emission ($Gg\ C\ yr^{-1}$)	Net flux ($Gg\ C\ yr^{-1}$)	Burial ($Gg\ C\ yr^{-1}$)	Export ($Gg\ C\ yr^{-1}$)	Storage (Gg C)
OPEN LOWLANDS	**18,500** [k,n]							**217,200 – 516,200**
Grasslands	2300 [n]							22,000 [k,n]
Croplands	9400 [k]	12,239 [f]						91,000 – 333,000 [k,n,v]
Coastal heathland	2700 [f]							79,800 [w]
Other heathland	4100 [f]							24,400 [w]
FRESHWATER	**18,000** [x]	17,789 [f] – 20,000 [k]			(−240) – (−1800)	40 – 160	500 – 1300	**200,000 – 2,000,000** [y]
Rivers					(−40) – (−190) [z]	0	500 – 1300 [z]	
Lakes			10 – 50 [z]	210 – 1650 [z]	(−200) – (−1600) [z]	40 – 160 [z]	0	200,000 – 2,000,000 [y]
MARINE	**8300 – 13,000**		3700 – 4400		3700 – 4100	280		**5000 – 22,000** [y]
Kelp, algae and seagrass	8000 [æ]		3600 [æ]		3600 [æ]	260 [æ]		4100 [y,æ,ø]
Saltmarshes and mudflats	1100 [å]	250 – 5000 [å]	100 – 650 [aa]	40 – 300 [aa]	60 – 350 [aa]	24 [bb,cc]		900 – 18,000 [y,bb]

v) Grønland et al. (2008)
w) Milne & Brown (1997)
x) Norwegian Water Resources and Energy Directorate (2018)
y) storage is predominantly in sediments
z) Algesten et al. (2003) for Sweden
æ) Gundersen et al. (2011)
ø) Röhr et al. (2018) for Skagerrak
å) "estimated guesses" in the absence of accurate habitat maps
aa) Spilmont et al. (2006) for France
bb) Chmura et al. (2003)
cc) Alonso et al. (2012)

6 References

Aaby, B. & Tauber, H. 1975. Rates of peat formation in relation to degree of humification and local environment, as shown by studies of a raised bog in Denmark. Boreas 4: 1–18.

Aarrestad, P.A., Bendiksen, E., Bjerke, J.W., Brandrud, T.E., Hofgaard, A., Rusch, G. & Stabbetorp, O.E. 2003. Effekter av treslagsskifte, treplanting og nitrogengjødsling i skog på biologisk mangfold. Kunnskapsgrunnlag for å vurdere skogtiltak i klimasammenheng. NINA rapport 959. Norsk Institutt for Naturforskning.

Achat, D., Fortin, M., Landmann, G. et al. 2015. Forest soil carbon is threatened by intensive biomass harvesting. Scientific Reports 5, 15991.

Ågren, G.I. & Hyvönen, R. 2003. Changes in carbon stores in Swedish forest soils due to increased biomass harvest and increased temperatures analysed with a semi-empirical model. Forest Ecology and Management 174: 25–37.

Alam, A., Strandman, H., Kellomäki, S. & Kilpeläinen, A. 2017. Estimating net climate impacts of timber production and utilization in fossil fuel intensive material and energy substitution. Canadian Journal of Forest Research 47: 1010–1020.

Algesten, G., Sobek, S., Bergström, A.-K., Ågren, A., Tranvik, L.J. & Jansson, M. 2003. Role of lakes for organic carbon cycling in the boreal zone. Global Change Biology 10: 141–147.

Alin, S.R. & Johnson, T.C. 2007. Carbon cycling in large lakes of the world: a synthesis of production, burial, and lake-atmosphere exchange estimates. Global Biogeochemical Cycles 21: GB3002.

Alm, J., Schulman, L., Walden, J., Nykänen, H., Martikainen, P.J. & Silvola, J. 1999. Carbon balance of a boreal bog during a year with an exceptionally dry summer. Ecology 80: 161–174.

Alonso, I., Weston, K., Gregg, R. & Morecroft, M. 2012. Carbon storage by habitat: Review of the evidence of the impacts of management decisions and condition of carbon stores and sources. Natural England Research Report 43.

Anesio, A.M., Hodson, A.J., Fritz, A., Psenner, R. & Sattler, B. 2009. High microbial activity on glaciers: importance to the global carbon cycle. Global Change Biology 15:955–960.

Angelstam, P. & Kuuluvainen, T. 2004. Boreal forest disturbance regimes, successional dynamics and landscape structures – a European perspective. Ecological Bulletins 51: 117–136.

Armentano, T.V. & Menges, E.S. 1986. Patterns of change in the carbon balance of organic soil wetlands of the temperate zone. Journal of Ecology 74: 755–774

Astrup, R., Dalsgaard, L., Eriksen, R. & Hylen, G. 2010. Utviklingsscenarioer for karbonbinding i Norges skoger. Oppdragsrapport fra Skog og landskap 16/2010. Norsk institutt for skog og landskap.

Austrheim, G., Bråthen, K.A., Ims, R.A., Mysterud, A. & Ødegaard, F. 2010. Alpine Environment. In Kålås, J.A., Henriksen, S., Skjelseth, S. & Viken, Å. (eds), Environmental conditions and impacts for Red List species. Pp. 107–117. Norwegian Biodiversity Information Centre, Trondheim.

Bakken, T.H., Aase, A.G., Hagen, D., Sundt, H., Barton, D.N. & Lujala, P. 2017. Demonstrating a new framework for the comparison of environmental impacts from small and large scale hydropower and wind projects. Journal of Environmental Management 140: 93–101.

Bárcena, T.G., Grønlund, A., Hoveid, Ø., Søgaard, G. & Lågbu, R. 2016. Kunnskapsgrunnlag om nydyrkning av myr. NIBIO Rapport 2(43). Norwegian Institute of Bioeconomy Research.

Bardgett, R.D. & van der Putten, W. 2014. Belowground biodiversity and ecosystem functioning. Nature 515: 505–511.

Barton, D.N., Ring, I. & Rusch, G.M. 2017. Policy mixes: aligning instruments for biodiversity conservation and ecosystem service provision. Environmental Policy & Governance 27: 395–499.

Battin, T.J., Luyssaert, S., Kaplan, L.A., Aufdenkampe, A.K., Richter, A. & Tranvik, L.J. 2009. The boundless carbon cycle. Nature Geoscience 2: 598–600.

Belyea, L.R. & Clymo, R.S. 2001. Feedback control of the rate of peat formation. Proceedings of the Royal Society, Biological Sciences 268: 1315–1321.

Bengtsson, F. 2019. Functional traits in Sphagnum (PhD dissertation). Uppsala. http://urn.kb.se/resolve?urn=urn:nbn:se:uu:diva-375011

Bernasconi, P., Blumentrath, S., Barton, D.N., Rusch, G.M. & Romeiro, A. 2016. Constraining forest certificate's market to improve cost-effectiveness of biodiversity conservation in São Paulo State, Brazil. Plos One 11: e0164850.

Berndes, G., Abt, B., Asikainen, A., Cowie, A., Dale, V., Egnell, G., Lindner, M., Marelli, L., Paré, D., Pingoud, K. & Yeh, S. 2016. Forest biomass, carbon neutrality and climate change mitigation. From Science to Policy 3. European Forest Institute.

