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# Land-Based Nature-Based Solutions for Climate Mitigation

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## Table of Contents

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# Executive Summary

This report assesses the potential of land-based Nature-Based Solutions (NBS) for carbon mitigation as part of the “Remove” and “Reduce” strategies in the Circular Carbon Economy framework. It assesses the benefits of land-based NBS, their trade-offs and unintended consequences, scalability, enablers, and degree of maturity to be deployed. Overall findings from this report are:

- Land-based NBS can play an important role in removing and reducing carbon, globally, and have the potential to minimize trade-offs with biodiversity and human wellbeing versus engineered solutions.
- Trade-offs with biodiversity, human wellbeing and adaptation are likely without careful design of options with consideration of interdependencies.
- Research is required to rigorously understand the intended benefits of land-based NBS versus alternative strategies in terms of costs, reliability and resilience to climate change. The evidence base is a barrier to implementation.
- Improved accounting of carbon emissions and sequestration is required, including refined assessment at national scale, and understanding of uncertainties.
- Mitigation potential is dependent on geographical setting, with the largest potential mitigation in fast-growing tropical and sub-tropical regions where capacity and financing are lowest.
- Wetland restoration and soil carbon storage invariably provide only positive impacts and therefore represent no-regrets actions.
- Agroforestry has a potentially significant role to play in climate mitigation because of its integration in livelihoods, as well as an array of other benefits, including climate adaptation. Better accounting for agroforestry in national and global assessments is needed.
- Implementing at scale requires overcoming financial and governance challenges, and requires multi-disciplinary approaches, new research and policy-interfacing.
- NBS is moving higher up the climate policy agenda, but policies with financial incentives need to be carefully managed to avoid land grabbing and compromising land tenure rights.
- Global climate finance is significantly underinvesting in land-based NBS; but there is potential for this to accelerate after COP27. Offsets from the NBS should not be used for continued emissions.
- The effectiveness of payments for ecosystem services schemes for forestry projects requires consideration of social and biodiversity benefits.
- Careful consideration is required on financing and governance of long-term land-based NBS projects, including account for saturation of mitigation, reversal and overshoot.
- Innovation and investment in research is required to fully understand the potential for land-based NBS to meet net-zero targets and contribute to sustainable development, in particular with respect to the rights of indigenous peoples and local communities.

# Background

NBS can be defined in many ways including "actions to protect, sustainably manage, and restore natural or modified ecosystems, that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (IUCN, 2016). The European Commission defines NBS as "Living solutions inspired by, continuously supported by and using Nature designed to address various societal challenges in a resource efficient and adaptable manner and to provide simultaneously economic, social and environmental benefits" (Maes and Jacobs, 2015). Other definitions focus on the types of challenges to be solved, e.g., "solutions [which] aim to use nature in tackling challenges such as climate change, food security, water resources, or disaster risk management" (Pauliet et al., 2017). Many other definitions exist, and there is substantial overlap with other concepts around enhancing, restoring and utilizing nature for a variety of purposes. NBS has evolved from the Ecosystem Approach (UNEP/CBD, 2000) and co-evolved with other and more established and targeted ecologically focused concepts of Ecosystem-based Adaptation (EbA; Colls et al., 2009; CBD, 2009), Ecosystem-based Mitigation (EbM; Epple et al., 2016), Ecosystem Services (ESS, MEA, 2005), Green Infrastructure (GI; Mell and Clement, 2019) and Natural Climate Solutions (NCS; Griscom et al., 2017), and now encompasses these more broadly as an umbrella term.

NBS sit in contrast to "hard" engineering approaches that dominate environmental interventions especially for climate adaptation, such as building flood defences (sea walls, flood barriers, levees) or exploiting renewable energy technologies. In general, NBS can be cheaper and offer co-benefits, such as environmental protection and generating economic benefits, and can be more resilient in the face of climate change or other pressures.

Broadly speaking, NBS can be applied in marine, coastal or land contexts and often are characterized separately. Land-based NBS (e.g., reforestation, natural flood management) have potential for meeting environmental goals outlined in national policy and international frameworks, including reducing flooding and increasing water quality, improving biodiversity, landscape aesthetics and social-cultural opportunities, all whilst sustainably intensifying agriculture and forestry and meeting carbon reduction targets.

### a) History of the framing of the concept of nature-based solutions

NBS concepts and applications have grown rapidly over the past 10 or so years, with an expanding scientific evidence base to support growing interest from governments, civil society and the business sector. The concept has gained more prominence as organizations seek ways to work with nature to tackle climate change and other global challenges of biodiversity loss, food and water security, and sustainable development. Understanding the value of nature and the services that it provides has long been embedded in traditional knowledge systems. However, NBS is still a relatively new concept in science and policy and has only recently been formalized into frameworks to guide design, application, monitoring and evaluation. These include frameworks focused on specific NBS approaches, such as for climate mitigation, as well as broader framings aimed at policy (e.g., WWAP, 2018).

The idea of working with nature and valuing the services that ecosystems provide emerged in the 1970s with the environmental movement but has its roots in earlier thinking around environmental conservation. Precursors stemmed from early discourses on sustainable development (e.g., UN Brundtland Commission, Brundtland et al., 1987), and the emergence of concepts of biodiversity (Wilson, 1988), natural capital (Barbier, 2011) and ecosystem services (MEA, 2005). These were reflected in subsequent framework and policy agreements including the adoption of the UNFCCC (UNFCC, 2022) and the 1992 Convention on Biological Diversity (CBD) (UNEP, 1993).

Global interest in NBS gained pace in the 2000s with the concept deriving from the Ecosystem Approach of the CBD, with an initial focus on the dependency of biodiversity and human wellbeing on well-functioning natural ecosystems (Cohen-Shacham et al., 2019), as well as focus areas, such as land and water resources management (e.g., Guo et al., 2000). The Millennium Ecosystem Assessment (MEA, 2005) was particularly important via its evaluation of the services provided by ecosystems and identified that ecosystems were changing in response to human management and impacts, and that this degradation was decreasing the ability of ecosystems to support human activities. The emergence of the term NBS elevated the concept of ecosystems services to recognize the potential for proactive management and restoration of ecosystems to maintain and enhance benefits. The World Bank presented the concept in 2008 (MacKinnon et al., 2008), with the IUCN in 2009 (IUCN, 2009) focusing more on conservation of biodiversity and its role in climate change mitigation and adaptation. These framings have expanded to include consideration of a wide variety of environmental sustainability and development goals, and their co-benefits (Eggermont et al., 2014; Seddon et al., 2019), such as integration in the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (UNEP, 2010) and UN Sendai Framework for Disaster Risk Reduction (UNDRR, 2022).

These earlier proposals represent a general shift in focus of nature conservation from natural systems to human benefits (Mace, 2014). For example, the IUCN has recognized and promoted the value of NBS for water, energy and food security, poverty alleviation and economic growth, in addition to climate mitigation and adaptation (IUCN, 2014). In the U.S., NBS concepts have been more narrowly defined and implemented such as “nature-based infrastructure” or “engineering with nature” referring to NBS measures that are focused on flood and erosion risk reduction (FEMA, 2009; USACoE, 2013; Sutton-Grier et al., 2018). In Europe, NBS is now recognized in policy frameworks and strategic documents as approaches to improve human health and wellbeing, such as the European Green Deal, EU Health Strategy, and EU Biodiversity Strategy. They are now a cornerstone of European funded research under the Horizon 2020 programme (Pauleit et al., 2017; European Commission, 2016), that marries ecosystem services and biodiversity conservation with economic growth, with a particular focus on urban environments and sustainable societal development (Maes and Jacobs, 2017). The fundamentally easier notion of NBS, as compared with ecosystem-based adaptation, for example, has enabled it to filter into policy and programmes (Nesshöver et al., 2017) and is now being incorporated in a variety of sectors, in the public and private realms. These include resolutions by the G7, G20 and the UN General Assembly and incorporation into international dialogues (e.g., WWAP, 2018; IUCN, 2021). In the private sector, guidance on the benefits and use of NBS and working with nature is emerging (e.g., WBCSD, 2020), building on a recognition that NBS can help bridge the gap between ambitions of economic development and conservation of ecosystems, and the risks in not doing so if biodiversity declines and ecosystems are damaged, as laid out in the World

Economic Forum's Global Risks Report (WEF, 2019; WEF, 2020) and shown by the proliferation of business coalitions focused on NBS (e.g., Business for Nature, 2020).

The NBS concept has garnered interest because of its flexibility and broad scope that can be considered and applied to multiple goals with co-benefits, and with potential for success in a range of contexts. However, there is also recognition that the concept is vague and overarching with missed opportunities to improve environmental management (Waylen et al., 2014), and potential to sow confusion and misuse in decision-making and application (Nesshöver et al., 2017), especially given the uncertainty of its relationship with other similar concepts. For example, concepts with longer legacies such as EbA/EbM prevail as the preferred concept or approach in many international fora. Current thinking is therefore focused on how best to frame the principles of NBS for the most benefit (Nesshöver et al., 2017; Sowińska-Świerkosz and García, 2022), highlight its potential to address global challenges and provide clarity about relationships with preceding and similar concepts. Given growing interest and demand in NBS from a range of sectors including government, civil society and business, there is a need to accelerate the implementation of pilot projects, develop standardized approaches and tools to help design, implement and scale-up NBS, minimize side-effects and trade-offs, and allow for monitoring and evaluation of progress and benefits. Recent progress in standardizing approaches by the IUCN (IUCN, 2020a) are presented in section 3a.

**b) Land based NBS for carbon mitigation – definitions and types**

In the context of climate mitigation, NBS are natural approaches to reduce concentrations of greenhouse gases (GHG) in the atmosphere through carbon storage and sequestration, and reduce emissions through land use management, and as such may also be referred to as Natural Climate Solutions (Griscom et al., 2017). Sequestration in general refers to the "transfer of atmospheric CO2 into other long-lived global pools including oceanic, pedologic, biotic and geological strata to reduce the net rate of increase in atmospheric CO2." (Lal, 2008). There are two main types of carbon in the context of NBS climate mitigation, which generally refer to organic carbon but are distinguished by their location: Green and Blue Carbon. Green Carbon is terrestrial-based carbon sequestration such as in forests and agriculture (although sometimes focused only on natural ecosystems), including soils. Sometimes peatland/wetland carbon is considered separately from Green carbon and referred to as Teal Carbon (Nahlik et al., 2016; Zinke, 2020), but here we refer to all terrestrial based NBS carbon mitigation as Green Carbon. Blue Carbon refers to coastal ecosystem-based sequestration, such as mangroves, tidal/salt marshes and seagrasses, and includes sediments. Other colour designations have emerged in recent years (Zinke, 2020) that provide a more nuanced view of organic carbon in the context of climate impacts and climate mitigation. Black, brown and red carbon further refer to different types of aerosols derived from wildfires and pollution that primarily affect the Earth's energy balance or promote cryospheric melting, often through changes to the surface albedo. Black carbon is also used to refer to biochar carbon storage, which has more recently been promoted as an option for carbon mitigation (see section 4b). Brown carbon has historically been used to refer to industrial forestry such as plantations (Mackey et al., 2008), although this terminology may have fallen from favour, given its more prevalent use in the context of aerosols and cryospheric processes.

**c) Scientific perspective on Green Carbon**

Green Carbon sits within the broader framing of the carbon cycle. Carbon is fundamental to life on Earth, in major part because of how it plays such a central role in climate regulation. The cycling of carbon is complex due to the range of interacting biogeochemical processes that connect the different stores and fluxes of carbon within the Earth system. Of interest here are the time scales of these processes and their interactions, especially within the terrestrial biosphere which is a major component of the global carbon cycle (Ciais et al., 2013) and represent the climate mitigation potential of Green Carbon. These processes can be categorized as fast and slow processes that act on different timescales.

Over the long term of 100s of thousands of years, slow processes of the global carbon cycle act to temper perturbations in climate as driven by variations in atmospheric carbon dioxide. These processes relate to chemical reactions and tectonic activity which move inorganic carbon in various forms between the rocks of the lithosphere, oceans and atmosphere. They act to prevent all carbon ending up in the atmosphere or all in rocks and keep the climate stable over these long time scales, rather like a thermostat (Berner, 2004; Isson et al., 2020). An increase in atmospheric carbon from increased volcanic activity will eventually be balanced with more chemical weathering due to higher temperatures and more rainfall that transports more carbon into the oceans and eventually to the ocean floor to become rock. Over such long time scales, the amount of carbon in the atmosphere is therefore regulated, leading to relatively invariant temperatures that are conducive to the development and proliferation of life on Earth.

Fast processes, on the other hand, occur on time scales of decades, years and shorter (Urbanski et al., 2007; Pappas et al., 2017). As such, they are the focus of carbon mitigation via Green Carbon because of the speed of current changes in the carbon cycle and the timescales over which we can intervene to prevent the worst of climate change. These time scales are dictated by the lifespans of life forms, as carbon forms the building blocks of living cells. They are expressed through the process of photosynthesis during growth of plants and phytoplankton in the oceans, which takes up carbon from the atmosphere, and through processes of decay and consumption, which return carbon to the atmosphere (Tharammal et al., 2019). Plants may die at the end of the season and decay via consumption by bacteria or are consumed by other life forms or by fire. In all cases, carbon dioxide is released back to the atmosphere.

Table 1 shows the estimated carbon storage in various carbon pools. The vast majority is stored in the lithosphere in marine sediments and sedimentary rocks and represents 99.985% of total carbon storage. The remaining 0.015%, although a tiny fraction of the total, has an outsized contribution to the carbon cycle because it is intimately linked to fast carbon cycle processes occurring at the timescales of interest for current climate change mitigation.

**Table 1.** Amount of carbon in the global component pools (values are approximate).

| COMPONENT | CARBON STORAGE (GTC) |
|---|---|
| SEDIMENTARY ROCKS | 100 MILLION |
| OCEAN | - |
| MARINE BIOTA | 3 |
| DISSOLVED ORGANIC CARBON | 700 |
| SURFACE OCEAN | 900 |
| INTERMEDIATE AND DEEP OCEAN | 37,100 |
| OCEAN FLOOR SURFACE SEDIMENTS | 1,750 |
| ATMOSPHERE | 829 |
| LAND | 3,650 - 4750 |
| VEGETATION | 450 - 650 |
| SOIL | 1,500 - 2400 |
| WETLAND SOILS | 300 - 700 |
| PERMAFROST | ~1700 |
| FOSSIL FUELS | ~10,000 |

**d) Green carbon within the carbon cycle**

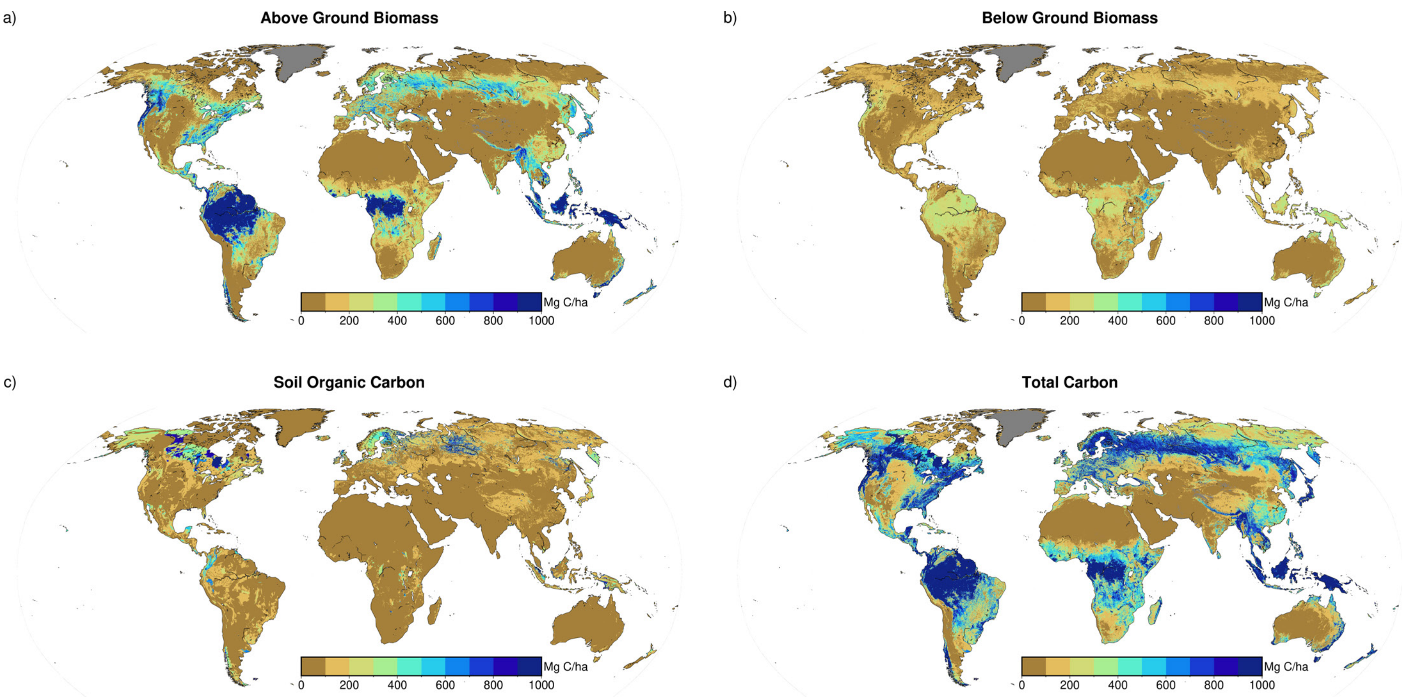
The biosphere in particular plays a significant role in carbon cycling and changes in carbon in the context of contemporary climate change. The biosphere contains, by definition, all organisms and the ecosystems in which they exist. Estimates of its carbon storage are about 1,950 – 3,050 GtC (excluding permafrost), with variability from year to year and longer term because of climate variability and change, and anthropogenic influences. For Green Carbon (the terrestrial part) this consists of plants, animals, and microorganisms (bacteria and fungi) on the earth's surface including in soils and aquatic environments, and this constitutes over 50% of organic carbon stores globally. The largest store of carbon in the biosphere is in soils of about 1500-2400 GtC globally. About 450-650 GtC are stored in plants, which is the second largest part of the biosphere by carbon storage. Trees make up the largest part of this because of the amount and density of wood. Peatlands contain about 250 GtC and are an important carbon store because of the anaerobic processes that decompose and retain carbon. The decaying remnants of plants (litter) make up the rest of the carbon storage of the terrestrial biosphere. The global distribution of carbon storage can be characterized to first order by broad biomes as shown in Table 2 and Figure 1. The majority is stored in tropical forests, followed by temperature and boreal forests.

**Table 2.** Breakdown of terrestrial carbon pools by broad biome (down to depth of 1m)

| Biome | Area (109 ha) | GLOBAL CARBON STOCKS (GT C) | | |
|---|---|---|---|---|
| | | Vegetation | SOIL | TOTAL |
| Tropical forests | 1.76 | 212 | 216 | 428 |
| Temperate forests | 1.04 | 59 | 100 | 159 |
| Boreal forests | 1.37 | 88 | 471 | 559 |
| Tropical savannas | 2.25 | 66 | 264 | 330 |
| Temperate grasslands | 1.25 | 9 | 295 | 304 |
| Deserts and semideserts | 4.55 | 8 | 191 | 199 |
| Tundra | 0.95 | 6 | 121 | 127 |
| Wetlands | 0.35 | 15 | 225 | 240 |
| Croplands | 1.60 | 3 | 128 | 131 |
| Total | 15.12 | 466 | 2011 | 2477 |

**Figure 1.** Density (Mg C ha-1) of (a) above- and (b) below-ground biomass, (c) soil organic content and (d) total carbon (sum of above-, below-ground and soil organic carbon).

Above- and below-ground biomass are from Spawn and Gibbs (2020) and downloadable from the Oak Ridge National Laboratories Distributed Active Archive Center (https://doi.org/10.3334/ORNLDAAC/1763). Soil organic carbon data are from the JRC (Hiederer et al., 2012).



The tight connection between the life cycle of plants and phytoplankton and fast carbon processes means that the Earth's carbon balance on the annual time scale is highly influenced by the life-cycle of the land biosphere. Figure 2a shows the evolution of atmospheric CO2 as measured from the Mauna Loa observatory in Hawaii over the past six decades, indicating the upward trend in CO2 but also the clear cycle driven by the seasonality of terrestrial vegetation. Figure 2b shows this seasonal ebb and flow over a typical year, averaged over global, northern hemisphere and southern hemisphere land. Figure 3 shows the equivalent spatial pattern in terms of vegetation gross primary production, which is often characterized as the "breathing" of the planet. In the northern hemisphere there are fewer plants in the winter and so the atmosphere contains a higher fraction of CO2. With the transition to northern hemisphere summer, plants grow and uptake carbon from the atmosphere. As the seasons cycle back into autumn and winter, plants die and return carbon to the atmosphere. Carbon is also added to the atmosphere by respiration in plants and animals (Tharammal et al., 2019). The opposite cycle occurs in the southern hemisphere, with a peak in carbon release and uptake at opposite times of the year. The global carbon cycle at seasonal time scales is dominated by the cycling of plants in the northern hemisphere because of the larger land mass and amount of vegetation.

**Figure 2.** Time series of (a) Global CO2 atmospheric concentrations from the Mauna Loa observatory (Tans and Keeling, 2022). Seasonal time series of global carbon cycle averaged over 2004-2013, for global land, northern hemisphere land and southern hemisphere land, showing the positive and negative fluxes of CO2 to and from the atmosphere (Data from Cheng et al., 2022).

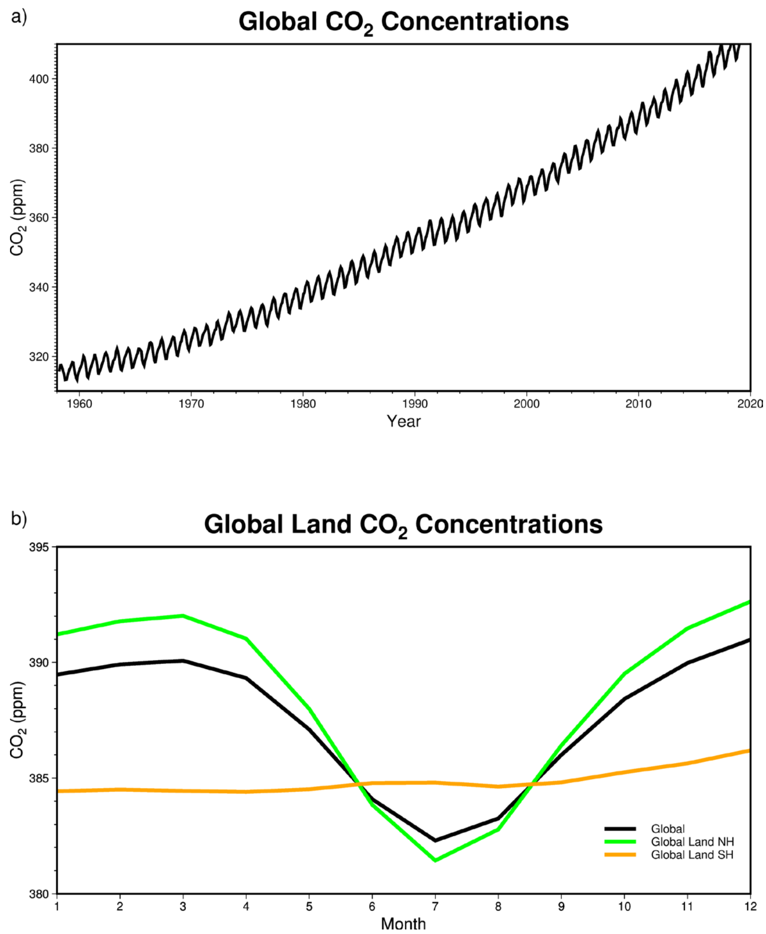
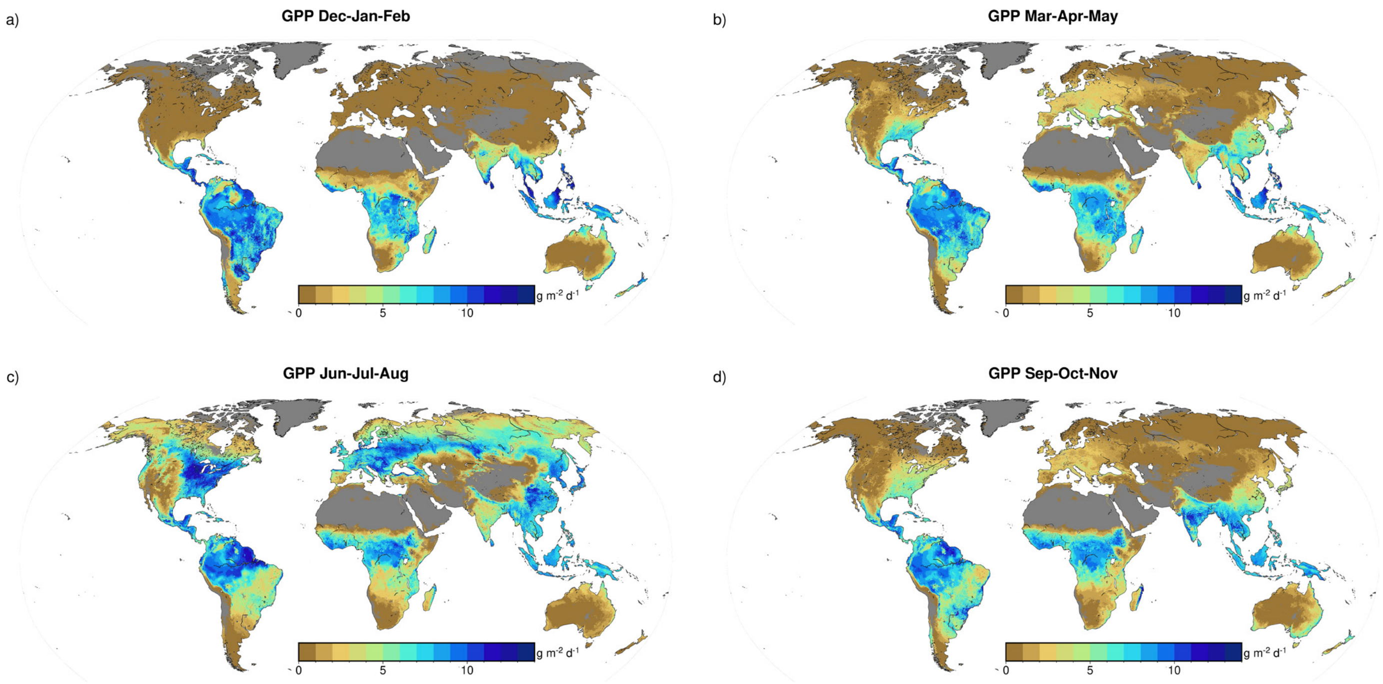
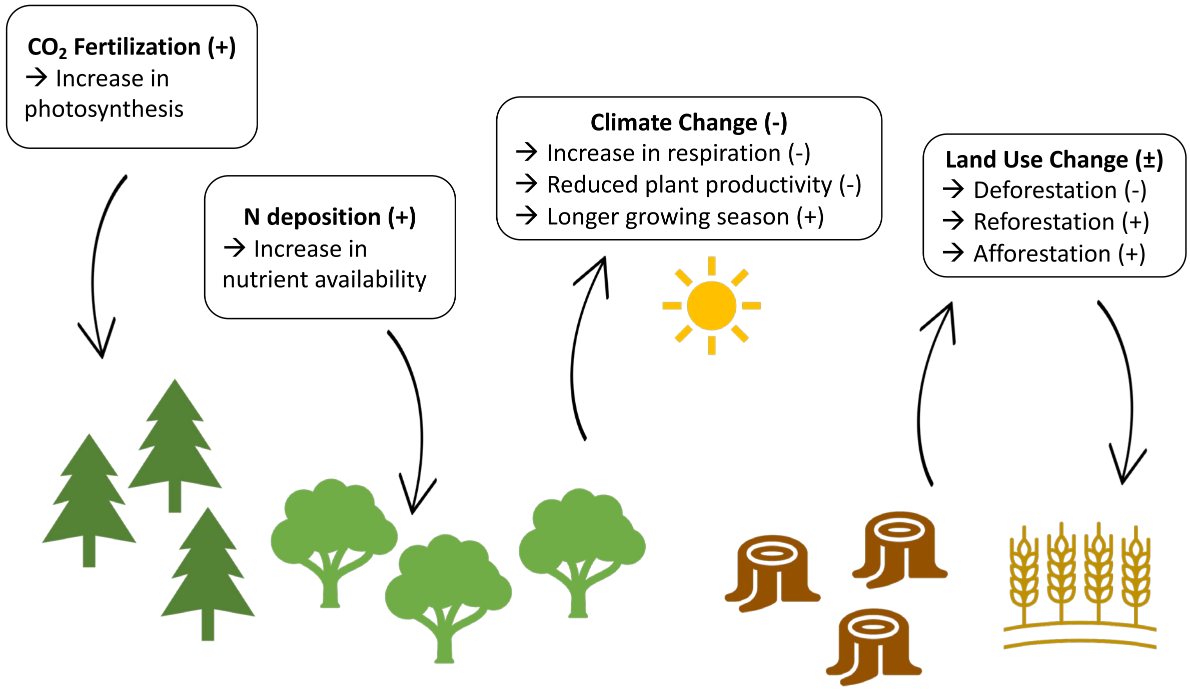


**Figure 3.** Maps of seasonal average gross primary productivity (GPP) of the land and oceans derived from data from the Moderate Resolution Imaging Spectroradiometer (MODIS) satellite sensor, showing the difference between seasons and the distinct change between northern hemisphere winter and summer in terms of the amount of carbon production. Data are from Madani and Parazoo (2020).




Carbon stored terrestrially evolves over longer time scales of years to centuries with climate variability and change, wildfire and volcanic activity, and via direct anthropogenic interventions, mainly land cover and land use change, and indirect impacts via $CO_2$ fertilization and nitrogen deposition (Adams and Piovesan, 2005, Stich et al., 2015; O'Sullivan et al. 2019) (Figure 4). Variations in climate drive variations in the uptake of carbon by vegetation (the terrestrial carbon sink) that contributes to the evolution of atmospheric $CO_2$ (Humphrey et al., 2018) as the land sink will uptake more or less carbon from year to year. In major part this is driven by large-scale variations in climate such as the El Niño Southern Oscillation (ENSO) (e.g., Keeling et al., 1995; Wang et al., 2016), especially in the tropics. These variations are driven by the tight coupling between carbon and water availability and in some energy-limited regions (e.g., high latitudes, wet tropics), by temperature and available energy. In addition, the role of water availability has been widely documented at the regional scale. The amount of carbon stored will increase during favourable climate conditions (moderate temperature and sufficient precipitation) and will decrease during less favourable conditions such as drought (Humphrey et al., 2018). Droughts and heatwaves especially can drive large reductions in vegetation growth and carbon uptake resulting in significant reductions in the carbon sink (Ciais et al., 2005; Phillips et al., 2009). Fire and other disturbances (e.g., disease, pests) will also tend to decrease carbon storage (Quirion et al., 2021). Overall shifts in climate are having longer term impacts, with increased precipitation and the length of the growing season increasing the land carbon sink, and decreases in precipitation and increases in temperature (in the tropics) limiting productivity and reducing the sink (Tharammal et al., 2019).

**Figure 4.** Major drivers of the terrestrial carbon sink: CO2 fertilization, nitrogen (N) deposition, climate change, and land use and land cover changes (LULCC). Downward arrows represent a carbon flux to the terrestrial biome (sink). Upward arrows represent a carbon flux from the terrestrial biome (source). Adapted from Tharammal et al. (2019)



# Background

Anthropogenic activities over longer timescales, have impacted biospheric carbon stores significantly, primarily as atmospheric pollution (mainly $CO_2$, and also methane, $CH_4$ nitrous oxide, $N_2O$, and other gases) but also land use changes over the last three centuries. Fossil fuel burning has perturbed the global carbon cycle, with about a 40% increase in atmospheric $CO_2$ concentrations since the pre-industrial period (Ciais et al., 2013). About half of total emissions have been offset by uptake by the terrestrial and oceanic carbon sinks. In turn, atmospheric pollution has acted to change the climate via global warming and changes in precipitation patterns and extreme events (Seneviratne et al., 2012), which impacts on carbon uptake at annual time scales as noted previously. There is also a feedback between atmospheric $CO_2$ levels and vegetative growth via modification of photosynthesis and transpiration rates via regulation of stomatal openings that tends to drive more efficient water use with increasing $CO_2$ concentrations (Gentine et al., 2019), so called "$CO_2$ fertilization". This has tended to offset some of the potential losses in vegetation productivity and associated carbon storage due to climate change (Terrer et al., 2021). Other human influences have provided small offsets to overall carbon losses driven by land use change. For example, nitrogen deposition from the atmosphere, which has accelerated due to anthropogenic activities (mainly fossil fuel burning and fertilizer production), has increased soil and biomass carbon stores (Tipping et al., 2017). Nitrogen availability can limit vegetation primary productivity and can therefore play a key role in the future evolution of the land carbon sink (Fisher et al., 2012; Goll et al., 2012).

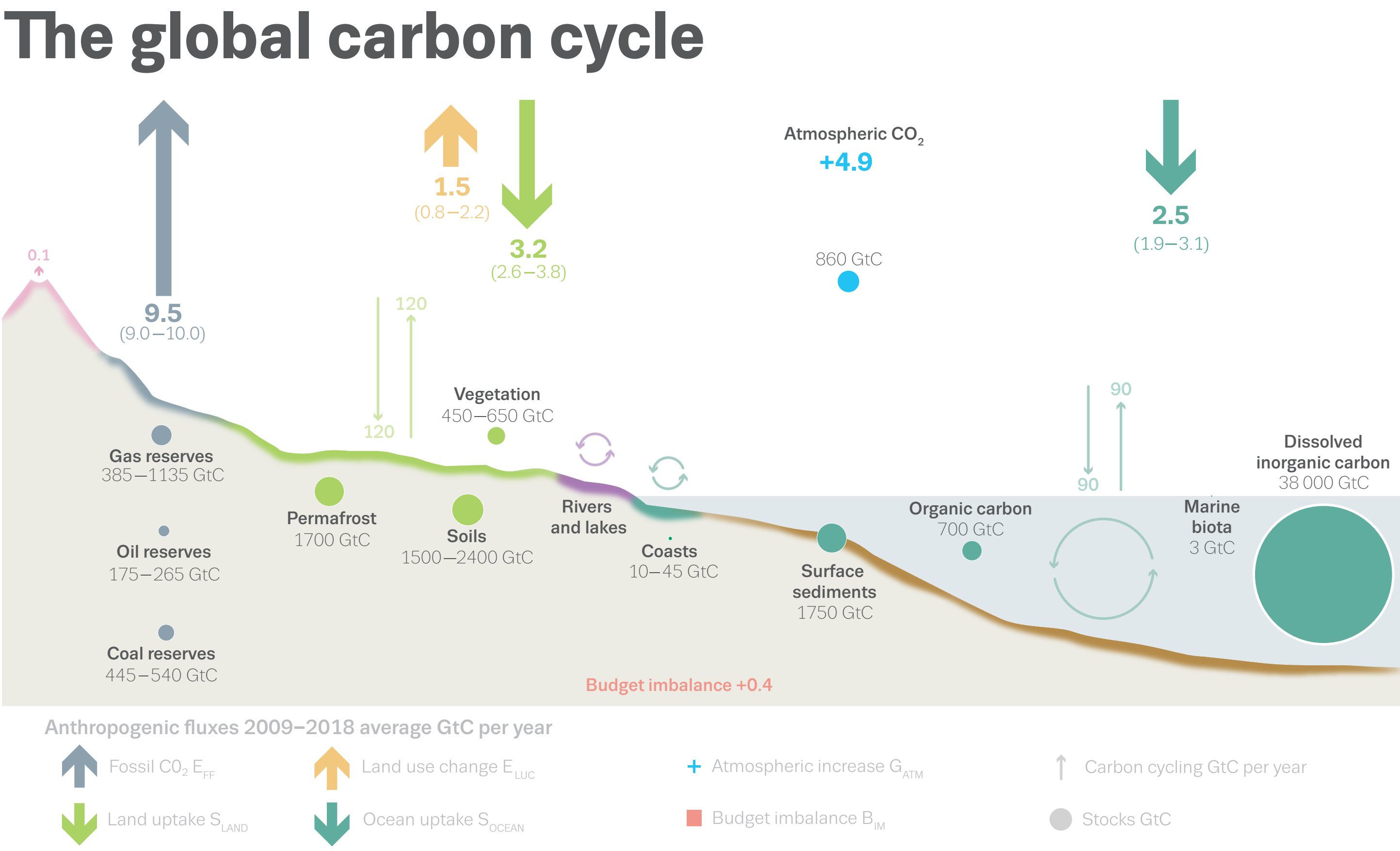
About one-third of the global land surface has been affected by anthropogenic land use changes in the last millennium (Arneth et al., 2019), which includes deforestation and afforestation, wood harvesting, grazing and shifting cultivation. Agricultural land has increased by 400% since the beginning of the 18th century and now accounts for more than one-third of global land use (Goldewijk et al., 2017). By area, primary land is estimated to have decreased by about 60%, from 125M $km^2$ in 1850 to 50M $km^2$ in 2005 (Hurtt et al., 2011), due to deforestation for agriculture and wood harvesting, and focused mostly on mid-high latitudes such as central and eastern N. America and Europe. Recent very high-resolution analyses indicate that this may be underestimated (Winkler et al., 2021), and to have affected a third of global land in the last six decades. Globally, these recent changes are characterized by opposing trends in the northern and southern hemispheres, with afforestation and cropland abandonment in the north, and deforestation and agricultural expansion in the south. The transition between forests and agricultural land induces large changes in carbon storage, with forest clearance to crop or pasture lands driving the largest decrease in the land carbon sink (Grassi et al, 2017; Le Quéré et al., 2018) and vice-versa. Land management practices associated with agriculture are also important for carbon, in addition to the bulk loss of carbon with deforestation. Lower vegetative activity from mono-crops, for example, leads to less plant matter to be decomposed. Coupled with management practices such as tillage and lack of erosion prevention measures, this reduces soil organic carbon stores (Smith et al., 2016a; Wei et al., 2014). Land use and land cover change can also impact on climate via alterations on the water and energy budgets, through changes in surface albedo, evapotranspiration and surface roughness (Boisier et al., 2012).

**Current understanding of the evolution of the global and regional carbon cycles**

The current state of the global carbon cycle is shown in Figure 5. This is updated almost every year from the global carbon budget in IPCC reports and by the Global Carbon Project. The budget consists of emissions (sources) from fossil fuels and land use change (mostly deforestation), and uptake (sinks) in the atmosphere, ocean and land. The latest estimates of fossil fuel-based emissions of carbon reached 10 GtC yr–1 for the first time in 2018 (Friedlingstein et al., 2019) and averaged over the last 10 years is about 9.5 GtC yr–1. These estimates are subject to variability from year to year and overall trends and are also subject to uncertainties in the data and methods used, which results in some uncertainties in individual budget components and an overall imbalance. Nevertheless, there has been a significant increase (~6.5 GtC) in emissions since the 1960s, although the rate of increase has declined up until the 1990s, but has started to increase since, at least at the global scale. The rate continues to increase per capita, despite regional declines in total emissions for the U.S. and Europe (Friedlingstein et al., 2019). China has been the largest country emitter of fossil fuel emissions since the mid-2000s, whilst India has also increased significantly over this period.