Bjerke, J.W., Strann, K.-B., Skei, J.K. & Ødegaard, F. 2010. Myr-kilde-flommark. In: Nybø, S. (ed.), Naturindeks for Norge 2010. Pp. 94–108. DN-utredning 3-2010. Direktoratet for naturforvaltning.
Bradford, M.A., Warren, R.J., II, Baldrian, P. et al. 2014. Climate fails to predict wood decomposition at regional scales. Nature Climate Change 4: 625–630.
Bradshaw, C.J.A, Warkentin, I.G. & Sodhi, N.S. 2009. Urgent preservation of boreal forest carbon and biodiversity. Trends in Ecology and Evolution 24: 541–548.
Bradshaw, C.J.A. & Warkentin, I.G. 2015. Global estimates of boreal forest carbon stocks and flux. Global and Planetary Change 128: 24–30.
Brown, I. 2020. Challenges in delivering climate change policy through land use targets for afforestation and peatland restoration. Environmental Science & Policy 107: 36–45.
Brumme, R. & Beese, F. 1992. Effects of liming and nitrogen fertilization on emissions of CO_2 and N_2O from a temperate forest. Journal of Geophysical Research 97: 851–858.
Bryn, A., Strand, G.-H., Angeloff, M. & Rekdal, Y. 2018. Land cover in Norway based on an area frame survey of vegetation types. Norsk geografisk tidsskrift 72: 131–145.
Bullock, J.M., Jefferson, R.G., Blackstock, T.H., Pakeman, R.J.,Emmett, B.A., Pywell, RJ., Grime, J. P. & Silvertown, J. 2011. Semi-natural grasslands. In: The UK National Ecosystem Assessment Technical Report. Pp. 161–195. NEP-WCMC, Cambridge.
Büntgen, U., Krusic, P.J., Piermattei, A. et al. 2019. Limited capacity of tree growth to mitigate the global greenhouse effect under predicted warming. Nature Communications 10: 2171.
Buzhdygan, O.Y., Meyer, S.T., Weisser, W.W. et al. 2020. Biodiversity increases multitrophic energy use efficiency, flow and storage in grasslands. Nature Ecology and Evolution 4: 393–405.
CAFF. 2001. Arctic Flora and Fauna: status and conservation. Https://www.arcticcentre.org/EN/arcticregion/Maps/definitions. Accessed 3rd February 2020.
Cahoon, S.M.P., Sullivan, P.F., Shaver, G.R., Welker, J.M. & Post, E. 2012. Interactions among shrub cover and the soil microclimate may determine future Arctic carbon budgets. Ecology Letters 15: 1415–1422.
Ceballos, G., Ehrlich, P.R., Barnosky, A.D., García, A., Pringle, R.M. & Palmer, T.M. 2015. Accelerated modern human-induced species losses: Entering the sixth mass extinction. Science Advances 1: e1400253.
Chmura, G.L., Anisfeld, S.C., Cahoon, D.R. & Lynch, J.C. 2003. Global carbon sequestration in tidal, saline wetland soils. Global Biogeochemical Cycles 17: 1111.
Christensen, T.R., Johansson, T., Åkerman, J.H., Mastepanov, M., Malmer, N., Friborg, T., Crill, P. & Svensson, B.H. 2004. Thawing sub-arctic permafrost: effects on vegetation and methane emissions. Geophysical Research Letters 31: L04501.
Ciais, P., Sabine, C., Bala, G. et al. 2013. Carbon and other biogeochemical cycles. In IPCC (ed.), Climate change 2013: the physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Pp. 465–570. Cambridge University Press, Cambridge.
CICERO. 2017. The carbon budget for dummies. Https://cicero.oslo.no/en/carbonbudget-for-dummies. Accessed 31st January 2020
Claesson, S., Duvemo, K., Lundström, A. & Wikberg, P.E. 2015. Skogliga konsekvensanalyser – SKA 2015. Rapport 10/2015. Skogsstyrelsen.
Clemmensen, K.E., Bahr, A., Ovaskainen, O., Dahlberg, A., Ekblad, A., Wallander, H., Stenlid, J., Finlay, R.D., Wardle, D.A. & Lindahl, B.D. 2013. Roots and associated fungi drive long-term carbon sequestration in boreal forest. Science 339: 1615–1618.
Clemmensen, K.E., Finlay, R.D., Dahlberg, A., Stenlid, J., Wardle, D.A. & Lindahl, B.D. 2015. Carbon sequestration is related to mycorrhizal fungal community shifts during long-term succession in boreal forests. New Phytologist 205: 1525–1536.
Conley, D.J., Paerl, H.W., Howarth, R.W. et al. 2009. Controlling eutrophication: Nitrogen and phosphorus. Science 323: 1014–1015.
Cyr, H. & Peters, R.H. 1996. Biomass-size spectra and the prediction of fish biomass in lakes. Canadian Journal of Fishery and Aquatic Science 53: 994–1006.
Dalen, L.S. 2017. Nye rekordtall for skogen i Norge. https://www.nibio.no/nyheter/nye-rekordtall-for-skogen-i-norge. Accessed 3rd February 2020.
Dass, P., Houlton, B.Z., Wang, Y. & Warlind, D. 2018. Grasslands may be more reliable carbon sinks than forests in California. Environmental Research Letters 13: 074027.
de Wit, H.A., Austnes, K., Hylen, G. & Dalsgaard, L. 2015. A carbon balance of Norway: terrestrial and aquatic carbon fluxes. Biogeochemistry 123:147–173

de Wit, H.A., Valinia, S., Weyhenmeyer, G.A., Futter, M.N., Kortelainen, P., Austnes, K., Hessen, D.O., Räike, A., Laudon, H. & Vuorenmaa, J. 2016. Current browning of surface waters will be further promoted by wetter climate. Environmental Science and Technology Letters 3: 430–435.
Dean, W.E. & Gorham, E. 1998. Magnitude and significance of carbon burial in lakes, reservoirs, and peatlands. Geology 26: 535–538.
Deluca, T.H. & Boisvenue, C. 2012. Boreal forest soil carbon: distribution, function and modelling. Forestry 85: 161–184.
Du, E., Xia, N. & de Vrie, W. 2019. Effects of nitrogen deposition on growing-season soil methane sink across global forest biomes. Biogeosciences Discussions: under review (doi:10.5194/bg-2019-29).
Eaton, J.M., McGoff, N.M., Byrne, K.A., Leahy, P. & Kiely, G. 2008. Land cover change and soil organic carbon stocks in the Republic of Ireland 1851–2000. Climatic Change 91: 317–334.
Eberling, B., Christiansen, H.H. & Hansen, B.U. 2010. High nitrous oxide production from thawing permafrost. Nature Geoscience 3: 332–335.
Edenhofer, O., Pichs-Madruga, R., Sokona, Y. et al. (eds). 2011. Renewable Energy Sources and Climate Change Mitigation: Special Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.
Ergon, Å., Kirwan, L., Bleken, M.A., Skjelvåg, A.O., Collins, R.P. & Rognli, O.A. 2016. Species interactions in a grassland mixture under low nitrogen fertilisation and two cutting frequencies: 1. dry-matter yield and dynamics of species composition. Grass and Forage Science 71: 667–682.
Evans, C., Morrison, R., Burden, A. et al. 2016. Final report on project SP1210: Lowland peatland systems in England and Wales – evaluating greenhouse gas fluxes and carbon balances. Centre for Ecology and Hydrology.
Evju, M., Bratli, H., Hanssen, O., Stabbetorp, O. E. & Ødegaard, F. 2015. Salt meadows – a hotspot-habitat. Final report from the third period of the ARKO project. NINA Report 1170. Norwegian Institute for Nature Research.
Falloon, P.D., Smith, P., Smith, J.U., Szabó, J., Coleman, K. & Marshall, S. 1998. Regional estimates of carbon sequestration potential: linking the Rothamsted Carbon Model to GIS databases. Biology and Fertilty of Soils 27: 236–241.
Framstad, E., de Wit, H., Mäkipää, R., Larjavaara, M., Vesterdal, L. & Karltun, E. 2013. Biodiversity, carbon storage and dynamics of old northern forests. TemaNord 2013:507. Nordic Council of Ministers.
Framstad, E. (ed.). 2015. Naturindeks for Norge 2015. Tilstand og utvikling for biologisk mangfold. Rapport M-441|2015. Norwegian Environment Agency (Miljødirektoratet).
Friedlingstein, P., Jones, M.W., O'Sullivan, M. et al. 2019. Global carbon budget 2019. Earth Systems Science Data 11: 1783–1838.
Gallego-Sala, A.V., Charman, D.J., Brewer, S. et al. 2018. Latitudinal limits to the predicted increase of the peatland carbon sink with warming. Nature Climate Change 8:907–913.
Gisnås K., Etzelmuller, B., Lussana, C., Hjort, J., Sannel, A.K., Isaksen, K., Westermann, S., Kuhry, P., Christiansen, H.H., Frampton, A. & Åkerman, J. 2016. Permafrost map for Norway, Sweden and Finland. Permafrost and Periglacial Processes 28: 359–378.
Gleixner, G., Tefs, C., Jordan, A., Hammer, M., Wirth, C., Nueske, A., Telz, A., Schmidt, U.E. & Glatzel, S. 2009. Soil Carbon Accumulation in Old-Growth Forests. In: Wirth, C., Gleixner, G. & Heimann, M. (eds), Old Growth Forests. Function, Fate and Value. Pp. 231–266. Springer, Berlin.
Gobin, A., Campling, P., Janssen, L., Desmet, N., van Delden, H., Hurkens, J., Lavelle, P. & Berman, S. 2011. Soil organic matter management across the EU – best practices, constraints and trade-offs. Final Report for the European Commission's DG Environment, September 2011.
Gómez-Baggethun, E. & Barton, D.N. 2013. Classifying and valuing ecosystem services for urban planning. Ecological Economics 86: 235–245.
Gonzalez-Domingues, B., Niklaus, P.A., Studer, M.S., Hagedorn, F., Wacker, L., Haghipour, N., Zimmermann, S., Walthert, C., McIntyre, C. & Abiven, S. 2019. Temperature and moisture are minor drivers of regional-scale soil organic carbon dynamics. Scientific Reports 9: 6422.
Government. 2014. Forestry. Https://www.regjeringen.no/en/topics/food-fisheries-and-agriculture/skogbruk/innsikt/skogbruk/id2009516/. Accessed 31st January 2020
Government. 2016. Norway has ratified the Paris Agreement. https://www.regjeringen.no/en/aktuelt/norge-har-ratifisert-parisavtalen/id2505365/. Accessed 30th January 2020
GRIDA. 2015. Worlds biomes and carbon storage. Https://www.grida.no/resources/6940. Accessed 31st January 2020.