The land uptake by terrestrial ecosystems is about 30% of the fossil fuel emissions (Friedlingstein et al., 2019), and most of this is due to forest uptake. The terrestrial sink has more than doubled from 1.3±0.4 GtC yr–1 in the 1960s to 3.2±0.7 GtC yr–1 in the decade up to 2018. This has been driven mainly by increased net primary productivity due to increases in atmospheric CO2 (Tharammal et al., 2019; O'Sullivan et al., 2019). Within this, there have been relatively large interannual variations driven mostly by El Niño events that caused decreases in the sink (Rödenbeck et al., 2018). Despite deforestation being the main source of emissions in many tropical countries, net forest sinks are dominant in temperate and boreal regions and contribute significantly to the global picture (Pan et al., 2011). Emissions associated with land use, including agriculture and forests, are about 10% of GHG emissions for CO2 only and about 25% including CH4 and N2O. The net terrestrial sink accounting for the emissions due to land use change has increased since the 1960s from a source of 0.2±0.8GtCyr–1 to a sink of 1.7± 0.9 GtC yr–1 in the last decade (Gilfillan et al., 2022).

**Figure 5.** Schematic of the carbon budgets as of now, shown as carbon stores (circles) and the perturbation/ change since pre-industrial period (shown as arrows). From Friedlingstein et al., (2019).




The combined effects of climate variability and change, direct anthropogenic land use change and management, and feedbacks between the atmosphere and terrestrial biosphere have led to significant changes over the past decades in the storage and cycling of terrestrial carbon (Bloom et al., 2016; Friedlingstein et al., 2019). This has implications for how the carbon cycle evolves in the future and how we manage carbon in the context of mitigation. However, our knowledge of how these stores and cycling of carbon are changing, and the influence of human activities, especially at regional scales, is generally lacking. This is because of the reliance on small-scale, and often short-term, data collection (Janes-Basset et al., 2021), and therefore heavy reliance on modeling approaches, which are not well constrained by the limited observational data and have inherent uncertainties (Shi et al., 2018; see section 6). Whilst global estimates are scientifically useful and promote awareness in public and policy arenas, they are uncertain and do not match well with the atmospheric concentrations of CO2 measured in the atmosphere and its growth rate (Friedlingstein et al., 2019). Much of the global imbalance is due to year-to-year variability in the land (and ocean) sink as well as around the contribution of land use change. Regional estimates may help to resolve much of this uncertainty (Le Quéré et al., 2018), to better understand land carbon cycle processes and feedbacks and to constrain the land carbon models that are used to provide future projections of climate, and vitally for providing the baseline for land-based mitigation efforts (Ciais et al., 2022).

Regional initiatives and research projects (e.g., RECCAP/2, Raymond et al., 2013; Ciais et al., 2022) have attempted to reconcile available observations and model estimates to develop regional budgets, including drivers from a process and cause perspective, and also to reconcile top-down approaches based on disaggregating atmospheric concentrations/values with bottom-up approaches based on observations and accounting (book-keeping) methods (Kondo et al., 2019). Recently, Bastos et al. (2020) found that estimates of the carbon budgets in tropical regions, particularly Brazil, Southeast Asia, and Oceania, contribute the most to uncertainties at the global level, and are mostly related to uncertainties in the land carbon models and their response to climate variability, and also driven by lack of data in these regions to constrain the models.

Tropical regions are of high importance given their large contribution to the global carbon budget (Saatchi et al., 2011) but large uncertainty, which is vital to reduce to enable better understanding of drivers of perturbations and accounting to develop baselines for mitigation. Much uncertainty revolves around the debate about whether tropical regions are a source or sink of carbon. Recent consensus points towards carbon neutrality (neither source or sink) (Gaubert et al., 2019; Kondo et al., 2019), but with recognition of the strength of processes that provide perturbations to the carbon cycle, including the high effect of CO2 fertilization (Schimel et al., 2015), and susceptibility to large variability in climate driven by ENSO variations that increases net carbon emissions during drought, and persistently high deforestation rates (Achard et al., 2014). Uncertainties remain about the sensitivity of the terrestrial biome as a carbon sink to CO2 fertilization, especially in the tropics (Smith et al., 2016), as well as the role of forest degradation (Achard et al., 2014).

There are considerable uncertainties in other components of the terrestrial carbon cycle and how they are changing. Wetlands (peatlands, inland wetlands and water bodies) in particular are important stores of carbon, containing 20-30% of global soil carbon (Lane et al., 2016) which is disproportionate to the total land surface area of 5-8% of global land. Wetlands are also some of the most changed and changing environments (Lal, 2008), with global losses of 35% since 1970s (CoW, 2021) and losses of up to half in places like the U.S. (Nahlik and Fennessy, 2016) and Europe (Voerhoven, 2014) with net transfer of carbon to the atmosphere from the lost carbon storage. Estimates of the amount of carbon loss is very uncertain, but with some estimates at about 10% of global fossil fuel emissions.

High latitude environments are also extremely important and our knowledge is evolving in how they function and respond to climate change (MacDougall et al., 2015). Although they have low productivity because of light, temperature, energy and nutrient limitations and are a relatively small part of global terrestrial productivity (< 10%) there are very large uncertainties in the carbon budget in terms of its magnitude, variability and sensitivity to external factors, such as climate change. The latter is especially important to understand future change given that high latitudes (and elevations) have seen the largest changes in climate globally, especially for temperature, with double temperature increases compared with the global mean and extension of the growing season (IPCC, 2021). Vegetative growth has increased, as shown by greening trends (e.g., Arndt et al., 2019), by direct $CO_2$ effects on productivity as well as warming temperatures, extended growing season and increased nutrient availability and cycling (Rogers et al., 2022), although the changes are complex due to the effects of land cover change (Wang and Freidl, 2019). Thawing of permafrost is a large concern given the potential to release significant quantities of carbon via methane, as well as effects on local to regional hydrology and landscapes (Carpino et al., 2018; Hugelius et al., 2020). Peatland regions overall may switch from a sink to a source in the near future (Schuur et al., 2015; Zhong et al., 2020). Furthermore, disturbances such as wildfires, pest and disease outbreaks, and drought are critical drivers in the region but also very sensitive to climate change (Seidl et al., 2017).

Drylands are a key global biome because they cover ~40% of the earth's surface (excluding Greenland), are home to 2 billion people whose livelihoods are highly dependent on the dryland ecosystems and high climate variability, and are likely to become more arid and variable overall with future climate change (Prăvălie et al., 2019). Carbon storage in drylands is generally low because of a range of constraints, especially water availability and poor soils. Globally, carbon stocks in drylands are about 1404 PgC, with a little over half (56%) in above-ground biomass (Hanan et al., 2021). Dryland soils are estimated to contain about 646 PgC which represents about 30% of global above- and below-ground and surface soil carbon storage (Plaza et al., 2018). This diverges from global estimates (e.g., Friedlingstein et al., 2020) because of undercounting of dryland tree biomass and herbaceous plants (Hanan et al., 2020). Soil inorganic stocks are generally higher in drylands with about 80% of the global stock (1558 PgC; Plaza et al., 2018).

In general, the balance of carbon in the terrestrial biome is uncertain because it is driven by the balance of a complex set of drivers, processes and feedbacks. Nitrogen deposition, forest regrowth and climate change ($CO_2$ fertilization, increase warming particularly in high latitudes, and longer growing season) will tend to increase the sink, whilst drought, conversion of forests to agriculture and disturbances, can reduce carbon uptake (Huntzinger et al., 2017). There is further significant uncertainty in land use changes and associated emissions (Kondo et al., 2019). The lack of observations that can be used to tease apart these various factors and the uncertainty in land carbon models means that much more research is required (Ciais et al., 2022; Rogers et al., 2022).

**e) Relationship with GHG emission trajectories and relevance to mitigation policy --- options for Green Carbon to mitigate climate change**

Climate change is recognized as a planetary emergency. To avoid the worst of projected climate change, global warming must be kept well below 2oC and preferably at 1.5oC (IPCC, 2018). The emissions reductions required to meet these targets are laid out by the IPCC as part of the Paris Agreement of 2015 and entail reaching net-zero emissions by 2050. This in turn requires large-scale capture and sequestration of carbon already in the atmosphere, as well as massive and rapid decarbonization of our economies with cuts in emissions by nearly 50% by 2030 (Tollefson, 2018). Nevertheless, and despite intense scientific and policy debates and international commitments, global temperatures continue to rise. To date, global warming has reached about 1.1oC since pre-industrial levels (IPCC, 2021), with GHGs continuing to rise at about 1.4% per year. If this trajectory continues, then it is increasingly likely that 1.5oC of warming will be exceeded in the next couple of decades (UNEP, 2020). This requires transformative changes (Fazey et al., 2018; Pörtner et al., 2021) given the relatively small progress in reducing emissions so far (Lamb et al., 2021).

To reach these global warming targets requires a significant portion of action around how land is managed to improve and increase carbon sinks, and to reduce emissions from land use activities (Field and Mach, 2017, Grassi et al., 2017; IPCC 2019). Although not explicitly called NBS by the IPCC, Green Carbon has an important role to play in carbon mitigation, with estimates that use of NBS increases the chances of achieving the Paris Agreement by 66% (Griscom et al., 2017). NBS present many opportunities for sequestration and reducing emissions of carbon, as well as other GHGs, through conserving, intensifying and extensifying the coverage of terrestrial vegetation. This can also include restoration and improved management of land, across different biomes of forest, grasslands and wetlands, as well as more sustainable management of agricultural lands (IUCN, 2021).

Interest in the role of Green Carbon as one part of the climate solution has been around for many decades but has been most emphasized in international climate negotiations only since the 2007 UNFCCC COP13 in Bali, where a focus on the role of natural forests for storing carbon came to formal prominence (Mackey et al., 2008). An important part of this in the context of forest-based carbon storage is the international REDD (“reducing emissions from deforestation and degradation”) policy as part of the UNFCCC, as an approach to incentivize reductions in emissions from deforestation and forest degradation. Importantly, carbon storage in natural ecosystems such as tropical forests was not taken into account in climate mitigation before then. REDD+ emerged at COP14 in 2008 to expand the scope of REDD to cover conservation, sustainable management and enhancement of forest carbon in developing countries.

In practice, carbon mitigation has to be addressed at the national level through Nationally Determined Contributions (NDCs; UNFCCC, 2015), as committed to in the Paris Agreement. As of the end of 2020, 189 countries and the EU had submitted intended Nationally Determined Contributions to the UNFCC as contributions to the Paris Agreement. Much of the NDCs relate to land-based carbon, in the form of land use, land use change and forestry (Strohmaier et al., 2016) (see section 3b for a detailed review of NDCs), and often refer to NBS, in particular to forests rather than other natural ecosystems. Land management (or stewardship) represents one of the most mature approaches for carbon mitigation; yet there are large uncertainties in how these commitments can be prioritized by governments, and funding can be targeted to on the ground actions at the sub-national level (Haltman et al., 2020; Griscom et al., 2020) (see section 8).

**f) Co-benefits and trade-offs; synergies with other programmes and climate strategies.**

There are large benefits of NBS directly on carbon mitigation, but also many co-benefits including for climate adaptation and other ecosystem services (Smith et al. 2013; Seddon et al., 2020). This is because a key character of NBS is their multi-functionality, which provides unique opportunities to deliver on multiple ecosystem services simultaneously (Gómez Martín et al., 2020). NBS have potential to promote synergies and reduce trade-offs by taking a more holistic approach (Seddon et al., 2019) and realize synergies with sustainable development as represented by the UN Sustainable Development Goals (SDGs). Such synergies cannot be realized with engineered approaches that are generally focused on one goal (e.g., flood protection) and often at the expense of ecosystem services and biodiversity.

If well designed, NBS for climate mitigation can be cost-effective and increase resilience and adaptation to climate change, and a range of ecosystem services (Roe et al., 2021) essential for livelihoods and wellbeing such as water resources and hazard mitigation, food, fibre and fuel provision, and climate regulation. For example, reforestation can store more carbon, and at the same time decrease flood risk from climate change and enhance conservation of biodiversity (Newell et al., 2013). In urban areas, planting of trees can further decrease air pollution and provide cooling, as well as a range of recreation and health benefits (Janhäll, 2015). Managing and increasing carbon storage in soils has a range of co-benefits because of how it increases water holding capacity and nutrient retention (Rawls et al., 2003; Almaraz et al., 2021) with multiple benefits to biodiversity, water and food security (Davies, 2017). Synergies with biodiversity are one of the main co-benefits of land-based NBS for carbon, with long-standing evidence of the association between carbon stocks and species richness, especially for forests (Strassburg et al., 2010).

In contrast, the conversion of natural ecosystems, such as deforestation of tropical forests, is the second largest source of anthropogenic CO2 emissions, only behind fossil fuel burning (Lamb et al., 2021), and is the main driver of declining biodiversity and loss of species (Alroy, 2017; Giam et al., 2017). Deforestation rates have decreased since the 1990s, but continue relatively unabated, especially in the tropics, despite the overwhelming scientific evidence and political backing for their preservation for climate mitigation and regulation, and biodiversity conservation (FAO and UNEP, 2020).

In practice, NBS is often implemented in conjunction with other measures to meet a specific challenge, e.g., reforestation can be complemented by improved market access to improve food security. Ideally, solutions should be designed with this in mind to ensure that other benefits are materialized and side-effects and trade-offs are minimized (Smith et al., 2020). Potential trade-offs and unintended consequences are serious concerns, especially if policy is poorly designed and implementation is not well-thought through. Trade-offs can occur if NBS is implemented without consideration for other potential benefits (Seddon et al., 2020) such as reforestation projects based on monocultures which have low biodiversity value and low resilience in the face of climate change, or bioenergy (Humpenöder et al., 2018) (see section 6c and 9).

The climate mitigation potential of NBS and multiple co-benefits means that there are overlaps and synergies with intergovernmental climate strategies in addition to the UNFCCC, and multilateral programmes around sustainable development. These include the SDGs, Convention on Biological Diversity (CBD), Aichi Biodiversity Targets, Land Degradation Neutrality (LDN), the Bonn Challenge, the UN Decade on Ecosystem Restoration and the Convention on Wetlands of International Importance especially as Waterflow Habitat (RAMSAR) (IUCN, 2021; Roe et al. 2021). The benefits of NBS have led to its recognition and endorsement in a range of high-level reports (Seddon et al., 2020) including the IPBES Global Assessment (IPBES, 2019), the IPCC Climate Change and Land Report (IPCC, 2019) and the Global Adaptation Commission Report (GCA, 2019).

# Timeline and steps of Land-based nature-based solutions

**a) Land based NBS science: state-of-the-art and research gaps in the context of uptake, implementation, scaling up and mitigation targets.**

Despite growing interest in NBS, it is generally an under-researched topic, and as such evidence to promote, guide policy and implement NBS is still emerging (Li et al., 2021; Sowińska-Świerkosz and García, 2022), and there is a lack of clarity overall on the concept and its principles. Research has significantly increased in recent years, along with general interest in governmental and non-governmental fora (Li et al., 2021), with prominence in European research funding across a range of programmes. This and other funding programmes have garnered more prominence in the research literature (including a dedicated research journal). Most research has emerged from Europe via substantial research and innovation investments of €282m (up to 2020) through the H2020 programme (and now Horizon Europe programme) as well other investment through the COST, ERDF, LIFE+ and EIB's Natural Capital Financing Facility programmes (EC, 2020). Much of the recent focus of European funding has been on the development of communities of stakeholders across disciplines, to share best practices and improve the evidence base and capacity to implement (Maes and Jacob, 2017; Faivre et al., 2017).

NBS emerged in the scientific literature from about the mid-2010s and has been promoted and discussed in more general public discourses from about that time also (EC, 2020; Liu et al., 2021). Most of the evidence developed so far has been focused on management of agricultural and forest lands, including afforestation and reforestation. There is also a rapidly growing research based focused on the potential for NBS in urban areas building on a longer history of research and applications in green infrastructure (O'Sullivan et al., 2020). Geographically, research has naturally focused on European research frameworks, but there are also pockets of interest in, e.g., Singapore, China, and Canada, where related concepts such as EbA are more prevalent. A number of reviews of NBS have appeared recently (e.g. Li et al., 2021; Nesshöver et al., 2017; Sowińska-Świerkosz and García, 2022) focused on different aspects of the conceptualization, science, policy and implementation of NBS, and focused in different areas, such as sustainable urban development. In general, a range of environmental challenges have been associated with NBS, and this has allowed it to be embedded into policy and research agendas, though there remain uncertainties in its connection and relationship with related concepts. This is emphasized by recent reviews that have highlighted the lack of specificity of the NBS concept and the implications of this (Nesshöver et al., 2017; Sowińska-Świerkosz and García, 2022).

Initial work focused on development of concepts and definitions (IUCN, 2016), and this has evolved rapidly to assess and compare case studies, and share best practice (Chausson et al., 2020). Recent focus has been on synthesizing evidence and understanding the

cost-effectiveness and potential for upscaling (Sowińska-Świerkosz and García, 2022). Work has also looked at the development and evaluation of interventions, yet more research is required to understand the socio-economic viability including perceptions of NBS, the relationship with underlying traditional systems and the governance contexts to ensure effective implementation and upscaling of interventions. Current research is aimed at developing standards and principles, and practical guidelines on approaches to NBS to help with design, implementation, monitoring and evaluation. These include the NBS impact evaluation framework (Raymond et al., 2017), NBS handbook (Somarakis et al., 2019), NBS core principles (Cohen-Shacham et al., 2019), and NBS global standard (IUCN, 2020b).

However, research is required to unify these approaches including for monitoring and evaluation in a range of contexts as well as the underlying concepts and guiding principles. Uncertainty around NBS lies fundamentally in the range of definitions, that tend to be vague and with wide scope (Sowińska-Świerkosz and García, 2022). This is a particular barrier around what can be classified as NBS (Belamy and Osaka, 2020), and there is ongoing debate on the impact of this on uptake of NBS (e.g., Sarabi et al., 2019). This essentially stems from the multidisciplinary and multi-functional character of NBS (Raymond et al., 2017), which can be viewed differently based on the desired outcome and approach, whether focused on, for example, climate mitigation, sustainability, conservation or socio-ecological viewpoint, and its orientation as an umbrella or alternative term for other ecosystem-orientated approaches (Lamb et al., 2020). Even when focused on land-based NBS for climate mitigation, the diversity of types of NBS and contexts means that there is wide diversity in how NBS is characterized. In order to enhance the inclusion of NBS in policy and decision-making, and to aid in operationalization and upscaling, there needs to be clarification on definitions on what constitutes NBS and how that can be translated into guidance for policy development and implementation (see section 5c and 8). Without this there is potential for mischaracterization of related approaches as NBS, and potential for misuse.

Research is also required on how to scale up solutions to landscape and national levels (Roe et al., 2021), given that experimentation and small-scale studies globally have been evaluated scientifically and have shown their effectiveness (although not necessarily the cost-effectiveness) (Carbon Market, 2020). Work is particularly needed on how to do this through financial incentives such as carbon markets (see section 8). To enable this, research is needed on developing data resources for effective scaling up, including development and open access to high-resolution estimates of current carbon storage and potential for carbon sequestration through land management, restoration or enhancement (Grassi et al., 2017). Research gaps also revolve around the understanding and monitoring of interactions between large-scale NBS interventions that can also lead to double counting, to ensure that the net impact is properly measured. Practically, standardized guidelines are needed to prevent such overlaps of benefits as well as leakage whereby gains in one area lead to losses in another (e.g., preservation of forests leads to deforestation elsewhere; see section 9 on risks) (Royal Society, 2018). More research is also needed on the potential impact of variations in practice across large-scale implementations and over time to ensure long-term net benefits, and to avoid reversal (e.g., for soil carbon storage).

Research remains to be done around how to implement NBS effectively, which includes cost and resilience, but also dimensions of design, legal and financial instruments, and governance (Albert et al., 2019). Resilience of NBS to climate change and other externalities is important for a number of reasons, including comparison with engineered approaches. For example, reforestation is best done with clear account of resilience to a range of pressures including longer droughts, which is generally best addressed through a diverse approach and avoiding mono-plantations which are more susceptible to climate change impacts, pests and disease. In particular, there are uncertainties over the reliability of NBS and their cost-effectiveness, especially when compared to alternatives, including engineered solutions (Seddon et al., 2020; Chausson et al. 2020). Effectiveness is also dependent on the time scale, given that NBS approaches which are effective in the long term, may have trade-offs in the short term or be more difficult and costly than other approaches in terms of start-up costs. In this sense there are also gaps in how NBS can be effectively integrated along with grey infrastructure, whether existing or new.

Overall, there is therefore some way to go before clear guidelines based on a solid research evidence base can be provided on effective implementation and on cost-effectiveness versus other solutions (Seddon et al., 2020). This presents barriers in many areas, including institutional, financial, political and social, to their uptake and upscaling (Kabisch et al., 2016; Saribi et al., 2019; Cortina-Segarra et al., 2021). More work needs to be done to identify and promote the multi-functional capabilities of NBS, their ability to address a range of goals simultaneously to provide benefits to a range of stakeholders and beneficiaries, and the synergies and trade-offs. Understanding how NBS connects across sectors and other global frameworks and policies (e.g., SDGs) and into the broader socio-economic landscape (e.g., in job creation), and whether policy coherence around NBS can be achieved, would be useful (Cohen-Shacham et al., 2019). As with any emerging approach to address environmental and societal challenges, work needs to be done to overcome the uncertainty and natural resistance to invest in what might appear to many stakeholders as a new or risky endeavour (Waylen et al., 2014; Pagano et al., 2019; Santoro et al., 2019). Promotion and mainstreaming are therefore key in effective implementation and upscaling (Pagano et al., 2019; Schanze, 2019) as well as ways to engage and coordinate the diversity of stakeholders throughout the process of implementation (Ferreira et al., 2020). New approaches and tools are likely needed.

**b) Current context of land-based NBS in national inventories and NDCs; range of current and potential NBS projects in inventories and NDCs.**

The Paris Agreement has a strong focus on the use of ecosystems for mitigation and adaptation that can be classed as NBS, including conservation, restoration and enhancement of carbon sinks. On a country basis these are manifest in Intended Nationally Determined Contributions (INDC) submitted to the UNFCC as an intended commitment to the long-term goals of the Paris Agreement and represent national efforts to reduce emissions, as well as adapt to climate change. With ratification of the Paris Agreement these then become NDCs. The potential to realize the commitments under the Paris Agreement lie in the notion that countries will revise and strengthen their commitments within their NDCs based on a periodic global stocktake of progress.

The land sector, including agriculture and land use more generally, has an important role to play in the global response to climate change and particularly around commitments to climate mitigation in the NDCs. Of relevance to land-based NBS are those targets and/or actions that are focused on the Agriculture, Forestry and Other Land Use (AFOLU) sector as defined in the IPCC Guidelines for National GHG Inventories (NGHGI). These are generally broken down into agricultural and land use, land use change and forestry (LULUCF) activities. Within individual NDCs, these further comprise a wide range of mitigation options such as enhancing forest carbon stocks, increasing afforestation, reducing deforestation, and improving management, with the majority of commitments focused on deforestation and management (Grassis et al., 2017). To date, 186 countries have incorporated AFOLU mitigation actions in their NDCs, either as specific land use actions mostly focused on agriculture and forestry, or as a broader and unspecified set of land-based actions (Crumpler et al., 2019; Roe et al., 2021). A more comprehensive assessment is complex, however, because of the diversity of individual NDCs, in terms of their structure, scope and detail (Grassi et al., 2017).

Forsell et al. (2016) notes that NDCs are mostly derived from LULUCF contributions, being about 20% of the total contributions, which is important because of the large potential to achieve NDCs through forestry for many countries. FAO's analysis (FAO, 2016) indicates that mitigation is addressed within country NDCs mostly by agricultural and/or LULUCF interventions (89% of countries), with prominence in eastern and southeastern Asia, SSA, LAC and southern Asia. One hundred and forty-eight countries include agricultural in their contributions, with the highest percentage in developed countries and the least in developing countries, and collectively contribute to 92% of global agricultural emissions, with sub-Saharan Africa (SSA) most prominent in terms of percentage of countries. One hundred and fifty-seven countries include LULUCF contributions, with 80% from developing countries, again with the highest contribution in SSA. This does indicate that a significant number of countries do not include LULUCF in their NDCs or will decide at a later stage (Forsell et al., 2016). However, quantitative targets are rarely provided, with only about 20% of forest sector promises having such targets (Seddon et al., 2019; Forsell et al., 2016).

This is particularly important as evidence-based targets are required to develop robust NDCs and associated implementation.

A number of barriers and gaps exist in the NDCs. Barriers essentially derive from the lack of resources or support that can be financial, technological or capacity-building (Crumpler et al., 2019). These have been reflected in the conditionality of mitigation targets for many countries as well as dependencies on actions of other countries and international cooperation (Forsell et al., 2016). As mentioned above, there are considerable uncertainties in GHG mitigation estimates, especially from land use change that are required to monitor actions, and challenges around additionality (relative to a well-defined baseline) and leakage (Forsell et al., 2016). This is complicated by the fact that countries choose to use different accounting processes or do not specify them. There are therefore uncertainties in how countries estimate, project and account for emissions and removals from the LULUCF sector. There is also a lack of detail on implementation and therefore uncertainties in how well commitments can be implemented and their effectiveness in attaining targets. There are also differences in reporting frequency so far, with developing countries tending to report less often than developed (Grassi et al., 2017).

Globally, it is estimated that if these NDCs were implemented they would result in significant reductions in GHGs relative to absence of climate policies, i.e., 11 Gt $CO_2e$ $yr^{-1}$ (range: 9–15) by 2030 (Rogelj et al., 2016). The AFOLU related actions represent about 25% of this (Grassi et al., 2017) with much of this focused on a small set of countries who have provided commitments to reduce net emissions from LULUCF of about 84% of the total expected reductions by 2030. These are Indonesia, United States, Brazil, China, Ethiopia, Gabon and the Democratic Republic of Congo. There are many remaining opportunities to increase the inclusion of NBS in the NDCs and their ambition, including consideration of synergies with climate adaptation measures, to include other biomes with large potential for mitigation such as grasslands, drylands and wetlands, and much more support is required to turn conditional commitments into unconditional commitments (Seddon et al., 2019). More transparency is needed in how targets will be achieved, including specifics on LULCUF actions to reduce net emissions (Grassi et al., 2017), which is expected to emerge from the Enhanced Transparency Framework of the Paris Agreement (Crumpler et al., 2019).

**c) Timeline and steps for enhancing and upscaling land-based NBS.**

Upscaling land-based NBS presents a large opportunity for more impactful interventions and addressing of a broader range of global challenges. This requires urgent and concerted effort through the NDC framework, as well as progress in implementation at the sub-national level. Current approaches and their progress across a range of global initiatives, in particular carbon mitigation, is not happening at a scale and synergistically across countries to meet targets (Cohen-Shacham et al., 2019; Roe et al., 2021). It is estimated that only about 8 $GtCO_2$ (0.5% of total emissions) of mitigation from AFOLU orientated policies has been achieved between 2009 and 2019 (Roe et al. 2019). In some cases targets have not been met (e.g., New York Declaration on Forests (NYDF) target of restoration of 150M ha by 2020) or progress has even reversed.

To address global mitigation targets, substantially more resources and effort are required to move to the necessary scale, whilst maximizing benefits and minimizing trade-offs. This will require a transformation in ambition and commitment to develop innovations in solutions that are large-scale and coherent with policy at the national level that factor in solutions to other societal challenges. These include climate change adaptation, food and water security, biodiversity loss, human health and socio-economic development (Seddon et al., Roe et al., 2021). This requires investment, guidance, and capacity development to identify, design, implement and monitor NBS at scale within the local context of implementation (Cohen-Shacham et al., 2019). To be effective and sustainable, this needs to be done at the regional to national scale, at which plans can be developed, coordinated and prioritized (Roe et al., 2019; Crumpler et al., 2019).

Nevertheless, within such larger-scale coordination, locally adapted measures should be favoured that minimize trade-offs with other environmental priorities and community livelihoods, rather than large-scale, top-down approaches (Hanan et al., 2021). The landscape scale bridges the gap between the ambition of the global challenge and the local context, and allows for consideration of sectoral connections and trade-offs; landscape-based approaches are therefore best used. This includes understanding and working within long-established legal and governance frameworks, with account for social and cultural norms at the local level (EC, 2020). Specifically, the extent of the benefits and risks depends greatly on how the NBS measure is implemented, including the scale and type, how it intersects with other measures and sectors, and the local context in terms of the climate and biome, land ownership/tenure, etc. (Smith et al., 2019). Measures need to be adaptable over time also (Hurlbert et al., 2019). The local context is of great importance to enhancing and upscaling of NBS because of the long-standing socio-economic and social-cultural arrangements, but also because NBS need to fit within the broader landscape and connect to the communities that inhabit this (Holl et al., 2017). Local feedbacks between climate, soils and plants within their unique ecosystem are crucial for the success of land-based NBS (Hanan et al., 2021). Climate change impacts will also drive local feasibility, especially in water-limited regions through impacts on productivity and carbon storage, and therefore locally adapted solutions are best.

Given the barriers that exist for implementation, as shown by the conditionality of many NDCs, especially in developing and least developed countries, support is needed to meet climate targets (Roe et al., 2021). These needs vary greatly by country and depend fundamentally on availability of funding, but also on institutional, technological and environmental capacities and constraints, as well as socio-cultural and socio-ecological conditions and contexts (de Coninck et al., 2018). As such, more targeted information is needed on the national and regional potential and feasibility of mitigation measures

(Cohen-Shacahm et al. 2019) (see section 5a for a summary of national potential and feasibility).

Steps to upscale therefore require clarity on feasibility, identification of priorities and adequate support and guidance for design, implementation and monitoring/evaluation. Clarity is needed in the form of principles, guidelines or operational framing for upscaling NBS (Cohen-Shacham et al., 2019; IUCN, 2020), to ensure that trade-offs are minimized and unintended consequences are avoided (Seddon et al., 2020). Such principles should be focused on the operationalization of NBS, rather than general or theoretical perspectives, with evidence-based standards and guidelines for implementation. This will encourage wider uptake and remove the uncertainties that result from interpretation of the broader principles by the decision-maker or implementor.

Currently, the most comprehensive and authoritative set of global principles are the IUCN/CME definition and principles that were developed through global consultation and public review (IUCN, 2016; IUCN, 2021), and consist of eight core principles:

- Principle 1 (Conservation): NBS embrace nature conservation norms (and principles)

- Principle 2 (Synergies): NBS can be implemented alone or in an integrated manner with other solutions to societal challenges.

- Principle 3 (Site specific context): NBS are determined by site-specific natural and cultural contexts that include traditional, local and scientific knowledge.

- Principle 4 (Transparency and broad participation): NBS produce societal benefits in a fair and equitable way in a manner that promotes transparency and broad participation.

- Principle 5 (Diversity and evolvement over time): NBS maintain biological and cultural diversity and the ability of ecosystems to evolve over time

- Principle 6 (Landscape scale): NBS are applied at a landscape scale

- Principle 7 (Trade-offs): NBS recognize and address the trade-offs between the production of a few immediate economic benefits for development, and future options for the production of the full range of ecosystem services.

- Principle 8 (Policy integration): NBS are an integral part of the overall design of policies, and measures or actions, to address a specific challenge.

These NBS principles generally go beyond principles and guidelines in other ecosystem-based approaches (Cohen-Shacham et al. 2019) and as such form an umbrella approach that encapsulates most others. In particular, NBS match the scale of solution to the scale of the problem and have a strong emphasis on integration with policy. Arguably the principles could go further with consideration of, for example, adaptive management and temporal scales, effectiveness of implementation and outcomes, including monitoring and uncertainties (Cohen-Shacham et al. 2019). For example, the temporal scales are particularly important because of the often delayed timing of the benefits and the expectations on when these will materialize, which can vary across stakeholders.

Practical operational guidance based on an overarching set of NBS principles can help implementation as it should cover all contexts including diversity of experience and expertise of implementors, whether they are focused on conservation or natural resource management or something else. Guidance should provide decision-makers "an efficient and common way to understand, measure and improve the efficiency of different types of interventions" (IUCN, 2016). The IUCN has developed the Gold Standard for NBS (IUCN, 2020), which provides practical guidance on how to design effective and scalable solutions through a user guide and self-assessment tool. It helps decision makers understand the concept and what is required for successful implementation, and crucially to avoid misuse of the concept and reduce unintended consequences through consideration of the broader socio-ecological context. It can also add credibility to a local intervention when talking to investors, stakeholders and beneficiaries. The standard consists of eight criteria and 28 indicators (Table 3). The EC has also provided guidance (EC, 2021) in the form of a set of questions to help clarify whether a measure can be framed as NBS. (1) Does the NBS use nature/natural processes? (2) Does it provide/improve social benefits? (3) Does it provide/improve economic benefits? (4) Does it provide/improve environmental benefits? (5) Does it have a net benefit for biodiversity?

**Table 3. Criterion and indicators of the IUCN NBS gold standard (IUCN, 2020).**

| Criterion | Indicators |
|---|---|
| Criterion 1. NBS effectively address societal challenges | 1.1 The most pressing societal challenge(s) for rights-holders and beneficiaries are prioritised<br>1.2 The societal challenge(s) addressed are clearly understood and documented<br>1.3 Human well-being outcomes arising from the NBS are identified, benchmarked and periodically assessed |
| Criterion 2: Design of NBS is informed by scale | 2.1 The design of the NBS recognises and responds to interactions between the economy, society and ecosystems<br>2.2 The design of the NBS is integrated with other complementary interventions and seeks synergies across sectors<br>2.3 The design of the NBS incorporates risk identification and risk management beyond the intervention site |
| Criterion 3: NBS result in a net gain to biodiversity and ecosystem integrity | 3.1 The NBS actions directly respond to evidence-based assessment of the current state of the ecosystem and prevailing drivers of degradation and loss<br>3.2 Clear and measurable biodiversity conservation outcomes are identified, benchmarked and periodically assessed<br>3.3 Monitoring includes periodic assessments of unintended adverse consequences on nature arising from the NBS<br>3.4 Opportunities to enhance ecosystem integrity and connectivity are identified and incorporated into the NBS strategy |
| Criterion 4: NBS are economically viable | 4.1 The direct and indirect benefits and costs associated with the NBS, who pays and who benefits, are identified and documented<br>4.2 A cost-effectiveness study is provided to support the choice of NBS including the likely impact of any relevant regulations and subsidies<br>4.3 The effectiveness of the NBS design is justified against available alternative solutions, taking into account any associated externalities<br>4.4 NBS design considers a portfolio of resourcing options such as market-based, public sector, voluntary commitments and actions to support regulatory compliance |

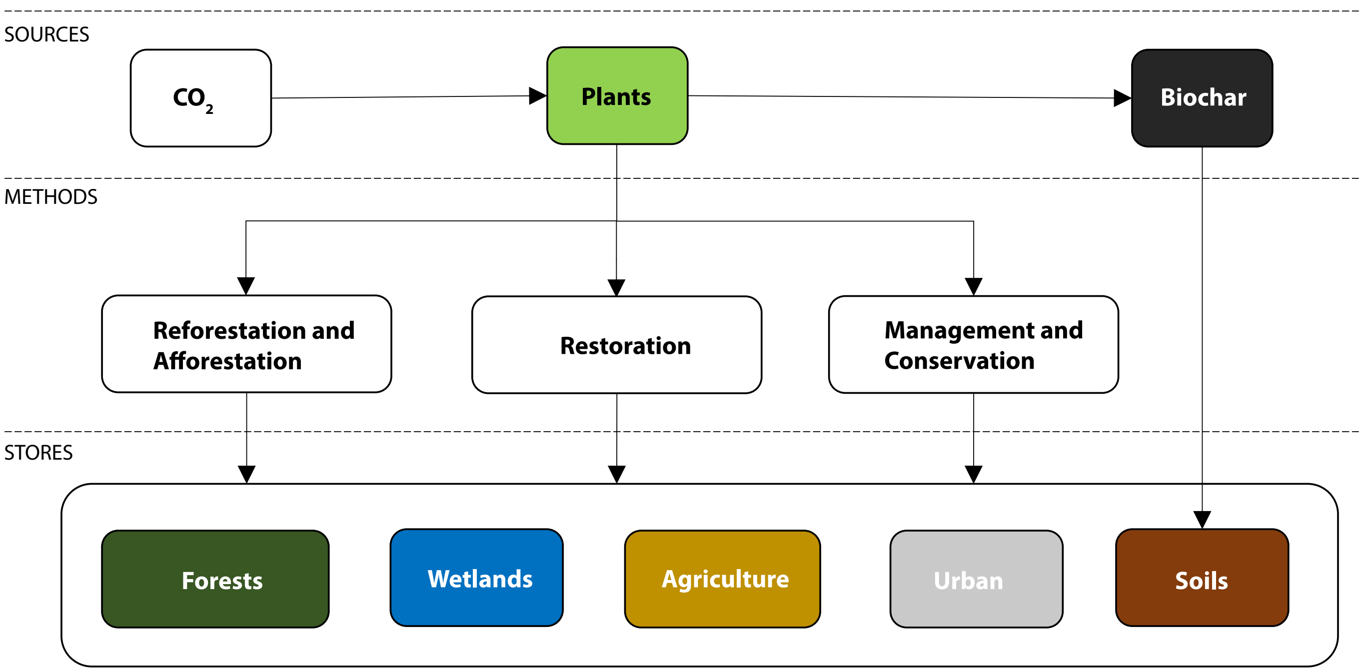
| | |
|---|---|
| Criterion 5: NBS are based on inclusive, transparent and empowering governance processes | 5.1 A defined and fully agreed upon feedback and grievance resolution mechanism is available to all stakeholders before an NBS intervention is initiated<br>5.2 Participation is based on mutual respect and equality, regardless of gender, age or social status, and upholds the right of Indigenous Peoples to Free, Prior and Informed Consent (FPIC)<br>5.3 Stakeholders who are directly and indirectly affected by the NBS have been identified and involved in all processes of the NBS intervention<br>5.4 Decision-making processes document and respond to the rights and interests of all participating and affected stakeholders<br>5.5 Where the scale of the NBS extends beyond jurisdictional boundaries, mechanisms are established to enable joint decision making of the stakeholders in the affected jurisdictions |
| Criterion 6: NBS equitably balance trade-offs between achievement of their primary goal(s) and the continued provision of multiple benefits | 6.1 The potential costs and benefits of associated trade-offs of the NBS intervention are explicitly acknowledged and inform safeguards and any appropriate corrective actions<br>6.2 The rights, usage of and access to land and resources, along with the responsibilities of different stakeholders, are acknowledged and respected<br>6.3 The established safeguards are periodically reviewed to ensure that mutually agreed trade-off limits are respected and do not destabilise the entire NBS |
| Criterion 7: NBS are managed adaptively, based on evidence | 7.1 A NBS strategy is established and used as a basis for regular monitoring and evaluation of the intervention<br>7.2 A monitoring and evaluation plan is developed and implemented throughout the intervention lifecycle<br>7.3 A framework for iterative learning that enables adaptive management is applied throughout the intervention lifecycle |
| Criterion 8: NBS are sustainable and mainstreamed within an appropriate jurisdictional context | 8.1 The NBS design, implementation and lessons learnt are shared to trigger transformative change<br>8.2 The NBS informs and enhances facilitating policy and regulation frameworks to support its uptake and mainstreaming<br>8.3 Where relevant, the NBS contributes to national and global targets for human well-being, climate change, biodiversity and human rights, including the United Nations Declaration on the Rights of Indigenous Peoples (UNDRIP) |

# Land-based nature-based solutions (resources, management and specificities)

### a) Overview of global extent and trajectories

Measures to mitigate climate through NBS are focused on reducing emissions or removal of CO2 from the atmosphere. Emission reductions can generally be achieved by land conservation, restoration or enhancement. Removal of CO2 from the atmosphere can be done via a range of measures that includes reforestation, forest management, agroforestry, cropland management, soil management and storage including biochar, wetland and peatland restoration, and urban applications (Belamy and Osaka, 2020) (Figure 6). This section provides an overview of the current global extent and trajectories of different measures as well as specifics for a set of measures that have the most traction currently or are likely to be taken forward in the future.