Griscom, B.W., Adams, J., Ellis, P.W. et al. 2017. Natural climate solutions. Proceedings of the National Academy of Sciences of the United States of America 114: 11645–11650.
Grønlund, A., Hauge, A., Hovde, A. & Rasse, D.P. 2008. Carbon loss estimates from cultivated peat soils in Norway: a comparison of three methods. Nutrient Cycling in Agrecosystems 81: 157–167.
Grønlund A., Bjørkelo K., Hylen G. & Tomter S.M. 2010. CO_2-opptak i jord og vegetasjon i Norge. Lagring, opptak og utslipp av CO2 og andre klimagasser. Bioforsk Rapport 5(162). Bioforsk.
Gundersen, H., Bekkby, T., Norderhaug, K.M., Oug, E. & Fredriksen, S. 2018a. Marin undervannseng. In Norwegian Biodiversity Information Centre (Artsdatabanken) (ed.), Norsk rødliste for naturtyper 2018. Https://artsdatabanken.no/RLN2018/18. Accessed 28 January 2020.
Gundersen, H., Bekkby, T., Norderhaug, K.M., Oug, E., Rinde, E. & Fredriksen, S. 2018b. Stortareskog i Norskehavet og Barentshavet. In Norwegian Biodiversity Information Centre (Artsdatabanken) (ed.), Norsk rødliste for naturtyper 2018. Https://artsdatabanken.no/RLN2018/343. Accessed 28 January 2020.
Gundersen, H., Christie, H.C., de Wit, H., Norderhaug, K.M., Bekkby, T. & Walday, M.G. 2011. Utredning om CO2-opptak i marine naturtyper. NIVA Rapport 6070. Norsk institutt for vannforskning.
Gundersen, H., Norderhaug, K.M., Rinde, E., Johnsen, T.M., van der Meeren, G., Nilssen, K.T. & Lorentsen, S.H. 2015. Kyst. In Framstad, E. (ed.), Naturindeks for Norge 2015. Tilstand og utvikling for biologisk mangfold. Pp. 50–58. Rapport M-441|2015. Norwegian Environment Agency (Miljødirektoratet).
Halme, P., Allen, K.A., Auniņš, A. et al. 2013. Challenges of ecological restoration: Lessons from forests in northern Europe. Biological Conservation 167: 248–256.
Hall, A. 2004. The role of surface albedo feedback in climate. Journal of Climate 17: 1550–1568.
Hanssen-Bauer, I., Førland, E.J., Haddeland, I., Hisdal, H., Lawrence, D., Mayer, S., Nesje, A., Nilsen, J.E.Ø., Sandven, S., Sandø, A.B., Sorteberg, A. & Ådlandsvik, B. 2017. Climate in Norway 2100 – a knowledge base for climate adaptation. Report nr 1/2017, M741. Norwegian Environment Agency.
Haugland, H., Backer, E.B., Løbersli, E.M. et al. 2014. Målrettet gjødsling av skog som klimatiltak. Egnede arealer og miljøkriterier. Rapport M174|2014. Norwegian Environment Agency (Miljødirektoratet).
Heathcote, A.J., Anderson, N.J., Prairie, Y.T., Engstrom, D.R. & del Giorgio, P.A. 2015. Large increases in carbon burial in northern lakes during the Anthropocene. Nature Communications 6: 10016.
Henriksen, S. & Hilmo, O. 2015. The Norwegian Red List of species. Norwegian Biodiversity Information Centre (Artsdatabanken), Trondheim.
Herrmann, S., Kahl, T. & Bauhus, J. 2015. Decomposition dynamics of coarse woody debris of three important central European tree species. Forest Ecosystems 2: 27.
Hicks Pries, C.E., van Logtestijn, R.S.P., Schuur, E.A.G., Natali, S.M., Cornelissen, J.H.C., Aerts, R. & Dorrepaal, E. 2015. Decadal warming causes a consistent and persistent shift from heterotrophic to autotrophic respiration in contrasting permafrost ecosystems. Global Change Biology 21: 4508–4519.
Hood, E., Battin, T.J., Fellman, J., O'Neel, S. & Spencer, R.G.M. 2015. Storage and release of organic carbon from glaciers and ice sheets. Nature Geoscience 8: 91–96.
Hopkins, D.W., Waite, I.S., McNicol, J.W., Poulton, P.R., Macdonald, A.J. & O'Donnell AG. 2009. Soil organic carbon contents in long-term experimental grassland plots in the UK (Palace Leas and Park Grass) have not changed consistently in recent decades. Global Change Biology 15: 1739–1754.
Hugelius, G., Strauss, J., Zubrzycki, S. et al. 2014. Estimated stocks of circumpolar permafrost carbon with quantified uncertainty ranges and identified data gaps. Biogeosciences 11: 6573–6593.
Hughes, A.R, Inouye, B.D, Johnson, M.T.J., Underwood, N. & Vellend, M. 2008. Ecological consequences of genetic diversity. Ecology Letters 11: 609–623.
Iordan, C., Hu, X., Arvesen, A., Kauppi, P. & Cherubini, F. 2018. Contribution of forest wood products to negative emissions: historical comparative analysis from 1960 to 2015 in Norway, Sweden and Finland. Carbon Balance and Management 13: 12.
IPBES. 2019a. Summary for policymakers of the global assessment report on biodiversity and ecosystem services. Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn.
IPBES. 2019b. Land degradation and restoration. Knowledge gaps and needs. Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn.

IPBES. 2019c. Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES Secretariat, Bonn.

IPCC. 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Intergovernmental Panel on Climate Change.

IPCC. 2018. Summary for Policymakers. In Masson-Delmotte, V., Zhai, P., Pörtner, H.-O. et al. (eds), Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change, sustainable development, and efforts to eradicate poverty. Pp. 3–24. Intergovernmental Panel on Climate Change, Genève.

IPCC. 2019. Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems. Https://www.ipcc.ch/srccl/

IPCC, SROOC. 2019. Summary for Policymakers. In Pörtner, H.-O., Roberts, D.C., Masson-Delmotte, V. et al. (eds), IPCC Special Report on the Ocean and Cryosphere in a Changing Climate. In press. Available at: https://www.ipcc.ch/srocc/chapter/summary-for-policymakers/. Accessed on 21st February 2020.

IPCC Wetlands Supplement. 2014. 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands. IPCC, Gèneve.

Jactel, H., Koricheva, J. & Castagneyrol, B. 2019. Responses of forest insect pests to climate change: not so simple. Current Opinion in Insect Science 35: 103–108.

Jakobsson, S., Bakkestuen, V., Barton, D.N., Lindhjem, H. & Magnussen, K. 2020. Utredning av tilgjengelige og relevante datagrunnlag for kategorisering av naturareal. NINA Rapport 1767. Norwegian Institute for Nature Research.

Janowiak, M., Connelly, W.J., Dante-Wood, K., et. al. 2017. Considering forest and grassland carbon in land management. General Technical Report WO-95. United States Department of Agriculture, Forest Service.