**Figure 6**. Pathways for land-based NBS carbon mitigation.



## Land-based nature-based solutions (resources, management and specificities)

Trajectories globally are based on pledges made by countries, often under multilateral agreements for climate mitigation, sustainable development and ecosystem restoration (Mackey et al., 2015). These include the NDCs and ecosystem restoration plans under the Bonn Challenge (Verdone and Siedl, 2017), and more recently the 2021 Glasgow Forest Declaration to collectively "halt and reverse forest loss and land degradation by 2030 while delivering sustainable development" (UNFCCC, 2021). This is emphasized by long-term global frameworks (e.g. REDD+), re-statement and reinvigoration of goals (e.g., the Bonn Challenge of 2011 and reaffirmations via the 2014 NYDF and the Glasgow Forest Declaration at CoP26 in 2021), recently begun initiatives (e.g. UN Decade of Ecosystem Restoration) and future promised goals within NDCs, as well as many regional initiatives of the Bonn Challenge: Initiative 20x20 in Latin America, AFR100 (the African Forest Landscape Restoration Initiative), Regional Mediterranean Initiative, and other regional initiatives for Southeast Asia and Central Asia. There is also a significant number and diversity of national and sub-national initiatives and stakeholder processes that cut across government, civil society, NGOs, indigenous communities, and the private sector. Most of the international to sub-national initiatives are focused on forestry (Table 4), as forest restoration provides a major opportunity for climate mitigation (GNF, 2019).

**Table 4.** Summary of international land use initiatives aimed at carbon emissions reduction and storage. (See here for detailed regional initiatives https://www.fao.org/in-action/forest-landscape-restoration-mechanism/our-work/regional/es/)

| Land Use Initiative | Year | Pledge, commitment or statement |
|---|---|---|
| Bonn Challenge | 2011 | Initially pledged to bring 150 million hectares of deforested and degraded land into restoration by 2020. 60 countries have pledged to restore 210 million hectares (800,000 square miles) of degraded ecosystems, with the ultimate goal of restoring 350 million hectares by 2030. |
| New York Declaration on Forests | 2014 | Calls for the end of natural forest loss and the restoration of 350 million hectares of degraded landscapes and forestlands by 2030. It was endorsed by nearly 200 governments, multinational companies, Indigenous Peoples, and civil society organizations. Adopted by Bonn Challenge. |
| Land Degradation Neutrality | 2015 | AS PART OF SDG 15.3 “LAND DEGRADATION NEUTRALITY” (LDN) STRIVES TO ACHIEVE A LAND DEGRADATION-NEUTRAL WORLD. ADOPTED BY THE UN GENERAL ASSEMBLY IN 2015. |
| Glasgow Forest Declaration | 2021 | In 2021, at the UNFCCC COP26 in Glasgow, leaders from around the globe announced the Glasgow World Leaders Declaration on Forests and Land Use – a commitment aligned with NYDF to end global deforestation by 2030, endorsed by over 140 countries and covering roughly 90% of global forests. |
| “4 per Mille” | 2015 AT COP21 IN PARIS | Annual 4‰ (0.4%) increase in global agricultural soil organic carbon stocks. If applied to all (also non-agricultural) soils, such a C sequestration rate could in theory fully compensate increases in atmospheric CO2 levels of 4300 Tg yr–1. |

| Regional Land Use Initiative | Year | Pledge, commitment or statement |
|---|---|---|
| Initiative 20x20 | 2014 at COP20 in Lima | Country-led effort seeking to change the dynamics of land degradation in Latin America and the Caribbean by beginning to protect and restore 50 million hectares of forests, farms, pasture, and other landscapes by 2030. The initiative supports the Bonn Challenge and the New York Declaration on Forests, global commitments to bring 350 million hectares of the world's deforested and degraded land into restoration by 2030. So far, 18 Latin American and Caribbean countries and three regional programs have committed to improve more than 52 million hectares of land (or about 124 million acres, an area roughly the size of Paraguay and Nicaragua combined) through Initiative 20x20. |
| Pan-African Program | 2007 | Launched in 2007 by the African Union and named the Great Green Wall of the Sahara and the Sahel Initiative (GGWSSI) to reverse land degradation and desertification in the Sahel and Sahara, boost food security and support local stakeholders to adapt to climate change. |
| AFR100 (African Forest Landscape Restoration Initiative) | 2015 AT COP21 IN PARIS | 100 million hectares of land in Africa into restoration by 2030 |

However, monitoring and evaluating the implementation and success of these initiatives and the pledges within is difficult and uncertain (see section 6), and therefore constrains our ability to effectively understand and plan for future investments. Monitoring large-scale changes from satellites is generally effective and has transformed how we track actions like reductions in deforestation (de Almeida et al., 2020). Nevertheless, accurate monitoring and evaluation requires ground measurements often over the long term, including measurements of carbon storage, as well as related benefits and impacts, such as biodiversity and local economies, which are difficult and sometimes impossible to measure via large scale monitoring from satellites. Progress requires contemporary and long-term historical estimates of net-emissions from the land-use sector (Fritz et al., 2016), and especially reduction in the large uncertainties in estimates of various sub-sectors such as deforestation and degradation, wetland and peatland degradation and conversion, and for carbon inventories.

Several barriers to tracking of progress exist, as progress is generally based on voluntary provision of data from countries. These include lack of government transparency, technical barriers around quality-controlled mapping of changes at the local scale and scaling up to national estimates, especially for biodiversity and local stakeholder involvement. In particular, this requires open data access and sharing of resources and best practices (Nabuurs et al., 2022). Much progress has been around mapping of deforestation, regeneration, planting and conservation, such as the Food and Agriculture Organization of the United Nations (FAO), and their Collect Earth platform and five-yearly Global Forest Resources Assessments (FRA), as well as various other platforms such as the IUCN Bonn Challenge Barometer and the WRI/IUCN Global Forest Watch. National efforts such as the Brazilian Restoration and Reforestation Observatory have seen significant progress, but much work is required to scale up to global monitoring. Restoration mapping is less developed than deforestation mapping because restoration is more vaguely defined and takes time to happen, often over many years, which is often beyond the time frame of investment windows. Little work is done on monitoring restoration on agricultural lands.

Progress in forest restoration within the Bonn Challenge Barometer is currently focused on 20 case study regions. The latest available report is from 2018 (IUCN, 2019) which covers the initial development phase of the Barometer with a focus on 6 countries: Brazil, El Salvador, Mexico, Rwanda, Sri Lanka and USA. 13 countries in total have reported area under restoration transition totalling 43.7 Mha, which is only about 29% of the Bonn Challenge overall target but represents about half the national commitments of these individual countries and would be equivalent to 1.4 $GtCO_2e$ sequestered.

Several global studies have estimated historic changes in emissions from LULUCF (Grissi et al., 2017; Jia et al., 2019; Roe et al., 2021). Net emissions are estimated to have decreased from 1.54 ± 1.06 $GtCO_2e$ $yr^{-1}$ in 1990 to 0.01 ± 0.86 $GtCO_2e$ $yr^{-1}$ in 2010, which is equivalent to a linear decrease of -0.08 $GtCO_2e$ $yr^{-1}$ (Grissi et al., 2017). Roe et al. (2021) estimate, based on a variety of global and national estimates, that policies have only achieved about 8 $GtCO_2$ from AFOLU or about 0.5% total emissions during that period. There is quite a bit of variability over time, however, with the influence of high deforestation rates in Brazil in certain years, and high peat fire emissions (e.g., in Indonesia in the strong El Niño of 2015/16). Freidlingstein et al. (2019) show no clear trend in $CO_2$ emissions from land use change over the most recent decade, though the data are very uncertain.

**b) Resources, management and specificities for different forms of LNBS for carbon mitigation**

**i. Afforestation projects**

There is great potential for afforestation and reforestation to tackle a large part of the carbon problem (Griscom et al., 2017; Fargione et al., 2018) (see section 5 on global potential), and this has been promoted as a key strategy for many years (Reyer et al., 2009; Pacala and Socolow, 2004) and through the various existing international to national initiatives mentioned above. Afforestation and reforestation is the establishment or re-establishment of forest areas, respectively, and are sometimes collectively referred to as "forestation" (Fahey et al., 2010). Forestation projects will generally see large gains in carbon storage in early growth but will reach a saturation point as trees reach maturity and the uptake of $CO_2$ slows down. Management is relatively low, although management approaches are important for long-lived storage and can also be used to enhance storage (see next section). Reforestation has numerous co-benefits including providing habitat, increasing biodiversity and soil fertility (including soil carbon storage – see section below), can improve water and air quality and reduce erosion and downstream flooding (Cunningham et al., 2015). Forests are often key to livelihoods, with cultural meaning and so conservation is particularly important (see next section). There are uncertainties around these co-benefits, especially in terms of the scale of effects (di Sacco et al., 2021), with evidence of benefits locally (e.g., biodiversity, water yield) but uncertainties at larger-scales (Jackson et al., 2005; Larsen et al., 2011).

Similar to other measures (e.g., wetland and peatland restoration) afforestation may compete with other land uses that it displaces with impacts on food and water security and potential biodiversity. Afforestation with faster growing species in mono-plantations can diminish biodiversity if replacing native forests, although improvements can be gained when replacing agricultural or degraded/marginal lands. Agroforestry can somewhat reduce these trade-offs (see section below). Furthermore, the land required for significant carbon storage is large, with estimates of about 0.1 ha per $tCO_2$ $y^{-1}$ over 100 years (Smith et al., 2015). There are also potential feedbacks with the climate to consider; for example, large-scale planting can affect land surface water and energy balances, with cooling effects in tropical regions and warming effects in cold regions through decreases in albedo (Betts, 2000; Alibakhshi et al., 2020).

**ii. Forest management**

Forest management is considered here as conservation of existing forests as well as improved management of existing and new forested areas (either afforested or reforested). Conservation (or avoided deforestation) as a management strategy is very important but the net emissions depend on how the avoided clearing would have been done, and what the wood is used for, as well as the subsequent land use. Management approaches should be implemented to increase growth, reduce removal of wood or reduce impacts of disturbances such as drought or pests, and generally can be cost effective without changes to land use or tenure (Rasolofoson et al., 2015). Specific examples of practices are listed in Table 5, which is based on guidance for management strategies for forest carbon management in North American forest systems as gleaned from the literature (Ontl et al., 2020; also see Griscom et al., 2017 for broad management pathways). For example, where forests are used for timber production then carbon and profit can be jointly maximized by extending harvest cycles to “carbon maturity”, albeit with reduced short-term yields, and either storing in long-lived wood products, converting to biochar (see section below) for further long-lived storage, or used for bioenergy to replace fossil fuels. The forest can then be regrown to capture further carbon from the atmosphere. Other tactics may include reducing undergrowth to promote storage in larger trees or packing carbon into a landscape with a diversity of tree structure.

Productivity can also be enhanced through use of fertilizers and irrigation, albeit with additional costs including carbon costs and the release of N2O. Although mono-planting is generally not recommended, the use of specific fast-growing or more resilient species in plantations can be beneficial in some circumstances (Liu et al., 2018). There is also a strong connection to soil carbon storage (see section below), whereby forest management activities can minimize soil disturbance to retain carbon (Jandl et al., 2007). Reduction in fire risk, with a focus on long-term retention of carbon stocks, can be done by selective removal of fuel loads or allowed burning. Finally, harvesting of forests should take full account of the how the harvested wood is used and potentially stored in wood products, or whether burning of wood replaces fossil fuel use and the associated differences in emissions. Where individual practices have significant trade-offs (e.g., reduced short-term yields), these can be minimized by co-implementing reforestation strategies such as new plantations (Griscom et al, 2017). As with afforestation and reforestation projects, co-benefits of conservation and management are numerous.

**Table 5.** Management strategies for forest carbon (from Ontl et al., 2020)

| Strategy | Specific practices |
|---|---|
| Strategy 1: Maintain or increase extent of forest ecosystems | 1.1 Avoid forest conversion to non-forest land uses<br>1.2 Reforest lands that have been deforested and afforest suitable lands 1.3 Increase the extent of forest cover within urban areas<br>1.4 Increase or implement agroforestry practices |
| Strategy 2: Sustain fundamental ecological functions | 2.1 Reduce impacts on soils and nutrient cycling<br>2.2 Maintain or restore hydrology<br>2.3 Prevent the introduction and establishment of invasive plant species and remove existing invasives<br>2.4 Maintain or improve the ability of forests to resist pests and pathogens<br>2.5 Reduce competition for moisture, nutrients, and light |
| Strategy 3: Reduce carbon losses from natural disturbance, including wildfire | 3.1 Restore or maintain fire in fire-adapted ecosystems<br>3.2 Establish natural or artificial fuel-breaks to slow the spread of catastrophic fire<br>3.3 Alter forest structure or composition to reduce the risk, severity, or extent of wildfire<br>3.4 Reduce the risk of tree mortality from biological or climatic stressors in fire-prone systems 3.5 Alter forest structure to reduce the risk, severity, or extent of wind and ice damage |
| Strategy 4: Enhance forest recovery following disturbance | 4.1 Promptly revegetate sites after disturbance<br>4.2 Restore disturbed sites with a diversity of species that are adapted to future conditions<br>4.3 Protect future-adapted seedlings and saplings<br>4.4 Guide species composition at early stages of development to meet expected future conditions |
| STRATEGY 5: PRIORITIZE MANAGEMENT OF LOCATIONS THAT PROVIDE HIGH CARBON VALUE ACROSS THE LANDSCAPE | 5.1 Prioritize low-vulnerability sites for maintaining or enhancing carbon stocks<br>5.2 Establish reserves on sites with high carbon density |

| | |
|---|---|
| Strategy 6: Maintain or enhance existing carbon stocks while retaining forest character | 6.1 Increase structural complexity through retention of biological legacies in living and dead wood<br>6.2 Increase stocking on well-stocked or understocked forest lands<br>6.3 Increase harvest frequency or intensity because of greater risk of tree mortality<br>6.4 Disfavour species that are distinctly maladapted<br>6.5 Manage for existing species and genotypes with wide moisture and temperature tolerances<br>6.6 Promote species and structural diversity to enhance carbon capture and storage efficiency<br>6.7 Use seeds, germplasm, and other genetic material from across a greater geographic range |
| Strategy 7: Enhance or maintain sequestration capacity through significant forest alterations | 7.1 Favour existing species or genotypes that are better adapted to future conditions<br>7.2 Alter forest composition or structure to maximize carbon stocks<br>7.3 Promote species with enhanced carbon density in woody biomass<br>7.4 Introduce species or genotypes that are expected to be adapted to future conditions |

**iii. Agro-ecosystems**

Agroecosystems cover a broad range of possibilities for land-based NBS, but are generally under-utilized compared to other NBS actions, and not accounted for in systematic ways in global or national accounting (Zomer et al., 2016; Skole et al., 2021). Given the large contribution of agriculture to the carbon problem (~24% of global emissions), and continued conversion of forests to agricultural land, especially in the tropics, significant progress is required to understand how to reduce net emissions including the use agroecosystem approaches. Simelton et al. (2021) define NBS in the agricultural sector as "the use of natural processes or elements to improve ecosystem functions of environments and landscapes affected by agricultural practices, and to enhance livelihoods and other social and cultural functions, over various temporal and spatial scales". They perform a number of functions (Simelton et al., 2021) that can be categorized as 1) sustainable production processes; 2) green infrastructure to naturally engineer the landscape; 3) amelioration focused on restoration, including carbon mitigation; and 4) conservation aimed at maintaining ecological health. Agroecosystems play an important role in soil carbon storage (see next subsection).

In general, agroecosystems are focused on tree-based agricultural production systems (agroforestry) that include tree cropping, alley-cropping, silvopastoralism, tree-based energy farms, shelterbelts, riparian buffers, community woodlots, and scattered individual trees (Skole et al., 2021) but span a range of other approaches and systems including the use of hedgerows (Hernández-Morcillo et al., 2018; Drexler et al., 2021) that are further under-appreciated. Tree-based and other approaches are embedded in many traditional farming systems because they provide a range of direct and indirect benefits, such as providing food, fibre and fuel, retention of water that can increase productivity and protection from floods, increased soil fertility including abundance of N-fixing bacteria, provision of habitat and conservation of biodiversity. Trees can also improve micro-climates which can stabilize soil organic carbon and fertility. In many of these respects, there is a strong contribution to climate adaptation. For smallholder farmers, in particular, they provide diversity and resilience in their farming systems, and opportunities for higher economic provision compared to annual crops (Lasco et al., 2014).

**iv. Soil management**

The total amount of carbon storage in soils globally is about 2500 GtC, which is a significant proportion (~80%) of the terrestrial biome storage (3170 GtC) (Lal, 2018), and is about three times the size of the atmospheric carbon store. Soil carbon comprises organic and inorganic carbon, which are about 1,505 GtC and 950 GtC, respectively, globally (Lal, 2018; Batjes, 1996, 2014). Soil organic carbon (SOC) is lowest in drier climates, where inorganic carbon is more prevalent and important, higher in temperature climates and highest in tropical wet climates and peatlands. In the majority of soils, organic matter comprises only a small fraction (< 10%) of the total mass, which is made up of grades of mineral matter such as sands, clays and silts. SOC is distributed with depth, with approximately 677 GtC within the top 0.3m globally, 993 GtC in the top 0.5-m, and 1,505 GtC to 1-m depth, and this distribution has implications for its stability and sequestration. The presence of organic carbon in soils provides a range of benefits to soil productivity through increased retention of water and nutrients (Bassio et al., 2020). It can also reduce erosion through improved soil structure, with knock-on benefits to water quality. This is associated with higher productivity in natural environments and agricultural sectors with benefits to ecosystem functioning and food security.

Depletion of organic matter in soils, through poor land management, or land degradation through mining and deforestation plays a significant role in the local carbon balance but can impact globally because of the large-scale impacts on the carbon balance. Land conversion over the past 12,000 years, has removed carbon from the soil by an estimated 135 PgC (range 115-154) (Lal, 2018). Many cultivated soils have lost 50-70% of their original carbon stock (Sanderman et al., 2017), and most cropland mineral soils have lost 30-50% of their top layer (0-30cm) organic carbon because of decreases in production and removal of harvested biomass (Lal, 2004; Paustian et al., 2019). This has also been compounded by soil disturbance and erosion, as well as nutrient depletion (Paustian et al., 1997). On grazing lands, carbon loss may be less but this is highly dependent on how they have been managed for grazing (Conant et al., 2016).

About one-third of the increase in atmospheric $CO_2$ has been derived from loss of soil organic carbon through deforestation, cultivation of land, and soil degradation and poor management (Lal, 2004). In total this has contributed about 12 PgC to the atmosphere, which is a cumulative loss of 30–40 MgC $ha^{-1}$. Soil organic carbon storage and historic loss therefore represents a major component of the global carbon cycle, and restoration and enhancement represents a large opportunity for climate mitigation. This potential is due to soils having a large capacity to store carbon and so small enhancements over large areas can make a large difference to overall carbon stocks.

The amount of organic carbon in soils is dictated by the balance between the inputs of carbon from plant residues and organic amendments such as manure and compost, and outputs that result from decomposition and are mostly $CO_2$ emissions. In wet soils, anaerobic decomposition will occur and will release carbon in the form of methane. This balance is a function of the climate, soil type, and management practices, and can be greatly affected by erosion, although this may act to either increase or decrease carbon loss depending on the local context. Inputs into the soil system depend on the type of land cover and whether it is annual or perennial, woody or herbaceous, and its productivity, where climate and nutrient availability are important (Paustian et al., 2019). Similarly, multiple factors drive the rate of decomposition, particularly soil temperature and moisture, drainage and pH (Wang et al., 2010), while additional factors affect how well organic matter is shielded from decomposition and therefore its stability. When land degradation and land use transition occur such as through deforestation and conversion to agricultural lands, carbon release from soils occurs through the decomposition via microbial activity that is not continually replaced from the removed or reduced vegetation. The rate of decomposition may be exacerbated by increased temperatures associated with removal of the vegetation canopy.

Sequestration of soil carbon can be achieved by increasing the rate of input into the system to remove $CO_2$ from the atmosphere, and by decreasing the rate of loss via decomposition and biomass removal. Sequestration is dependent on a number of factors including soil type, climate, landscape position, and previous land use and antecedent stocks (Lal, 2018). It should be noted that sequestration will be most efficient in the first few decades but will diminish as the soils become saturated and reach an equilibrium state such that further gains will be much smaller (Paustian, 2014). Potential increases in sequestration are also reversible, especially if management reverts to previous practices, and therefore all carbon gains can be lost. Therefore, practices that sequester carbon need to be sustained over the long-term with important implications for governance and incentives. There are also considerations around nutrient balances, with significant amounts of N being required to maintain optimal C:N ratios in the range of 10-12 (van Groenigen et al., 2017), although better N management can occur alongside carbon management and the use of N-fixing legumes can be promoted.

A range of management practices have been shown to increase sequestration of carbon, and therefore are primarily focused on cropland and grazing lands. Often the goal will be to improve agricultural productivity especially in developing regions where food security is a challenge but can also improve livelihoods through carbon credits that pay farmers to manage land for sequestration.

Additional sequestration can be promoted in non-agricultural soils but the potential scale of this is relatively minor in comparison with that for agricultural lands. Table 6 summarizes practices that are in current general use, although are not yet used at the scale required for meaningful mitigation for a number of reasons. Less proven technologies have the potential to contribute to future sequestration of carbon in soils, or

so-called frontier technologies (Paustian et al., 2019). These include biochar, perennial grain crops and annual crops with more developed root systems (Table 6). Biochar is discussed separately in a later section. The global potential for soil carbon sequestration and for individual management practices are discussed in Section 5.

**Table 6.** Agricultural management processes that can be used to sequester organic carbon in soils and increase net removal of atmospheric CO2 (from Paustian, 2014 and Pasutian et al., 2019)

| Conventional Management Practices | Increased C inputs | Reduced C losses |
|---|---|---|
| Improved crop rotations and increased crop residues | X | |
| Cover crops | X | |
| Conversion to perennial grasses and legumes | X | X |
| Manure and compost addition | X | |
| No tillage and other conservation tillage | | X |
| Rewetting organic soils | | X |
| Improved grazing land management | X | |
| Non-conventional Management Practices | Increased C inputs | Reduced C losses |
| Biochar | X | |
| Perennial grain crops | X | |
| Annual crops with more developed root systems | X | |

**v. Management of wetlands and peatland as nature-based solutions**

Wet terrestrial environments are generally defined as environments with soils that are inundated or saturated by a high water table level, which generates unique biochemical processes and saturation adapted vegetation (IUSS Working Group WRB, 2006; U.S. EPA 2015). These include peatlands, wetlands, lakes and reservoirs. These environments play an important role in the carbon cycle primarily because anoxic conditions slow down the microbial decomposition of carbon and accumulate carbon. Anaerobic decomposition is relatively slow and relies on a more complex set of microbial processes compared to aerobic decomposition in soils (Megonigal et al., 2004). This means that wet environments have accumulated a significant amount of carbon (350-535 GtC) and represent about one-third of soil carbon storage globally (Kayranli et al., 2010) and are the biggest store of carbon on land. Wetland soils can hold at least 40% carbon, compared to typical agricultural soils which hold less than 5% carbon (Vepraskas et al., 2016). On the other hand, anaerobic conditions also produce methane and sometimes $N_2O$ as important GHGs, which in the context of carbon mitigation need to be balanced and preferably exceeded by carbon sequestration. Wetlands therefore can play an outsized role in carbon mitigation, relative to their global extent of 12M $km^2$, of which 54% is permanently inundated and 46% is temporarily inundated (Davidson et al., 2018).

Wetlands provide a wide range of other ecosystem services such as conservation of biodiversity, provision of habitats for rare species, mitigation of pollutants, and reductions of flood risk through water regulation (Kolka et al, 2018). Conversely, when wetlands are drained or degraded they will become a net source of GHG emissions. As such, restoration can stop emissions and potentially switch to a net sink by removing atmospheric $CO_2$. Restoration of wetlands carries some risks in that a land use change is involved that, for example, reduces food production in the case of agricultural lands. Overall, there appears to be good evidence for an increase in net benefits (e.g., Valach et al., 2021).

Management of wetlands is generally aimed at conservation. This relates to relieving stressors on wetland systems that alter the hydrological regime via ditches, dikes, and levees, or the occurrence of agricultural or urban land cover at wetland margins. Less obvious stressors such as agricultural drainage can lower local water tables and lead to carbon degradation over time, even though anthropogenic stressors are not obvious. Management can range from intensive management to ensure a range of ecosystem services, to preservation to retain the wetland environment with the main goal of retaining the accumulated carbon stock. In general wetlands are carbon sinks but can transition to sources under certain circumstances (Valach et al., 2021). Wetland hydrology and the relationship with carbon accumulation is complex, for example, some management processes can lead to enhanced emissions of methane whilst accumulating carbon (Salimi et al., 2021).

Peatlands are a type of wet environment characterized by their peat soils, which consist of decomposed plant materials under waterlogged conditions. Similar to inland wetlands, production is faster than decomposition and peat accumulates along with carbon. It is estimated that peatland carbon storage is about 600 GtC, which represents about 44% of all soil carbon storage, which is more than other type of ecosystem globally, and twice as much a forest biomes. Arguably, peatlands store carbon more effectively and longer than any other ecosystem (CoW, 2021b). Peatlands are global in their extent and although there are large uncertainties in regional and global coverage it is estimated that the coverage is about 4.3M km2 which is about 2.8% of the land surface (Xu et al., 2018) with large uncertainties in regions like the Congo (Dargie et al., 2017). They are mostly found in boreal and temperate regions of the Northern Hemisphere, where they mostly consist of sphagnum mosses, sedges and shrubs. They are also extensive in the tropics where they consist mostly of graminoids and woody vegetation, predominantly in southeast Asia, Central America and the Caribbean, South America, Africa, parts of Australasia and a few Pacific Islands (Leifeld and Menichetti, 2018).

As such they can play a key role in climate mitigation primarily through their preservation. Similar to inland wetlands, they also provide many other ecosystem services such as water flow regulation and biodiversity conservation (CoW, 2021b). They provide food security and fibre for supporting livelihoods and local economies, and preserve archaeological artifacts and records of ecosystem variability such as pollen records. Degradation and damage to peatlands also drives biodiversity loss, and water quality impacts (Crump, 2017). Peatland restoration is focused restoring the natural hydrological regime and the high water table to reinstate vegetative growth (Price et al., 2016). This generally involves removal and breaching of berms, river diversions, and restoration of upland wetlands. Risks are associated with the likely increase in methane emissions, although the risks tend to decline over time (Nugent et al. 2019) with an overall decrease in net emissions.

**vi. Urban components of nature-based solutions**

Urban areas play an important role in achieving global climate targets because of their extent, growth and impact on the carbon budget. Over half of the world's population (3.5 billion) lives in cities with over 5 billion expected to be city dwellers by 2030 and 70% in urban areas by 2050, with most growth in Asia and Africa (CoW, 2021a). Urban areas generate 75% of carbon emissions from two-thirds of global energy consumption, with only 3% of the land surface (Satterthwaite, 2008; Baró and Gómez-Baggethun, 2017), and by their nature provide challenges to climate mitigation and adaptation. For example, they tend to suppress and threaten ecosystems and associated biodiversity (UN, 2010; Haaland and van den Bosch, 2015; Bush and Doyon, 2019). They expand at the expense of rural landscapes with implications for carbon storage and associated ecosystem services (Breuste et al., 2015; Haase et al., 2014). They also have traditionally favoured grey infrastructure to manage environmental challenges.

With these pressures, there is growing interest in the use of green and blue spaces within urban environments, and to provide co-benefits to human wellbeing including economic and social benefits (Keniger et al., 2013; Hartig et al., 2014). A range of possible measures are available with varying levels of uptake and efficacy: green roofs, urban gardens, green spaces, city trees, community gardens, green indoor areas, green infrastructure and urban forests. Perhaps best known in the urban context are green roofs, walls and spaces (Bush and

Doyon, 2019) that serve to reduce temperatures and decrease energy consumption (Alexandri and Jones 2008; Enzi et al., 2017; Wu et al., 2021; Bowler et al., 2010) as well as improve air quality (Baró et al., 2014; Baró and Gómez-Baggethun, 2017) and provide aesthetic benefits (Nurmi et al., 2016).

**vii. Biochar**

Production of biochar through the process of pyrolysis (heating of biomass at high temperatures in the total or partial absence of oxygen) is a relatively new option in the context of climate mitigation since its first proposed use in the mid-2000s (Lehman et al., 2006; Woolf et al., 2010; Lehman and Joseph, 2015). It has a long history as a soil amendment going back thousands of years (Glaser et al., 2001). Its high carbon content of up to 70-80% provides large potential in itself for carbon storage relative to about 40% in plant residues, especially given its recalcitrant nature and long decomposition times, with potential for storage for hundreds of years in appropriate conditions (Kuzyakov et al., 2009). Modern production methods also produce gas (syngas) and liquid (bio-oil) products that can be used as fuels (Stewart et al., 2012; Yaashikaa et al., 2020). Specific technologies such as fast pyrolysis can produce bio-oil with about 10-30% production of biochar form the original biomass (Stewart et al., 2021), whilst slow pyrolysis generates more biochar but takes much longer (hours to days).

Biochar is associated with a range of co-benefits including as a renewable bioenergy source (via syngas and bio-oil) (Crombie and Mašek, 2015). Its long history in low intensity agricultural systems indicates its use for soil fertility and degraded land improvements with implications for sustainable food security for poor communities (Whitman et al., 2009). This in turn can increase productivity, further increasing carbon uptake by plants. It lowers the rate of losses of nutrients due to runoff and erosion, and at the same can improve water retention of soils, which is especially relevant in water scarce regions. It may also reduce net soil emissions of N2O and CH4 (Lehmann et al., 2006; Woolf et al., 2010). Furthermore, it can be produced from a range of biomass types, most notably waste from a variety of sources, including agricultural residues, biomass crops and agroforestry (Woolf et al., 2010), providing diversity in its geographical and sectoral source.

Biochar production is a mature approach in terms of its production and sequestration in soils given that it can be produced at local scales by farmers as has been done for centuries, and also at industrial scales (Woolf et al., 2010). There are nevertheless challenges to implementation. For example, careful management is required to ensure that the biochar is undisturbed and that the storage is not reversed. There are inherent challenges in this because of difficulties in measuring the amount of biochar and its carbon equivalent in the soil and tracking this over time. How the biochar interacts with the soil system and ultimately how well the carbon can be retained depends on the conditions of production such as the type of feedstock and the type of pyrolysis. There is also an upper limit to the amount of biochar that soils can store (Smith, 2016b). Furthermore, there are a range of limitations and constraints to its production, transport and use. Of these, of particular note is the sustainability of biomass production and appropriation, which is currently at 24% of net primary production (Haberl et al., 2007).

# Global potential of Land-based nature-based solutions

**a) Global estimates (and uncertainties including atmosphere exchanges and variability) of avoided emissions from conservation of terrestrial ecosystems and carbon removal from land-based NBS restoration and creation strategies.**

The potential for land-based NBS to play a significant role in carbon mitigation is gathering interest in science and policy circles, yet much more work needs to be done to understand this potential, including better estimates of the direct carbon benefits but also the co-benefits and trade-offs, costs and cost effectiveness, risks and the technological, economic and social barriers that need to be broken down. Headline stories of the potential for planting trillions of trees help to raise awareness of this potential, but such large-scale approaches need to be tempered by the technical, social and economic feasibility and an understanding of options to maximize co-benefits and minimize trade-offs (Humpenöder et al., 2018; Smith et al., 2020). Account also has to be taken of the considerable uncertainties in how benefits and trade-offs change over time, and how to design measures that are resilient and adaptable to changing pressures and circumstances, including climate change.

To put the potential of land-based NBS in perspective, it is important to relate this to the current magnitude of the components of the global carbon budget (see section 2) and their relationship to net emissions. The land use sector (agriculture, forestry and other land use) is a small part (~20%) of overall emissions at 1.5 $PgCO_2$ y-1 but consists of much larger source and sink components (Griscom et al., 2017; IPCC, 2019). The sink uptake by the biosphere is about 9.5 $PgCO_2$ y-1 which is about 30% of anthropogenic fossil fuel emissions. This is offset by land use change emissions of about 4.9 $PgCO_2$ y-1 (mostly forestry with deforestation partially offset by afforestation and reforestation and other land use activities) and agricultural activities of 6.1 $PgCO_2$ y-1. These large numbers imply that there is great potential to intervene via land use change and practices to increase the sink and decrease the source components and have a significant impact of global net emissions.

Given this, the potential for land-based NBS to contribute to climate mitigation needs to be assessed to design policies for implementation and upscaling. Potential can be estimated at the technical level, i.e., what is feasible given current technological approaches and understanding, what is feasible given safeguards to ensure no or limited impacts on biodiversity, food and water security and other ecosystem services, and what is practically feasible, given financial and human resources, and limitations of governance, socio-cultural and legal constraints and barriers. Potential based on cost effectiveness is particularly important as this relates to how willing the public are to pay for climate mitigation.

Various estimates of the global potential of NBS to provide emissions reductions and removals range between 5 – 11.7 $GtCO_2$ yr-1 by 2030 and 10 – 18 $GtCO_2$ yr-1 by 2050 (Griscom et al., 2017; Jia et al., 2019; Roe et al., 2019; UNEP, 2017; IUCN, 2021). This represents about 20-30% of the mitigation needed to keep global temperature increases below 1.5oC (Roe et al., 2021). These global estimates also include the potential of coastal and marine NBS which are not the focus of this report but have a small contribution at < 5% globally. There are substantial uncertainties in these estimates because of assumptions about land availability, effectiveness of methods, time to deployment, costs and so on. These studies vary considerably based on their accounting methods, categorization of NBS approaches and focus on type of potential, and so are only broadly comparable. Other estimates are not specific about time periods but are of the order of 7.4 – 57 $GtCO_2$ yr-1 for technical potential (Royal Society, 2018) that does not consider socio-economic barriers for implementation.

Roe et al. (2021) estimate a global cost-effective potential of 13.8 ± 3.1 $GtCO_2$ yr-1, across 20 mitigation activities, where cost-effectiveness is based on a threshold of \$100 / $tCO_2$. This is about 40% of the technical potential. Griscom et al. (2017) estimate the cost-effective potential to be 11.3 $GtCO_2$ y-1 which is about 48% of their estimated maximum potential. Other estimates include the IPCC-AR5 economic potential based on land use activities of 7.2-10.6 $GtCO_2$ yr-1 in 2030 (Smith et al., 2014). These figures compare well with what is estimated as necessary to keep within global warming targets (Griscom et al., 2017; Erb et al., 2018). Roe et al. (2019) estimate that reductions of 10-15 $GtCO_2$ yr-1 between 2030 and 2050 in net emissions are required from the land sector (20-30% of total) to keep below 1.5°C warming. However, further work is required on a national basis to ensure that these estimates are in line with what is feasible at the sub-national level (e.g., Griscom et al., 2019).

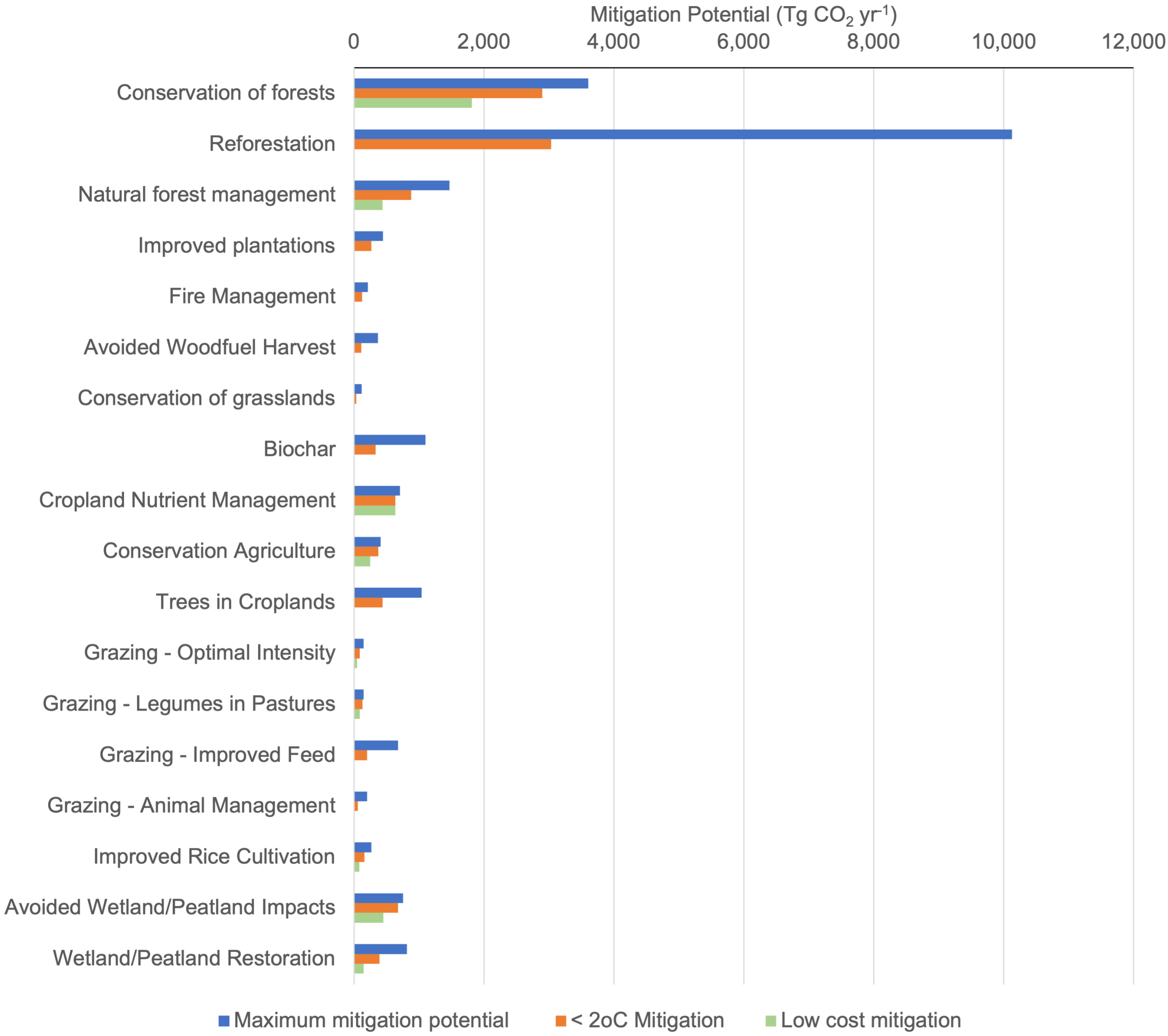
Global potential estimates broken down by type of land-based NBS are shown in Table 7 and Figure 7. This splits out management operations for forests into natural forest management, improved plantations, fire management and avoided woodfuel harvest, following Griscom et al. (2017). Agricultural management includes cropland and grassland and is broken down into major categories. The majority (two-thirds) is expected to come from afforestation and reforestation projects, with about one-quarter from croplands and grassland and the remainder from wetlands/ peatlands. By comparison, coastal and marine based projects have potential to deliver about 5% of the land based potential (IUCN, 2021).