Järveoja, J., Peichl, M., Maddison, M., Soosaar, K., Vellak, K., Karofeld, E., Teemusk, A. & Mander, Ü. 2016. Impact of water table level on annual carbon and greenhouse gas balances of a restored peat extraction area. Biogeosciences 13: 2637–2651.

Johansen, L., Hovstad, K.A. & Åstrom, J. 2015. Åpent lavland. In Framstad, E. (ed.), Naturindeks for Norge 2015. Tilstand og utvikling for biologisk mangfold. Pp. 92–102. Rapport M-441|2015. Norwegian Environment Agency (Miljødirektoratet).

Joosten, H., Barthelmes, A., Couwenberg, J., Hassel, K., Moen, A., Tegetmeyer, C. & Lyngstad, A. 2015. Metoder for å beregne endring i klimagassutslipp ved restaurering av myr. Naturhistorisk rapport 2015-10. NTNU Vitenskapsmuseet.

Kaltenborn, B.P., Andersen, O. & Gundersen, V. 2014. The role of wild reindeer as a flagship species in new management models in Norway. Norsk geografisk tidsskrift. 68: 168–177.

Kirschbaum, M.U.F., Zeng, G., Ximenes, F., Giltrap, G.L. & Zeldis, J.R. 2019. Towards a more complete quantification of the global carbon cycle. Biogeosciences 16: 831–846.

Körner, C. 2003 Alpine Plant Life: Functional plant ecology of high mountain ecosystems. Springer, Berlin.

Kosmo, J. 2007. Riksrevisjonens undersøkelse av bærekraftig arealplanlegging og arealdisponering i Norge. Dokument nr. 3:11 (2006–2007). Riksrevisjonen.

Kuuluvainen, T. 2009. Forest Management and Biodiversity Conservation Based on Natural Ecosystem Dynamics in Northern Europe: The Complexity Challenge. Ambio 38: 309–315.

Kyrkjeeide, M.O., Pedersen, B., Magnussen, K., Handberg, Ø.N., Evju, M, Øien, D.I., Myklebost, H.E., Aalberg Haugen, I.M., Jackson, C. & Thomassen, J. 2018. Actions to conserve threatened nature. NINA Report 1554. Norwegian Institute for Nature Research.

Laffoley, D.d'A. & Grimsditch, G. (eds). 2009. The management of natural coastal carbon sinks. International Union for Conservation of Nature and Natural Resources, Gland.

Lange, M., Eisenhauer, N., Sierra, C.A. et al. 2015. Plant diversity increases soil microbial activity and soil carbon storage. Nature Communications 6: 6707.

Larsen, S., Andersen, T. & Hessen, D.O. 2011. Climate change predicted to cause severe increase of organic carbon in lakes. Global Change Biology 17: 1186–1192.

Leskinen, P., Cardellini, G., González-García, S., Hurmekoski, E., Sathre, R., Seppälä, J., Smyth, C:, Stern, T. & Verkerk, P.J. 2018. Substitution effects of wood-based products in climate change mitigation. From Science to Policy 7. European Forest Institute.

Li, J., Delgado-Baquerizo, M., Wang, J.-T., Hu, H.-W., Cai, Z.-J., Zhu, Y.-N., Singh, B.-K. 2019. Fungal richness contributes to multifunctionality in boreal forest soil. Soil Biology and Biochemistry 136: 107526,

Lillesand, V.F., Hagen, D., Michelsen, O., Foldvik, A. & Barton, D.N. 2017. Comparing land use impacts using ecosystem quality, biogenic carbon emissions, and restoration costs in a case study of hydropower plants in Norway. International Journal of Life Cycle Assessment 22: 1384–1396.

Liski, J., Pussinen, A., Pingoud, K., Mäkipää, R. & Karjalainen, T. 2001. Which rotation length is favourable to carbon sequestration? Canadian Journal of Forest Research 31: 2004–2013.

Liski, J., Ilvesniemi, H., Mäkelä, A. & Starr, M. 2003. Model analysis of the effects of soil age, fires and harvesting on the carbon storage of boreal forest soils. European Journal of Soil Science 49: 407–416.

Liu, X., Trogisch, S., He, J.-S. et al. 2018. Tree species richness increases ecosystem carbon storage in subtropical forests. Proceedings of the Royal Society, Biological Sciences 285: 1–9.

Lundmark, T., Poudel, B.C., Stål, G., Nordin, A. & Sonesson, J. 2013. Carbon balance in production forestry in relation to rotation length. Canadian Journal of Forest Research 48: 672–678.

Lundmark, T., Bergh, J., Nordin, A., Fahlvik, N. & Poudel, B.C. 2016. Comparison of carbon balances between continuous-cover and clear-cut forestry in Sweden. Ambio 45: 203–213.

Luyssaert, S., Schulze, E.D., Börner, A. et al. 2008. Old-growth forests as global carbon sinks. Nature 455: 213–215.

Lyngstad, A., Moen, A. & Øien, D.-I. 2018. Eksentrisk høymyr. In Norwegian Biodiversity Information Centre (Artsdatabanken) (ed.), Norsk rødliste for naturtyper 2018. Https://artsdatabanken.no/RLN2018/146. Accessed 28th January 2020.

Mace, G.M., Barrett, M., Burgess, N.D., Cornell, S.E., Freeman, R., Grooten, M. & Purvis, A. 2018. Aiming higher to bend the curve of biodiversity loss. Nature Sustainability 1: 448–451.

Magnussen, K., Bjerke, J.W., Brattland, C., Nybø, S. & Vermaat, J. 2018. Verdien av økosystemtjenester fra våtmark. Menon-publikasjon 42/2018. Menon Economics.

Magnussen, K., Wifstad, K., Seeberg, A.R., Stålhammar, K, Bakken, S.E., Banach, A., Hagen, D., Rusch, G.M., Aarrestad, P.A., Løset, F. & Sandsbråten, K. 2017. Naturbaserte løsninger for klimatilpasning. Menon-publikasjon 61/2017. Menon Economics.

Maljanen, M., Sigurdsson, B.B., Gudmundsson, J., Oskarsson, H., Huttunen, J.T. & Martikainen, P.J. 2010. Greenhouse gas balances of managed peatlands in the Nordic countries – present knowledge and gaps. Biogeosciences. 7: 2711–2738

Mäkinen, H., Hynynen, J., Siitonen, J. & Sievänen, R. 2006. Predicting the decomposition of Scots pine, Norway spruce, and birch stems in Finland. Ecological Applications 16: 1865–1879.

Måren, I.E. & Nilsen, L.S. 2008. Coastal heathlands of Central to Northern Norway. Blyttia 66: 11–22

Mekonnen, Z.A., Riley, W. & Grant, R.F. 2018. 21st century tundra shrubification could enhance net carbon uptake of North American Arctic tundra under an RCP8.5 climate trajectory. Environmental Research Letters 13: aabf28.

Mendonça, R., Barros, N., Vidal, L.O., Pacheco, F., Kosten, S. & Roland, F. 2012a. Greenhouse gas emissions from hydroelectric reservoirs: what knowledge do we have and what is lacking? In Liu G. (ed.), Greenhouse Gases – Emission, Measurement and Management. http://www.intechopen.com/books/greenhouse-gases-emission-measurement-and-management/greenhouse-gas-emissions-from-hydroelectric-reservoirs-what-do-we-have-and-what-is-lacking-

Mendonça, R., Kosten, S., Sobek, S. et al. 2012b. Hydroelectric carbon sequestration. Nature Geoscience 5: 838–840.

Midolo, G., Alkemade, R., Schipper, A.M., Benítez-López, A., Perring, M.P. & De Vries, W. 2019. Impacts of nitrogen addition on plant species richness and abundance: A global meta-analysis. Global Ecology and Biogeography 28: 398–413.

Miller, P.A. & Smith, B. 2012. Modeling tundra vegetation response to recent Arctic warming. Ambio 41: 281–291.

Milne, R. & Brown, T.A. 1997. Carbon in the Vegetation and Soils of Great Britain. Journal of Environmental Management 49: 413–433.

Ministry of Climate and Environment (Klima og miljødepartementet). 2015. Nature for life — Norway's national biodiversity action plan. Melding til Stortinget nr. 14 (2015–2016).

Minkkinen, K., Korhonen, R., Savolainen, I. & Laine, J. 2002. Carbon balance and radiative forcing of Finnish peatlands 1900–2100 – the impact of forestry drainage. Global Change Biology 3: 785–799.