**Table 7.** Agricultural management processes that can be used to sequester organic carbon in soils and increase net removal of atmospheric CO2 (from Paustian, 2014 and Pasutian et al., 2019)

| Land-based NBS pathway | Maximum mitigation potential (TGCO2E YR-1) | Uncertainty (95% confidence bounds) (+- TGCO2E YR-1) | < 2OC Mitigation (TGCO2E YR-1) | Low cost mitigation (TGCO2E YR-1) |
|---|---|---|---|---|
| Conservation of forests | 3,603 | 2,999 - 4,209 | 2,897 | 1,816 |
| Reforestation | 10,124 | 2,727 - 17,867 | 3,037 | 0 |
| Natural forest management | 1,470 | 921 - 8,224* | 882 | 441 |
| Improved plantations | 443 | 168 - 1,009* | 266 | 0 |
| Fire Management | 212 | 166 – 411* | 127 | 0 |
| Avoided Woodfuel Harvest | 367 | 326 - 407 | 110 | 0 |
| Forest subtotal | 16,219 | 11,291- 28,133 | 7320 | 2257 |
| Conservation of grasslands | 116 | 75 – 373* | 35 | 0 |
| Biochar | 1,102 | 642 - 1,455 | 331 | 0 |
| Cropland Nutrient Management | 706 | 399 - 959 | 635 | 635 |
| Conservation Agriculture | 413 | 310 - 516 | 372 | 248 |
| Trees in Croplands | 1,040 | 469 - 1,855* | 439 | 0 |
| Grazing - Optimal Intensity | 148 | 148 - 699 | 89 | 45 |
| Grazing - Legumes in Pastures | 147 | 14 - 1,500* | 132 | 88 |
| Grazing - Improved Feed | 680 | 35 - 1,014 | 204 | 0 |
| Grazing - Animal Management | 200 | 75 - 214 | 60 | 0 |
| Improved Rice Cultivation | 265 | 227 - 319 | 159 | 80 |
| Agriculture & Grasslands Subtotal | 4,817 | 4,398 - 6,926 | 2456 | 1095 |
| Avoided Wetland/Peatland Impacts | 754 | 237 - 1,212* | 678 | 452 |
| Wetland/Peatland Restoration | 815 | 705 - 2,471 | 394 | 149 |
| Wetland/Peatland Subtotal | 1569 | XXX | 1072 | 601 |

***expert elicitation**

**Figure 7.** Global mitigation potential of different forms of land-based NBS and other agricultural management activities (from Griscom et al., 2017) broken down into maximum technical potential, what is needed as a contribution to keeping to a 2oC warming target, and what is feasible at low cost.



**Potential of afforestation and reforestation**

Several studies have estimated the large-scale potential based on current forest distributions (e.g., Bastin et al., 2019; Lewis et al., 2019). However, careful identification of locations is necessary to scale-up afforestation and ensure sustainability and long-lived storage, including assumptions about land availability, management practices, and biophysical constraints on forest types (Royal Society, 2018). Sustainability relates to resilience to climate change (e.g., more severe and longer drought) and other factors (e.g., pests, fire) that may be exacerbated by climate change, which generally points to the use of native and diverse planting. With account of land use to safeguard future food and fibre needs these estimates vary widely from about 3 – 18 $GtCO_2$ y-1 which is mostly driven by available land which is estimated to range from 350 to 1780 Mha (Griscom et al., 2017), where upper limits include reforesting all grazing lands.

The largest challenge is identifying suitable and useable land, which requires the building of consensus among land-owners, the public and other stakeholders who have interests in land use change, its co-benefits and drawbacks (Yamanoshita, and Amano, 2012; Di Sacco et al., 2021). Landowners would particularly benefit from assurances around the payments between planting and potential harvesting at maturity. There is also a danger that emphasis on large-scale afforestation overlooks the many benefits and uses, meanings and effects on livelihoods (Oldekop et al. 2020). Benefits are generally maximized if projects are implemented on marginal or degraded land. There are also risks around the time scales of sequestration as maximum sequestration rates are attained after 10 years and maximum potential saturates after about 20-100 years all depending on the species. Other factors such as climate changes, disturbances, and management can all affect the potential and how it is reached over time.

**Potential of forest conservation and management**

Preservation of forests provides a large contribution potential, second only to reforestation and afforestation, given the large extent and storage of forests, especially in the tropics. Estimates are of 3.6 $GtCO_2$ y-1 (3 - 4.2) for conserving existing forests globally, which is about 20% of the total technical potential. It also has the highest mitigation density per hectare of 320 $tCO_2$ ha–1. Estimates of global potential of forest management are around 1-2 $GtCO_2$ y-1 (Griscom et al., 2017), with much of this derived from selective and reduced logging practices.

The significant potential of forests in land-based NBS mitigation may be undermined by climate risks (Hof et al., 2017) from overall warming and more extreme temperatures, longer and more severe droughts (including multi-year droughts) and disturbances such as wildfires, pests and pathogens (Anderegg et al., 2020). Natural forests will tend to be more resilient and will continue to store carbon with adequate resources, but management to maintain and potentially increase resilience is also very important, as well as careful design of afforestation projects as noted previously. Monoculture plantations and commercially logged forests have a role to play but will store less carbon in the long-term and have trade-offs to consider, including account of the carbon emissions associated with land use and management operations. Careful design and management are needed, especially to ensure resilience to climate change and disturbances because of their inherent lower biodiversity, genetic diversity, and structural complexity relative to natural forests (Feng et al., 2022). Overall, there remain considerable uncertainties about the carbon storage potential of management, because of how to incorporate the multiple overlapping impacts of different management activities.

**Potential of agroecosystems**

Agroforestry is prevalent across developing regions and generally increasing (Miller et al., 2017), where traditional farmer-managed systems provide a range of benefits including for climate adaptation (Mbow et al., 2014). Even within desert and degraded savannah regions, the prevalence of trees outside of forests is remarkable (Brandt et al., 2020), with much of this actively managed by farmers in agricultural systems. As these systems are embedded in traditional practices, there is scope to significantly upscale their mitigation potential (Skole et al., 2021). Although there remain large uncertainties, recent studies (e.g. Zomer et al., 2016) and new approaches using satellite remote sensing and deep learning (e.g. Brandt et al., 2020) can now start to document the prevalence and carbon storage of trees outside forests and in agroforestry, and contribute to identifying the potential for increasing this globally.

The estimated biomass carbon storage in agricultural lands is likely much higher than previously thought and as it is not included in global and national inventories there is great potential to leverage its potential (Nair, 2012; Mbow et al., 2014) (see section X). Current estimates (e.g., Zomer et al., 2016) based on the global area of agricultural land of 22.2 million km2 (GLC2000; Bartholomé and Belward, 2005) is 45.3 PgC in above- and below-ground biomass carbon on agricultural land, compared to standard IPCC agricultural land estimates of 11.1 PgC (IPCC Tier 1 global estimates; Ruesch and Gibbs, 2008) which do not consider trees, and a density of about 20 tC ha-1, which is 5 times the standard estimate, albeit with substantial regional differences. There has also been an increase of about 2 PgC from 2000 to 2010 with much of that coming from increased tree cover. Interestingly this rate of increase of 0.2 PgC yr–1 compares with estimates of above-ground carbon loses due to tropical deforestation of 0.6–1.2 PgC yr–1. There is therefore high potential to increase the role of agroecosystems and particularly the use of trees in reducing net emissions globally, with higher potential at national scales. This is especially the case given the extent of agricultural lands and the integration of trees in traditional farming systems. As with forestation, there are multiple co-benefits (including soil carbon sequestration – see next). Trade-offs may be significant and especially if not managed well, including impacts on food production (e.g., through shading and displacement of land) and local hydrology.

**Potential of SOC sequestration**

Estimates of potential sequestration of carbon in soils, most of which are agriculture, is of the order of 1B tons of carbon each year (Paustain et al., 2019), that is 20 Pg C in 25 years. Much of this could be done through the range of improved farming management processes as well as restoration of marginal lands (Lal, 2004). The potential for sequestration using conventional practices (Table 7) is large, although implementation at large-scale is challenging for a number of reasons (see later). On the other hand, soil carbon approaches have the advantage over other land-based NBS, in that the largest depletion of soil carbon stocks is on agricultural lands and no land conversion is necessary to carry out sequestration.

Improved crop rotations and cover cropping increase the supply of organic carbon to soils and include use of high-residue crops, seasonal cover crops, continuous cropping and use of perennial grasses. These have a potential sequestration of 0.3 to at least 0.1 tC/ha/y (Paustain et al., 2019; Poeplau and Don, 2015). Manure and compost additions increase carbon content directly but almost via improving productivity and crop residues. Quantifying potential is difficult because of uncertainties in the source of amendments and requires a full life cycle assessment. However, reported values from case study sites indicate values of the order of 1-3 tCO2e ha-1 y-1) and higher values (~8 tCO2e ha-1 y-1) with account for compost source. No or low-till practices have primary benefits in reducing soil erosion but also provide stabilization of soil aggregates which slows down decomposition of organic carbon (Ogle et al., 2019). Global estimates indicate increases of 0.1 and 0.22 tC ha-1 y-1 in dry and humid climates, respectively (Six et al., 2004). The potential is highly dependent on the background organic content, soil type and climate, which will dictate how much extra can be sequestered, and in some circumstances can minimize sequestration. Conversion of croplands to perennial vegetation (grasses and trees) will tend to increase carbon storage and reduce soil disturbance (Post et al., 2000). Often this is done through land retirement or “set-aside”. Global estimates are about 0.9 tC ha-1 y-1 and can approach native carbon stock over time (Conant et al., 2016). Rewetting of previously drained organic soils such as peatlands, does have mitigation impacts on CO2 and N2O emissions, with relatively lower increases in methane emissions, but the potential to do this at scale is limited because of the relatively small areal extent that has been drained for agriculture (a larger potential is for restoration of degraded peatlands as discussed below). Management of grazing lands can enhance carbon stocks through reducing biomass removal from grazing or increasing forage production. Estimates of sequestration rates are context specific and variable, ranging from ~0.1 to less than 10 tC ha-1 y-1 (Conant et al., 2016; Morgan et al., 2010; Teague et al. 2011). Of the non-conventional practices, switching to perennial grain crops could reach of the order of 1 tC ha-1 y-1 over a number of years, and use of crops with better developed root systems could produce 0.5 Gt CO2 ha-1 y-1 (Paustian et al. 2019).

Overall, the global technical potential for SOC sequestration using conventional practices is of the order of about 1-3 GtC y-1, with different meta-estimates ranging from 1.4-3.4 GtC y-1 (Lal, 2018) and 1-1.5 GtC y-1 (Paustian et al., 2019). These practices can be focused on hotspots of potential sequestration, which have the highest storage deficit below saturation, such as degraded and depleted soils. The areal extent of feasible sequestration is estimated as 4,900 Mha of agricultural land, 400 Mha of urban land and ~2000 Mha of degraded lands (Lal, 2018). If non-conventional technologies are adopted at full scale then the global potential could increase to 8 GtCO2 y-1, although with high uncertainty (Paustian et al., 2016).

A number of non-technical barriers exist for implementation of conventional practices at the scale necessary to have a significant global impact. Firstly, hundreds of millions of farmers worldwide would need to implement possible a range of practices meaning changing the way they farm for decades to centuries come, which is likely infeasible without transformative incentives and support (Smith et al. 2005; Stockmann et al., 2013; Searchinger, 2019). Secondly, climate change will itself have a large impact on soil carbon sequestration potential, particular warming which will increase decomposition rates and create a positive feedback with further warming (Melillo et al., 2017). For non-conventional practices, although there is evidence of a positive contribution to sequestration, there are significant technological and/or economic barriers, and especially to sustain long-term storage. For example, yields of perennial grain crops are 5-10 times lower than conventional grains and so the economic feasibility is questionable (Paustian et al., 2019). Much more research is required to understand the feasibility of non-conventional practices and their scalability, which is especially challenging as results may take years to decades to measure.

**Potential of wetland and peatland restoration**

Wetland and peatland restoration is a key element of climate change mitigation (Humpenöder et al., 2020), especially as the land area is large, and also relative to tidal wetlands (blue carbon) (Nahlik and Fennessy, 2016). Globally, peatlands are relatively intact, with about 3M km2 of near natural peatland. They sequester 0.37 GtC y-1, but with large year to year changes due to climate variability, with for example, tropical peatland fires in Indonesia in 2015 driven by ENSO, emitting nearly 11.3 MtCO2 per day (Huijnen et al., 2016). Peatland loss through drainage is estimated to cover about 65M ha mainly in the US and Europe for agricultural use. This is only about 0.3% of the land surface but 15% of known peatland area (Joosten et al. 2016) and contributes about of 1.9 GtC annually (with cumulative emissions of about 81 GtC; Leifeld and Menichetti, 2018), which is equivalent to 5% of global anthropogenic GHG emissions (Günther et al. 2020).

The technical potential for mitigation via restoration of degraded and drained wetlands and peatlands is of the order of 10% of global emissions. The feasible potential would be lower because of the trade-offs with current land use such as agriculture or housing. The potential mitigation via preservation of wetlands is also very high (Nahlik and Fennessy, 2016), especially for high carbon dense environments. Peatland conservation and restoration has begun in certain regions (e.g., Krauss et al., 2021). However, there are significant challenges to upscaling this to the necessary scales to have meaningful global mitigation impact. For example, 50M ha of drained peatlands will have to be restored (Humpenöder et al. 2020), which is over half of drained peatlands in agricultural use. The main challenge is therefore the compensation for the lost economic benefits of the drained peatland, although some efforts are being made to transition to livelihoods based on wet agriculture (paludiculture).

Certain risks are associated with restoration, especially with drought and fire, which are expected to become more frequent and severe with climate change, again posing the threat of positive feedbacks with atmospheric CO2 and climate. However, wetlands are generally quite resilient in the face of short-term shocks. Leakage is a concern whereby restoration in one area leads to increased pressure on other wetlands, for example to further conversion to agricultural lands given demand (Doelman et al., 2020). Proper accounting at the national scale should be done to avoid overestimating mitigation impacts. Often leakage can be reduced by focusing restoration on marginal agricultural lands where potential food production losses are low.

**Potential for Urban-Based Mitigation**

Although carbon mitigation in urban areas is considered in most applications and studies (Haase et al., 2014), its potential is small because it represents a small fraction of overall carbon mitigation targets and pales into insignificance relative to overall urban emissions. Nevertheless, urban vegetation does offset emissions through sequestration to some extent (Nowak et al., 2013b) as well as reducing emissions through their impacts on micro-climate through shading and evapotranspiration which will tend to cool the local environment (McPherson et al., 2013).

Estimates of carbon offsetting by vegetation in urban areas is dependent on the spatial scale and is generally less than 5% of annual city emissions and sometimes as low as < 1% (Baró and Gómez-Baggethun, 2017). In this sense, NBS can play a role in urban climate plans, but should be considered in the context of their other ecosystem service benefits and potentially coupled with offset activities outside of urban areas (Gebre and Gebremedhin, 2019). Overall, the mitigation potential of urban greening is low and is concentrated at the local scale, with low scope for scaling up (Baró and Gómez-Baggethun, 2017). There are considerable uncertainties, however, in current estimates of sequestration potential especially as regards tree growth in often constrained urban environments, and usually a lack of consideration of full life cycle carbon assessments including maintenance. There are also considerations around urban soils which can act as sinks or sources depending on the type and management (Pouyart et al., 2006; Velasco et al., 2016), as well as other urban, and peri-urban environments such as urban riparian forests and wetlands (Haase, 2017).

**Potential for biochar**

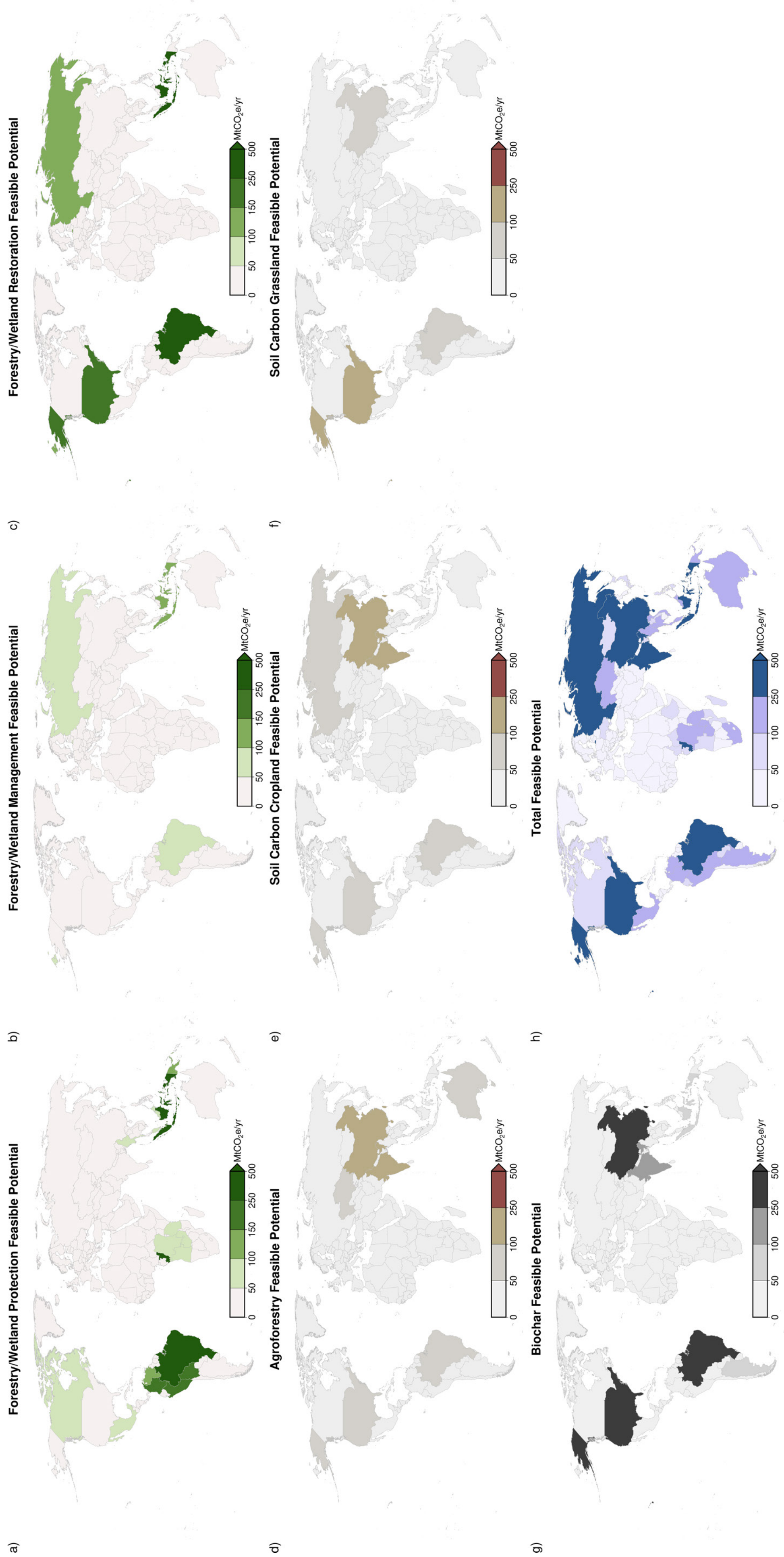
Estimates of the potential of biochar that is sustainable and technically feasible is 1.8 – 4.8 $GtCO_2$ $y^{-1}$ (Woolf et al., 2010; Smith, 2016b), which is about 12% of current (2010) anthropogenic emissions in equivalent $CO_2$. Sustainability refers to minimizing impacts on food security, biodiversity and soil conservation (Woolf et al., 2010). Much of the uncertainty resides in the rate of decomposition of different types of biochar that are produced, which is a function of the biomass feedstock type and the use of fast or slow pyrolysis (Royal Society, 2018). The potential is also highly dependent on the location (Woolf et al., 2010), such that the most effective use is where biochar fuels can be substituted for bioenergy, where either poor soils benefit from the biochar amendment or where coal is the fuel being offset. There are, however, challenges to upscaling, including the development of production facilities at scale, and the need to take into account the full life cycle of biomass production, biochar production, transport and application in the field to properly understand the potential.