Moen, A., Lyngstad, A. & Øien, D.-I. 2011. Faglig grunnlag til handlingsplan for høgmyr i innlandet (typisk høgmyr). Botanisk rapport 2011-3. NTNU Vitenskapsmuseet.

Monteux, S., Weedon, J.T., Blume-Werry, G., Gavazov, K., Jassey, V.E.J., Johansson, M., Keuper, F., Olid, C. & Dorrepaal, E. 2018. Long-term in situ permafrost thaw effects on bacterial communities and potential aerobic respiration. ISME Journal 12: 2129–2141.

Morriën, E., Hannula, S.E., Snoek, L.B. et al. 2017. Soil networks become more connected and take up more carbon as nature restoration progresses. Nature Communications 8: 14349.

Mosier, S.L., Kane, E.S., Richter, D.L., Lilleskov, E.A., Jurgensen, M.F., Burton, A.J. & Resh, S.C. 2017. Interactive effects of climate change and fungal communities on wood-derived carbon in forest soils. Soil Biology and Biochemistry 115: 297–309.

Munroe, J. & Brencher, Q. 2019. Holocene carbon burial in lakes of the Uinta Mountains, Utah, USA. Quaternary 2: 13.

Mysterud, A. & Austrheim, G. 2008. The effect of domestic sheep on forage plants of wild reindeer; a landscape scale experiment. European Journal of Wildlife Research 54: 461–468.

Nayak, D.R., Miller, D., Nolan, A., Smith, P. & Smith, J. 2008. Wind farms on Scottish peat lands: a new approach. https://www.gov.scot/publications/calculating-carbon-savings-wind-farms-scottish-peat-lands-new-approach/pages/0/

Nellemann, C., Corcoran, E., Duarte, C.M., Valdés, L., De Young, C., Fonseca, L. & Grimsditch, G. (eds). 2009. Blue carbon: a rapid response assessment. United Nations Environment Programme, Nairobi.

Niemelä, T., Wallenius, T. & Kotiranta, H. 2002. The kelo tree, a vanishing substrate of specified wood-inhabiting fungi. Polish Botanical Journal 47: 91–101.

Nordén, B. & Olsen, S.L. 2017. TransForest: Edelløvskog, gjengroing og naturgoder. Norsk Skogbruk 9: 38–39.

Nordén, J., Penttilä, R., Siitonen, J., Tomppo, E. & Ovaskainen, O. 2013. Specialist species of wood-inhabiting fungi struggle while generalists thrive in fragmented boreal forests. Journal of Ecology 101: 701–712.

Nordén, J., Åström, J., Josefsson, T., Blumentrath, S., Ovaskainen, O., Sverdrup-Thygeson, A. & Nordén, B. 2018. At which spatial and temporal scales can fungi indicate habitat connectivity? Ecological Indicators 91: 138–148.

Norderhaug, A., & Johansen, L. 2011. Norwegian Red List for Ecosystems and Habitat Types. The 2011 Norwegian Red List for Ecosystems and Habitat Types. Norwegian Biodiversity Information Centre, Trondheim, Norway.

Norderhaug, A. & Svalheim, E. 2009. Faglig grunnlag for handlingsplan for trua naturtype: Slåttemark i Norge. Bioforsk Rapport 4 (57). Bioforsk.

Nord-Larsen, T., Vesterdal, L., Bentsen, N.S. & Larsen, J.B. 2019. Ecosystem carbon stocks and their temporal resilience in a semi-natural beech-dominated forest. Forest Ecology and Management: 447: 67–76.

Norton, M., Baldi, A., Buda, V. et al. 2019. Serious mismatches continue between science and policy in forest bioenergy. GCB Bioenergy 11: 1256–1263.

Norwegian Biodiversity Information Centre (Artsdatabanken). 2018. Natur i Norge. https://artsdatabanken.no/NiN.

Norwegian Environment Agency (Miljødirektoratet). 2016. Plan for restaurering av våtmark i Norge (2016-2020). Rapport M-644. Norwegian Environment Agency and Norwegian Agriculture Agency (Miljødirektoratet & Landbruksdirektoratet).

Norwegian Environment Agency (Miljødirektoratet). 2018. Marine and Coastal waters. Https://www.environment.no/topics/marine-and-coastal-waters/. Accessed 4th February 2020.

Norwegian Environment Agency (Miljødirektoratet). 2019a. Greenhouse Gas Emissions 1990-2017, National Inventory Report. Rapport M-1271.

Norwegian Environment Agency (Miljødirektoratet). 2019b. Skogvernet øker – fem prosent av skogen er vernet. https://www.miljodirektoratet.no/aktuelt/nyheter/2019/desember-2019/skogvernet-oker--fem-prosent-av-skogen-er-vernet/. Accessed 30th January 2020.

Norwegian Environmental Agency (Miljødirektoratet). 2020. Tiltak og virkemidler mot 2030. Rapport M-1625.

Norwegian Institute of Bioeconomy Research (NIBIO). 2017. Sau per km^2. In Norsk Institutt for Bioøkonomi (ed.), Kilden - Arealinformasjon. Https://kilden.nibio.no.

Norwegian Ministries. 2018. National Pollinator Strategy. A strategy for viable populations of wild bees and other pollinating insects. Ministry of Agriculture and Food, Ministry of Climate and Environment, Ministry of Local Government and Modernization, Ministry of Transport and

Communications, Ministry of Defence, Ministry of Research and Higher Education, Ministry of Education and Integration & Ministry of Petroleum and Energy, Oslo.
Norwegian Water Resources and Energy Directorate (Norges Vassdrags- og Energidirektorat, NVE). 2018. Innsjødatabase. Https://www.nve.no/karttjenester/kartdata/vassdragsdata/innsjodatabase/. Accessed 21st September 2018.
Norwegian Water Resources and Energy Directorate (Norges Vassdrags- og Energidirektorat, NVE). 2019. Glaciers. Https://www.nve.no/hydrology/glaciers/. Accessed 22nd January 2020.
NOU. 2013. Naturens goder - om verdier av økosystemtjenester. Norges offentlige utredninger 2013:10. Miljøverndepartementet.
Nybakken, L., Lie, M.H., Julkunen-Tiitto, R., Asplund, J. & Ohlson, M. 2018. Fertilization changes chemical defense in needles of mature Norway spruce (*Picea abies*). Frontiers in Plant Science 9: 770.
Odland, A. 2015. Effect of latitude and mountain height on the timberline (*Betula pubescens* ssp. *czerpanovii*) elevation along the central Scandinavian mountain range. Fennia 193: 260–270.
Ohtsuka, T., Hirota, M., Zhang, X., Shimono, A., Senga, Y., Du, M., Yonemura, S., Kawashima, S. & Tang, Y. 2008. Soil organic carbon pools in alpine to nival zones along an altitudinal gradient (4400–5300 m) on the Tibetan Plateau. Polar Science 2: 277–285.
Øien, D.-I., Fandrem, M., Lyngstad, A. & Moen, A. 2017. Utfasing av torvuttak i Norge – effekter på naturmangfold og andre økosystemtjenester. Naturhistorisk rapport 2017-6. NTNU Vitenskapsmuseet.
Olsson, M.T., Erlandsson, M., Lundin, L., Nilsson, T., Nilsson, Å. & Stendahl, J. 2009. Organic carbon stocks in Swedish Podzol soils in relation to soil hydrology and other site characteristics. Silva Fennica 43: article id 207.
Olsson, P., Linder, S., Giesler, R. & Högberg, P. 2005. Fertilization of boreal forest reduces both autotrophic and heterotrophic soil respiration. Global Change Biology 11: 1745–1753.
Onarheim, K. 2018. Market and regulatory issues related to Bio-CCUS. Workshop summary. IEA Bioenergy, Task 41.
Ontl, T.A. & Schulte, L.A. 2012. Soil Carbon Storage. Nature Education Knowledge 3: 35.
Page, S.E. & Baird, A.J. 2016. Peatlands and Global Change: Response and Resilience. Annual Review of Environment and Resources. 41: 35–57.
Pan, Y., Birdsey, R.A., Fang, J. et al. 2011. A large and persistent carbon sink in the World's forests. Science 333: 988–993.
Persvingelen, E. 2019. Impacts on carbon budgets of increased use of Norwegian forest resources for energy. MSc thesis. Geophysical Institute, University of Bergen.
Peura, M., Burgas Riera, D., Eyvindson, K., Repo, A. & Mönkkönen, M. 2017. Continuous cover forestry is a cost-efficient tool to increase multifunctionality of boreal production forests in Fennoscandia. Biological Conservation 217: 104–112.
Piao, S., Ciais, P., Friedlingstein, P. et al. 2008. Net carbon dioxide losses of northern ecosystems in response to autumn warming. Nature 451: 49–52.
Post, W.M., Emanuel, W.R., Zinke, P.J. & Stangenberger, A.G. 1982. Soil carbon pools and wild life zones. Nature 298:156–159.
Pregitzer, K.S. & Euskirchen, E.S. 2004. Carbon cycling and storage in world forests: biome patterns related to forest age. Global Change Biology 10: 2052–2077.
Pukkala, T. 2017. Optimal nitrogen fertilization of boreal conifer forest. Forest Ecosystems 4: 3.
Rantala, M.V., Luoto, T.P. & Nevalainen, L. 2016. Temperature controls organic carbon sequestration in a subarctic lake. Scientific Reports 6: 34780.
Rappe-George, M.O., Hansson, L.J., Ring, E., Jansson, P.E. & Gärdenäs, A.I. 2017. Nitrogen leaching following clear-cutting and soil scarification at a Scots pine site – A modelling study of a fertilization experiment. Forest Ecology and Management 385: 281–294.
Repo, A., Tuovinen, J.-P. & Liski, J. 2015. Can we produce carbon and climate neutral forest bioenergy? GCB Bioenergy 7: 253–262.
Rinne-Garmston, K.T., Peltoniemi, K., Chen, J., Peltoniemi, M., Fritze, H. & Mäkipää, R. 2019. Carbon flux from decomposing wood and its dependency on temperature, wood N2 fixation rate, moisture and fungal composition in a Norway spruce forest. Global Change Biology 25: 1852–1867.
Rognstad, O. & Steinset, T.A. 2012. Landbruket i Norge 2011. Agriculture in Norway 2011. Statistics Norway, Oslo.
Röhr, M.E., Boström, C., Canal-Vergés, P. & Holmer, M. 2016. Blue carbon stocks in Baltic Sea eelgrass (*Zostera marina*) meadows. Biogeosciences 13: 6139–6153.

Röhr, M.E., Holmer, M., Baum, J.K. et al. 2018. Blue carbon storage capacity of temperate eelgrass (*Zostera marina*) meadows. Global Biogeochemical Cycles 32: 1457–1475.
Roulet, N.T., Lafleur, P.M., Richard, P.J., Moore, T.R., Humphreys, E.R. & Bubier, J.I.L.L. 2007. Contemporary carbon balance and late Holocene carbon accumulation in a northern peatland. Global Change Biology 13: 397–411.
Rusch, G.M. 2012. Climate and ecosystem services. The potential of Norwegian ecosystems for climate mitigation and adaptation. NINA Report 791. Norwegian Institute for Nature Research.
Russenes, A.L., Korsaeth, A., Bakken, L.R. & Dorsch, P. 2019. Effects of nitrogen split application on seasonal N_2O emissions in southeast Norway. Nutrient Cycling in Agroecosystems 115: 41–56.
Rye, J.F. & Berg, N.G. 2011. The second home phenomenon and Norwegian rurality. Norsk geografisk tidsskrift 65: 126–136.
Scharlemann, J.P.W., Tanner, E.V.J., Hiederer, R. & Kapos, V. 2014. Global soil carbon: understanding and managing the largest terrestrial carbon pool. Carbon Management 5: 81–91.
Schartau, A.K., Dolmen, D., Hesthagen, T., Mjelde, M., Walseng, B., Ødegaard, F., Økland, J., Økland, K.A. & Bongard, T. 2010. Fresh water. In Kålås, J.A., Henriksen, S., Skjelseth, S. & Viken, Å. (eds), Environmental conditions and impacts for Red List species. Pp. 97–106. Norwegian Biodiversity Information Centre, Trondheim.
Schartau, A.K., Pedersen, B., van Dijk, J. & Solheim, A.L. 2015. Ferskvann. In Framstad, E. (ed.), Naturindeks for Norge 2015. Tilstand og utvikling for biologisk mangfold. Pp. 59–67. Rapport M-441|2015. Norwegian Environment Agency (Miljødirektoratet).
Schröter, M., Rusch, G.M., Barton, D.N., Blumentrath, S. & Nordén, B. 2014. Ecoystem services and opportunity costs shift spatial priorities for conserving forest biodiversity. PLoS One 9: e112557.
Seppälä, J. Heinonen, T., Pukkala, T., Kilpeläinen, A., Mattila, T., Myllyviita, T., Asikainen, A. & Peltola, H. 2019. Effect of increased wood harvesting and utilization on required greenhouse gas displacement factors of wood-based products and fuels. Journal of Environmental Management 247: 580–587.
Siitonen, J. 2001. Forest management, coarse woody debris and saproxylic organisms: Fennoscandian boreal forests as an example. Ecological Bulletins 49: 11–41.
Similä, M. & Junninen, K. (eds) 2012. Ecological restoration and management in boreal forests – best practices from Finland. Metsähallitus, Helsinki.
Skjeggedal, T., Overvåg, K. & Riseth, J.Å. 2015. Land-use planning in Norwegian mountain areas: Local development or nature protection? European Planning Studies 24: 344–363.
Skogsstyrelsen. 2020. Klimatanpassning av skogen och skogsbruket - mål och förslag på åtgärder (Climate adaptation of forest and forestry – targets and recommendations of measures). Rapport 2019/23. Skogsstyrelsen.
Snyder, C.S., Bruulsema, T.W., Jensen, T.L. & Fixen, P.E. 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agriculture, Ecosystems and Environment 133: 247–266.
Søgaard, G., Allen, M., Astrup, R. et al. 2019. Effekter av planting av skog på nye arealer. Betydning for klima, miljø og næring. NIBIO Rapport 5(3). Norwegian Institute of Bioeconomy Research.
Soimakallio, S., Saikku, L., Valsta, L. & Pingoud, K. 2016. Climate Change Mitigation Challenge for Wood Utilization—The Case of Finland. Environmental Science and Technology 50: 5127-5134.
Solheim, H., Eriksen, R. & Hietala, A. 2011. Dutch elm disease has currently a low incidence on wych elm in Norway. Forest Pathology 41: 182–188.
Sørensen, M.V., Strimbeck, R., Nystuen, K.O., Kapas, R.E., Enquist, B.J. & Graae, B.J. 2017. Draining the Pool? Carbon Storage and Fluxes in Three Alpine Plant Communities. Ecosystems 21: 316–330.
Speed, J.D.M., Martinsen, V., Mysterud, A., Mulder, J., Holand, Ø. & Austrheim, G. 2014. Long-term increase in aboveground carbon stocks following exclusion of grazers and forest establishment in an alpine ecosystem. Ecosystems 17: 1138–1150.
Spilmont, N., Davoult, D. & Migné, A. 2006. Benthic primary production during emersion: In situ measurements and potential primary production in the Seine Estuary (English Channel, France). Marine Pollution Bulletin 53: 49–55.
Statistics Norway (SSB). 2019a. Land use and land cover, 1 January 2019. Https://www.ssb.no/en/natur-og-miljo/statistikker/arealstat/aar/2019-05-27#content. Accessed 10th February 2020.
Statistics Norway (SSB). 2019b. Norway emissions to air, 14 January 2020. Available from: https://www.ssb.no/en/klimagassn. Accessed 31st January 2020.

Stendahl, J., Johansson, M.-B., Eriksson, E., Nilsson, Å. & Langvall, O. 2010. Soil organic carbon in Swedish spruce and pine forests – differences in stock levels and regional patterns. Silva Fennica 44: 5–21.

Stephenson, N., Das, A., Condit, R. et al. 2014. Rate of tree carbon accumulation increases continuously with tree size. Nature 507: 90–93.

Stivrins, N., Ozoloa, I., Galka, M. et al. 2017. Drivers of peat accumulation rate in a raised bog: impact of drainage, climate, and local vegetation composition. Mires and Peat, 19: 8.

Storaunet, K.O. & Rolstad, J. 2015. Mengde og utvikling av død ved i produktiv skog i Norge. Med basis i data fra Landsskogtakseringens 7. (1994-1998) og 10. takst (2010-2013). Oppdragsrapport fra Skog og landskap 06/2015. Norsk institutt for skog og landskap.

Strimbeck, R., Sørensen, M.V., Graae, B.J. & Lang, S.I. 2019. Functional group contributions to carbon fluxes in arctic-alpine ecosystems. Arctic, Antarctic, and Alpine Research 51: 58–68.

Svendgård-Stokke, S., Ulfeng, H., Dalsgaard, L., Eisner, S., Lågbu, R., Klakegg, O., Solbakken, E., Søgaard, G. & Strand, G.H. 2019. Utvikling av kart over organisk karbon i jord Norge. Forprosjekt. NIBIO Rapport 5(59). Norwegian Institute of Bioeconomy Research.

Sverdrup-Thygeson, A., Søgaard, G., Rusch, G.M. & Barton, D.N. 2014. Spatial overlap between environmental policy instruments and areas of high conservation value in forest. PLoS ONE 9: e115001.

Swindles, G.T., Morris, P.J., Mullan, D.J. et al. 2019. Widespread drying of European peatlands in recent centuries. Nature Geoscience 12: 922–928.

Taeroe, A., Mustapha, W.F., Stupak, I. & Raulund-Rasmussen, K. 2017. Do forests best mitigate CO2 emissions to the atmosphere by setting them aside for maximization of carbon storage or by management for fossil fuel substitution? Journal of Environmental Management 197: 117–129.

Tagesson, T., Schurgers, G., Horion, S. et al. 2020. Recent divergence in the contributions of tropical and boreal forests to the terrestrial carbon sink. Nature Ecology & Evolution 4: 202–209.

Ter-Mikaelian, M.T., Colombo, S.J. & Chen, J. 2015. The Burning Question: Does forest bioenergy reduce carbon emissions? A review of common misconceptions about forest carbon accounting. Journal of Forestry 113: 57–68.

Teodoru, C.R., Bastien, J., Bonneville, M.C., del Giorgio, P.A., Demarty, M., Garneay, M., Helie, J., Pelletier, L., Prairie, Y.T., Roulet, N.T., Strachan, I.B. & Tremblay, A. 2012. The net carbon footprint of a newly created boreal hydroelectric reservoir. Global Biogeochemical Cycles 26: GB2016.

Titterton, P., Benson, J., Thorpe, K. & Crouch, T. (2019) Carbon Audit Update Report 2019. Moors for the Future Partnership, Edale.

Tømmervik, H., Johansen, B., Høgda, K.A. & Strann, K.B. 2012. High-resolution satellite imagery for detection of tracks and vegetation damage caused by all-terrain vehicles (ATVs) in northern Norway. Land Degradation and Development 23: 43–52.

Tomter, S.M. & Dalen, L.S. (eds). 2018. Bærekraftig skogbruk i Norge. Norsk institutt for bioøkonomi.

Turetsky, M.R., Abbott, B.W., Jones, M.C. et al. 2019. Permafrost collapse is accelerating carbon release. Nature. 569: 32–34.

Turunen, J., Tomppo, E., Tolonen, K. & Reinikainen, A. 2002. Estimating carbon accumulation rates of undrained mires in Finland – application to boreal and subarctic regions. Holocene 12: 69–80.

United Nations Framework Convention on Climate Change (UNFCC). 1997. Kyoto Protocol of the United Nations Framework Convention on Climate Change. https://unfccc.int/kyoto_protocol.

United Nations Framework Convention on Climate Change (UNFCC). 2015. Paris Agreement. Https://unfccc.int/files/essential_background/convention/application/pdf/english_paris_agreement.pdf. Accessed 30th January 2020.

Upson, M.A., Burgess, P.J. & Morison J.I.L. 2016. Soil carbon changes after establishing woodland and agroforestry trees in a grazed pasture. Geoderma 283: 10–20.

van der Linden, M., Heijmans, M.M. & van Geel, B. 2014. Carbon accumulation in peat deposits from northern Sweden to northern Germany during the last millennium. The Holocene 24: 1117–1125.

van der Wal, A., Ottosson, E. & de Boer, W. 2015. Neglected role of fungal community composition in explaining variation in wood decay rates. Ecology 96: 124–133.

van Sundert, K., Horemans, J.A., Stendahl, J. & Vicca, S. 2018. The influence of soil properties and nutrients on conifer forest growth in Sweden, and the first steps in developing a nutrient availability metric. Biogeosciences 15: 3475–3496.

Vanhala, P., Repo, A. & Liski, J. 2013. Forest bioenergy at the cost of carbon sequestration? Current Opinion in Environmental Sustainability 5: 41–46.

Venter, Z., Stabbetorp, O.E., Garnåsjordet, P.A. & Aslaksen, I. 2019. Naturindeks for Norge: Åpent lavland. Metoder for utarbeiding av kart og beregning av areal. NINA Rapport 1715. Norsk institutt for naturforskning.

Vesterdal, L. & Christensen, M. 2008. The carbon pools in a Danish semi-natural forest. Ecological Bulletins 52: 113–121.

Villa, J.A. & Bernal, B. 2018. Carbon sequestration in wetlands, from science to practice: An overview of the biogeochemical process, measurements methods, ands policy framework. Ecological Engineering 114: 115–128.

Wagner, I. & Simons, A.M. 2009. Divergent norms of reactions to temperature in germination characteristics among populations of the arctic-alpine annual, *Koenigia islandica*. Arctic, Antarctic and Alpine Research 41: 388–395.

Wang, X., Olsen, L.M., Reitan, K.I. & Olsen, Y. 2012. Discharge of nutrient wastes from salmon farms: environmental effects, and potential for integrated multi-trophic aquaculture. Aquaculture Environment Interactions 2: 267–283.

Wardle, D.A., Jonsson, M., Bansal, S., Bardgett, R.D., Gundale, M.J. & Metcalfe, D.B. 2012. Linking vegetation change, carbon sequestration and biodiversity: insights from island ecosystems in a long-term natural experiment. Journal of Ecology 100: 16–30.

Were, D., Kansiime, F., Fetahi, T., Cooper, A. & Jjuuko, C. 2019. Carbon sequestration by wetlands: A critical review of enhancement measures for climate change mitigation. Earth Systems and Environment 3: 327–340.

Wessel, W.W, Tietema, A., Beier, C., Emmett, B.A., Peñuelas, J. & Riis-Nielsen, T. 2004. A Qualitative Ecosystem Assessment for Different Shrublands in Western Europe under Impact of Climate Change Ecosystems 7, 662–671.

Wong, W.K., Beldring, S., Engen-Skaugen, T., Haddeland, I. & Hisdal, H. 2011. Climate change effects on spatiotemporal patterns of hydroclimatological summer droughts in Norway. Journal of Hydrometeorology 12: 1205–1220.

Yu, Z., Loisel, J., Brosseau, D.P., Beilman, D.W. & Hunt, S.J. 2010. Global peatland dynamics since the Last Glacial Maximum. Geophysical Research Letters 37: L13402.

Zeng, D.-H., Li, L.-J., Fahey, T.J., Yu, Z.-Y., Fan, Z.-P. & Zhen, F.-S. 2010. Effects of nitrogen addition on vegetation and ecosystem carbon in a semi-arid grassland. Biogeochemistry 98: 185–193.

Zhang, T., Chen, H.Y.H. & Ruan, H. 2018. Global negative effects of nitrogen deposition on soil microbes. ISME Journal 12: 1817–1825.

Zhang, W., Miller, P.A., Smith, B., Wania, R., Koenigk, T. & Doscher, R. 2013. Tundra shrubification and tree-line advance amplify arctic climate warming: results from an individual-based dynamic vegetation model. Environmental Research Letters 8: 034023.

7 Appendix

7.1 Glossary

Ablation – The removal of material (glacial ice) through erosive processes such as radiative induced surface melt or ice calving.

Active layer – The thawing and respiring layer of permafrost.

Aeolian – Wind driven.

Aerobic – Cellular respiration taking oxygen and respiring CO_2 and water.

Anaerobic – Cellular respiration without oxygen, and respiring CO_2, water, and in the case of some microbes, CH_4.

Autotrophic – Carbon fixing organisms that create energy through light via photosynthesis.

Boreal – Cold temperature region south of the Arctic between approximately 50°N to 70°N, which is dominated by coniferous and birch forests.

BECCS – Bioenergy with Carbon Capture and Storage.

C – Carbon.

Carbon cycle – The complex series of reactions by which carbon passes through the Earth's atmosphere, biosphere, hydrosphere, and lithosphere. For example, plants remove carbon in the form of CO_2 from the atmosphere and use it to produce carbohydrates in living organisms (photosynthesis). When those organisms die, the carbon is returned to the Earth as carbon dioxide, as fossil fuels (during decay), or as inorganic compounds such as calcium carbonate (limestone).

Carbon dioxide (CO_2) – The main greenhouse gas affected directly by human activities. CO_2 also serves as the reference to compare all other greenhouse gases (carbon dioxide equivalents). The major source of CO_2 emissions is fossil fuel combustion. CO_2 emissions are also a product of forest clearing, biomass burning, and non-energy production processes such as cement production.

Carbon Dioxide Capture and Storage (CCS) – A process in which a relatively pure stream of carbon dioxide (CO_2) from industrial and energy-related sources is separated (captured), conditioned, compressed and transported to a storage location for long-term isolation from the atmosphere.

Carbon removals, emissions and storage. Removals result from the capacity of plants and soils to absorb and retain greenhouses gases from the atmosphere through the process of photosynthesis. Removals take place when plants grow or organic material builds up in soils. Emissions take place for instance when plants die and decay or when soils are disturbed so that their capacity to store is decreased. This would be the case when trees or crops are harvested, if wetlands are drained or if grasslands are ploughed (EU Climate action – Land-based emissions[11])

Carbon sequestration. Carbon removals from the atmosphere and enhanced storage. Uptake and long-term storage of carbon dioxide or other forms of carbon in a reservoir, to either mitigate or defer global warming. It can refer to, for example, carbon reservoirs in the soil or dead wood, or to land use change that enhances the soil carbon storage and contributes therefore to carbon sequestration.

Carbon sink – Any reservoir (e.g. ecosystem, vegetation, soil) that removes carbon released from some other part of the carbon cycle. For example, the atmosphere, oceans, forests and mires are major carbon sinks because much of the CO_2 produced elsewhere on the Earth ends up in these bodies.

Carbon source – Any process, activity, or mechanism that releases carbon to another part of the carbon cycle.

[11] https://ec.europa.eu/clima/policies/forests_en

Carbon stock – The absolute quantity of substance of concern (for example, carbon or a greenhouse gas) held within a reservoir at a specified time. A reservoir is a component of the climate system, other than the atmosphere, which has the capacity to store, accumulate, or release a substance of concern. For instance: vegetation, soils, oceans.

CH_4 – Methane, a potent greenhouse gas.

CO_2 – Carbon dioxide.

CO_2-e (Carbon dioxide equivalent) – A metric measure used to compare the emissions from various greenhouse gases based upon their global warming potential. The carbon dioxide equivalent for a gas is derived by multiplying the tons of the gas by the associated global warming potential. Carbon may also be used as the reference and other greenhouse gases may be converted to carbon equivalents. To convert carbon to carbon dioxide, multiply the carbon by 44/12 (the ratio of the molecular weight of carbon dioxide to carbon).

Cryoconite holes – Small freshwater ecosystems that can cover a glacier or ice sheet.

Cryosols – Soils affected by permafrost.

Cryosphere – Regions on Earth covered in snow and/ or ice, including frozen ground (permafrost), glaciers, snow cover and ice sheets.

DOC – Dissolved organic carbon.

Eutrophication – An aquatic system that has become overly enriched with nutrients (e.g. through nitrogen fertiliser run-off) leading to the excessive growth of primary producers like algae. It causes a reduction in oxygen (hypoxia) and disturbs normal ecosystem functions.

GHG – Greenhouse gases. In this context CO_2 and CH_4 – gases that cause warming by absorbing and emitting heat energy.

Heterotrophic – Respiration of saprotrophs, CO_2 released to the atmosphere.

Littoral – The near, or on-shore environment of a lake, river, or ocean.

Mass balance – The net change in a glaciers mass, measured annually. If accumulation exceeds ablation, the mass balance is positive, and vice versa.

Mire/Fen – Synonym for peatland – a wetland type dominated by peat forming plants and the incomplete decompositions of the organic matter.

N_2O – Nitrous oxide, laughing gas. Nitrogen-based fertilisers used in agriculture and forestry are a major source for this potent greenhouse gas**.**

Nival – Habitat defined as that above the snowline and/or covered in permanent snow or ice. Nival species, are those characterised by living in or growing in this region or associated with perpetual snow cover.

OM – Organic matter.

POC – Particulate organic carbon.

PP – Primary production. The sequestration of carbon by primary production, for example, photosynthesis.

ppm – Parts per million. Atmospheric CO_2 concentrations are measured in parts per million. 1 ppm equals 0.0001% and corresponds to roughly 2.1 Pg of atmospheric carbon (see **Table 1**).

RCP – Representative Concentration Pathways. Describes four different climate futures based on the volume of GHG in the atmosphere: RCP2.6, RCP4.5, RCP6 and RCP8.5. RCP's do not take into account the role of the carbon cycle, they are based only on GHG concentrations.

Saprotrophic organisms – Fungi, bacteria and animals that break down dead organic material, releasing the carbon stored in it.

SOC – Soil organic carbon

7.2 The three pillars for carbon emissions reductions in Klimakur 2030

Klimakur 2030 consists of three pillars, following the EU 2030 climate and energy framework 2021 – 2030 and measures to achieve the key target of reducing at least 40% greenhouse gas emissions from 1990 levels.

The EU emissions trading system (EU ETS, Pillar 1) is EU's key tool for reducing greenhouse gas emissions cost-effectively. It is the world's first major carbon market and remains the biggest one. It operates in all EU countries, plus Iceland, Liechtenstein and Norway, and limits emissions for more than 11,000 heavy energy-using installations such as power stations, and industrial plants and airlines operating between these countries.

Emission mitigating actions under Pillar 2 are regulated by the Effort Sharing Regulation (ESR, 2018), and address those sectors of the economy that fall outside the EU ETS, e.g. including transport, buildings, agriculture, non-ETS industry and waste. In the EU, they account for almost 60% of total domestic EU emissions. ESR sectors must reduce emissions by 30% by 2030 compared to 2005 as their contribution to the overall target of reducing emissions of at least 40% by 2030 compared to 1990.

Pillar 3 is based on emission neutrality from the management of vegetation and soils of land-systems (i.e. a null net GHG emission, resulting from source emissions and sink capture). It is founded on the United Nations Framework Convention on Climate Change (Kyoto Protocol 1997) and is based on the recognition that atmospheric CO2 can accumulate as carbon in vegetation and soils in terrestrial ecosystems and that human activities impact terrestrial carbon sinks and stocks through land use, land-use change and forestry (LULUCF) activities. Emissions from LULUCF activities are reported to the UNFCCC with the purpose “to provide information on anthropogenic greenhouse gas emissions by sources and removals by sinks from land use, land-use change and forestry activities under Article 3, paragraph 3, and, if any, elected activities under Article 3, paragraph 4, in accordance with Article 5, paragraph 2 of the Kyoto Protocol (1998)” (UNFCCC 1997). The Norwegian LULUCF classes include managed areas of forest, arable land, pasture (not used as arable land), water and mire, and other open areas (i.e. mainly alpine heathland and ice).

www.nina.no

The Norwegian Institute for Nature Research, NINA, is as an independent foundation focusing on environmental research, emphasizing the interaction between human society, natural resources and biodiversity.

NINA was established in 1988. The headquarters are located in Trondheim, with branches in Tromsø, Lillehammer, Bergen and Oslo. In addition, NINA owns and runs the aquatic research station for wild fish at Ims in Rogaland and the arctic fox breeding center at Oppdal.

NINA's activities include research, environmental impact assessments, environmental monitoring, counselling and evaluation. NINA's scientists come from a wide range of disciplinary backgrounds that include biologists, geographers, geneticists, social scientists, sociologists and more. We have a broad-based expertise on the genetic, population, species, ecosystem and landscape level, in terrestrial, freshwater and coastal marine ecosystems.

1774

NINA Report

ISSN: 1504-3312
ISBN: 978-82-426- 4531-9

Norwegian Institute for Nature Research
NINA head office
Postal address: P.O. Box 5685 Torgarden,
NO-7485 Trondheim, NORWAY
Visiting address: Høgskoleringen 9, 7034 Trondheim
Phone: +47 73 80 14 00
E-mail: firmapost@nina.no
Organization Number: 9500 37 687
http://www.nina.no

NINA

Cooperation and expertise for a sustainable future