Risks associated with biochar are related to reversibility of sequestration similar to organic carbon sequestration, but especially the low current uptake and limited evidence of implementation in practice. There is therefore limited policy support currently. There are also potential pollution problems from pyrolysis generation and treatment of waste products. If waste is not used for biomass feedstock, then there are risks associated with different source types such as forest degradation and competition for land from other sectors.

**Summary of national potential and feasibility**

Overall, the global technical potential is distributed spatially, based on the availability and suitability of land relative to the different measures. Figure 8 shows maps of cost-effective potential (at $100 $tC^{-1}$) for different mitigation categories, based on estimates from Roe et al. (2021). It is also highly dependent on the conditions and capacities to effectively implement the measures at the national scale, which connects to the NDCs and how they are the main vehicle for implementation at the scale necessary to meet global targets. Figure 9 shows the breakdown of mitigation potential by country and in relation to feasibility (Roe et al., 2021). Feasibility represents the enabling conditions and is estimated using a proxy that incorporates the IPCC dimensions of economic, institutional, geo-physical, technological, socio-cultural, and environmental-ecological feasibility (de Coninck et al., 2018).

**Figure 7.** Cost effective mitigation potential by country for different options, as well as their total estimated mitigation potential (data from Roe et al., 2021).



**b) Estimates of other potential NBS and SDGs benefits linked to the conservation, restoration and creation of land-based NBS.**

Given the synergies with other climate and sustainability goals, and the many co-benefits, there is potential for land-based NBS to have a significant impact on other sectors and challenges, including the SDGs (Faivre et al., 2017). In fact, it can be argued that progress in any aspect of sustainable development requires acknowledgement of the synergies across multiple sectors and goals, and that progress has been hampered by treating individual goals in isolation. The SDGs exemplify the notion of synergies and the need for cross-sectoral holistic approaches (Le Blanc, 2015; Pedercini et al., 2019; Smith et al., 2019). The SDGs are fundamentally entwined in the philosophy and approach of NBS, in that they "stress the importance of sustainable management of natural resources and the functioning of ecosystems to maintain economic activities and well-being of local communities" (Gomez Martin et al., 2020). This is explicit in many of the goals and their targets such as SDG14 which promotes the protection of aquatic resources for sustainable development, and climate action (SDG 13) with targets to mitigate and adapt to climate change, leveraging nature to do so, and SDG 15 (life on land) on conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services, with a strong focus on forest lands. NBS is also directly relevant to SDG 2 (food security), 3 (health and well-being), 6 (clean water and sanitation), 11 (sustainable cities and communities).

NBS will generally enhance the resilience of ecosystems to climate change and other disturbances and maintain or enhance a range of ecosystem services. These include adaptation to climate change, food and water security, biodiversity, disaster risk reduction, socio-economic development and human health (IUCN Gold Standard, 2021), and will act as a key selling point for investment and uptake (e.g., SDG co-benefits are being incorporated in NBS standards). Estimates of these contributions are dispersed across the literature and generally focused on case studies, but some global highlights are emerging. For example, benefits linked to water are potentially some of the highest because of the tight coupling between carbon and water, via ecosystem functioning, soils and land management. Climate change impacts will most likely be felt via water also, and so NBS that have synergies with improving water security can contribute to climate adaptation as well as mitigation. WWAP (2018) highlights the connection between increased vegetative land cover and improved land use management in providing key benefits for soils and water resources, such as reforestation in reducing soil erosion and land degradation (Eekhout and de Vente, 2022), which can negatively impact on water availability and water quality (FAO/ITPS, 2015).

On the other hand, global reforestation could increase water use (Smith et al., 2016) by approximately 1,765 m3 t–1 C y–1 (up to 1,040 km3 y–1 for removing 12.1 GtCO2 y–1) with potentially positive impacts on flood risk but negative impacts on groundwater recharge and downstream water resources availability. There can also mixed impacts on climate regulation with increases in precipitation in semi-arid regions if afforestation and reforestation are large-scale (Yosef et al., 2018). Links between NBS for carbon mitigation and biodiversity (Strassburg et al., 2010) and species conservation (Larsen et al., 2011) are highly relevant. Key regions for benefits to biodiversity conservation are in the global south, where forest conservation for carbon has high potential. However, there are significant potential trade-offs, such as with pollination if biodiversity if prioritised in some regions (Girardello et al., 2019) and a danger that biodiversity can be reduced due to afforestation in other regions (Strassburg et al., 2010). There are large concerns about the impacts on biodiversity and ecosystem services, and resilience to climate change and disturbances, if afforestation is focused on natural grassy biomes (Veldman et al., 2015).

Smith et al. (2019) summarize the synergies between land-based NBS, ecosystem services and the SDGs, highlighting the multiple overlaps that are often positive but also a range of constraints and possible negative impacts (e.g., land competition, physical climate feedbacks, water requirements, nutrient use, energy, and cost). All NBS provide positive contributions to SDG 13 (Climate Action) by design. Wetland restoration and soil carbon storage almost always provide only positive impacts and therefore represent no-regrets actions (Nagabhatla and Metcalfe, 2018; Lal, 2008). Others provide positive benefits for some SDGs of varying magnitude. A few SDGs are not directly impacted by NBS for carbon mitigation: SDGs 4 (Quality Education), 5 (Gender Equality), 10 (Reduced Inequalities), 10 (Sustainable Cities and Communities), 16 (Peace, Justice and Strong Institutions), and 17 (Partnerships for the Goals). Competition for land, water and nutrients is focused on forestry and biochar production because of the need for biomass, although this is highly dependent on the location and species involved. There is a need to understand the interactions of various NBS measures on the range of ecosystem services and SDGs, including where, and the scales at which these interactions and the synergies and trade-offs occur (WWAP, 2018; Smith et al., 2019).

**c) Potential for large scale implementation of strategies.**

Up to now there has been underutilization of NBS for meeting climate mitigation targets as well as other targets related to biodiversity and ecosystem services, despite ambition and promise. Arguably, NBS has been undervalued as a carbon mitigation approach at regional to global scales. This has led to commitments at the subnational jurisdictional level to be brought forward (Hultman et al., 2020). What is realized internationally will depend heavily on a range of socio-economic factors and importantly the commitment of governments to develop incentives relative to energy production.

Large-scale implementation of NBS strategies across regions requires overcoming a range of barriers. One of the largest barriers is the lack of evidence of NBS meeting specific needs and associated goals. Many pilot projects have been implemented, and especially in Europe (Oral et al., 2020), but more needs to be done, including in the rest of the world and developing countries in particular, as well as large-scale demonstration projects that show the feasibility at large scales, indicate the economies of scale to reduce costs, and reveal any unanticipated problems or constraints (Smith et al., 2019). This is particularly important for “frontier” approaches and technologies such as biochar that have little legacy of implementation beyond small pilot studies. Risks need to be managed, such as the potential for lower co-benefits or higher trade-offs than expected, as well as the length of time for benefits to be realized. Such evidence is needed to show that the solutions provide tangible and measurable outcomes that contribute to mitigation targets and other goals. Without this evidence, the recognition of NBS as a viable and cost-effective approach to tackling a large part of the climate problem, as well as many other goals, is missing, with implications for financing and investment. The EU (EC, 2020) has taken steps to financing research on a wide range of case studies, which should improve the evidence base, but this needs to be accelerated and connected to other initiatives globally. Work also needs to be done in areas of knowledge and data management; impact assessment (indicators); governance and business models; communication; and co-creation processes.

The high dependence on land-based measures to reach global temperature targets means that NBS will become increasingly important in international frameworks. Yet mainstreaming of NBS for mitigation and other benefits is needed in international policies and initiatives. At the EU level, this has been focused on international joint research programmes to share best practices (EC, 2020), cooperation with activities under the Covenant of Mayors initiative for Climate and Energy initiative (EU CoM, 2022) and international dialogues such as the EU-Brazil Sector Dialogue on NBS for Resilient Cities. NBS are being increasingly recognized and promoted in international agreements and policy frameworks such as the UNFCC, the Convention on Biological Diversity and by IPBES to help meet the SDGs. Mainstreaming alongside other goals such as the SDGs should be vital (Gerstetter et al., 2020).

Global efforts are driven by international initiatives and policies but have to be implemented at the national scale through individual projects. National-level potential is dependent on the technical potential but also the national capacity in terms of strength of governance and availability of financing (Griscom et al., 2017; Roe et al., 2021), with encouragement of adoption at all scales. For example, at the small farm level there needs to be encouragement and support for agricultural management, biochar application and enhancement of agroecosystems to get the widespread adoption necessary. Monitoring, verification and reporting is needed, and this requires wider adoption and application of processes for identifying NBS designs, and the standards and technologies to reliably measure sequestration at local to national scales. Policies are also required to enable technologies, especially those that are “frontier” technologies. Cross-sectoral and cross-goal benefits and synergies need to be considered, and policies developed across sectors.

Tropical nations may be best suited for the large-scale implementation of NBS, with much of this through avoided deforestation, given the magnitude of the potential impact, which could contribute to mitigation of more than 50% of national emissions (Seddon et al., 2020). Certain countries stand out (e.g., India) because of progress already made and potential to take this forward given the strength of governance and mitigation potential. As noted previously, there are likely no-regrets actions for preserving and restoring wetlands and sequestering carbon in soils.

# Assessment methods

**a) Carbon accounting: Stock, net emissions, emissions reductions**

Measuring the amount of carbon emitted, stored and removed from the atmosphere is crucial to realizing the potential, in terms of determining how effective an NBS approach is, who is responsible for implementing and maintaining the approach, and who will pay for it. There are various challenges in measuring how NBS practices alter carbon storage and sequestration (VonHedeman et al., 2020) and how to measure these effects, impairing our ability to measure the efficacy of NBS and whether it is contributing to mitigation targets (Fuss et al., 2016; Royal Society, 2018; Brander et al., 2021). The expectation is that the carbon mitigation problem will be addressed at the national level through the NDCs (UNFCCC, 2015), which requires a regular "global stocktake" to understand progress toward national targets and the overall global picture. NBS actions implemented through the NDCs and their impact on carbon, therefore, need to be readily measured, reported and verified. UNFCC guidelines ask for national reporting on GHG emissions from human activities in national GHG inventories and subjected to an international review process.

Accounting approaches are uncertain and in many respects controversial, as they do not fully capture all dimensions and details of the mitigation problem and as such can provide unintended and perverse incentives (Keith et al., 2021; Brander et al., 2021). Brander et al. (2021) identify five issues related to accounting: 1) Accounting for total system-wide change in emissions/removals with the recommendation that "carbon accounting methods are needed that include all emissions and removals that change, and a counterfactual baseline is needed to estimate the change caused by the decision in question". The baseline is necessary to assure additionality of the measure to what would have happened without the measure; 2) Non-permanence of negative emissions, e.g. to natural disturbances or harvesting – i.e., different types of carbon storage may be temporary or reversible. Temporary storage may, however, in some cases be subsumed in an aggregate accounting pool, such as for selective logging which over time does not change the overall carbon storage of a forest; 3) Non-equivalence of 'no overshoot' and 'overshoot and removal'. This relates to the situation where a global warming target is not achieved, but the carbon removal required to bring back temperatures below the target are not the same as those emitted to provide the original warming, i.e., emissions and removals are not equivalent over time; 4) Accounting and incentives for negative emissions technologies. Uncertainties in accounting can sometimes create perverse incentives whereby the original carbon removal intention is not met, which can happen, for example, when emissions from biochar production are accounted in the location or country where the biomass was grown, and not where it is was combusted; and 5) Accounting for the temporal distribution of emissions/removals, whereby emissions/removals happen over periods of time and not necessarily aligned with mitigation timeframes as set by policy. Other challenges relate to accounting for leakage, when the mitigated land use is displaced elsewhere. One of the largest concerns is that current accounting practices do not account for the longevity and stability of stocks that depend greatly on the ecosystem characteristics of the system (Keith et al., 2021).

Current international accounting approaches defined by the UNFCCC (2015) for GHG accounting or ecosystem services related to climate mitigation are based on carbon flows and so do not account for the stocks that underlie the stability and resilience of systems. It is also important that there is consistency between bottom-up estimates of CO2 fluxes and associated stocks, and top-down estimates based on global atmospheric GHG concentrations (Bastos et al., 2020). Reporting carbon inventories and net GHG emissions is, however, complex for the terrestrial biosphere, land use changes and management practices (Royal Society, 2018; VonHedeman et al., 2020; Brander et al., 2021). This is because the land is both a sink and a source due to a complex combination of natural and anthropogenic factors. The terrestrial biosphere responds to changes in climate and natural disturbances that affect plant growth, whilst direct human interventions are occurring through land use changes and land management. It is quite uncertain how to tease out or disentangle these effects and therefore estimate and understand current conditions and progress (Royal Society, 2018).

The focus on carbon flows in accounting also incentivises fast accumulation of sequestered carbon, which can lead to, for example, mono-plantations of quick growing tree species. Mackey et al. (2022) note that standard accounting approaches focus on net flows which hide the high potential for (especially forest) conservation, as emissions in one sector (e.g., fossil fuel burning) are offset by reductions in another sector (e.g., uptake of CO2 by forests). Keith et al. (2021) also note that current approaches do not account for the connection between stocks and flows in ecosystems and how this provides the synergies with other ecosystem services and may hamper efforts to preserve more established forests. It is the longevity, stability and quality of the carbon stocks that are important and should be measured (Sowińska-Świerkosz et al., 2021), as well as the associated benefits from ecosystem services (Keith et al., 2021). This potentially can be addressed by use of the SEEA accounting framework which integrates carbon and environmental-economic accounting (UNSD, 2021; Vardon et al., 2018; Keith et al., 2021).

A number of different methods are used for accounting which have their own uncertainties (Grassi et al., 2018; Hewitt et al., 2016; Shi et al., 2018), and the differences between approaches and the resulting uncertainties need to be understood better to help increase credibility and transparency in assessments and therefore uptake. For example, some accounting methods only focus on above-ground woody biomass and do not take into account herbaceous plants, litter, soil or root carbon. Other approaches do, but acknowledge large uncertainties (Yanai et al., 2020). For all landscapes, and especially forests, there is much the debate about the initial condition of the landscape in terms of its carbon storage which acts as a baseline against which NBS measures need to be measured against (Mckinley et al., 2011).

Overall, operationalizing mitigation commitments at scale requires accounting approaches with common methods for assessing the performance of NBS for carbon mitigation, including indicators and methods, as well as baseline datasets (EC, 2021b). Ultimately, we would like to know whether NBS are addressing the challenges that they were designed for, and what are the co-benefits and trade-offs, and includes monitoring (observation and measurements) and evaluation (analysis and interpretation and documentation) and can include a response element that can adjust the NBS action to improve performance and co-benefits or reduce trade-offs or dis-benefits. The latter is important because performance, and perceptions of benefits, will change over time and actions have to measured, evaluated and potentially adapted. Table 8 (from Keith et al. 2021) provides an overview of what is required to extend current accounting systems to meet these needs and fulfil the requirements for ecosystem accounting. Monitoring and evaluation also help to understand the potential for up-scaling and transferability to other contexts. Monitoring requires the use of key indicators of the performance of the action, with consideration for the purpose and scale of the NBS action, which could be local with the aim of providing specific local benefits with carbon mitigation as a co-benefit, or as part of national policy to provide carbon mitigation at larger scale, with a range of co-benefits. Underpinning carbon accounting is the data on stocks and fluxes, and for the co-benefits and trade-offs, which are discussed later.

Indicators for carbon mitigation are focused on direct impacts on GHG emissions through storage in the biosphere, either in vegetation or in soils, and indirect impacts that lead to reduced GHG emissions through avoided land use change and improved management. Storage indicators are generally based on intensity values (e.g., tons ha-1 y-1) that encapsulate the storage for a unit area (e.g., ha) over a unit length of time (e.g., year). Avoided emissions are generally measured in tCO2e y-1. Other indicators may be needed to capture co-benefits around climate resilience (e.g., local cooling), air quality, water management, hazards, biodiversity, health and wellbeing and so on (EC, 2021).

Large challenges remain in measuring the effectiveness of actions, because of the need to have initial conditions to compare with (baseline data), the potential need to have a comparison area (control data), and the general requirement to collect data periodically over time. Accounting is further hampered by a lack of data or data collection capacity, especially in LMICs. Other issues with data quality, harmonization across jurisdictions, etc., can hamper monitoring and evaluation.

**Table 8.** Management strategies for forest carbon (from Ontl et al., 2020)

| Essential components | Description |
|---|---|
| All land areas and the ecosystems that occur | • Land and associated ecosystems are classified by extant ecosystem types irrespective of degree of human management<br>• All ecosystems provide benefits of carbon storage and require some management<br>• Carbon stock change represents the total exchange with the atmosphere<br>• Spatially referenced to allow attribution of stocks and flows |
| All carbon pools | • Above- and below-ground biomass, dead standing biomass, coarse woody debris, litter, soil carbon, and aqueous carbon (dissolved and particulate organic carbon) |
| Quality or condition of carbon stocks | • Stability, magnitude, longevity, time required for restoration, and resilience related to risk of loss<br>• Differentiation by classification of ecosystem types and characteristics of their condition as reservoirs of carbon<br>• Capacity to produce the ecosystem service of<br>• climate regulation<br>• Decline in condition is reflected as a reduction in the asset (stock) quality |
| Definition of 'forest' | • Refers to the actual vegetation cover at the time of accounting<br>• Includes components of forest structure, carbon stocks and biodiversity |
| Biosphere and atmosphere | • Distinguished as separate spatial units in a<br>• three- dimensional delineation of the accounting system<br>• All stocks and flows between the biosphere, atmosphere and economy are counted |
| Reference level | • Natural condition that represents ecosystem integrity, and underpins the carbon carrying capacity, is used to assess changes in carbon stocks<br>• Initial loss of carbon from a natural ecosystem and historical changes are counted<br>• Scenarios using any other baselines or counterfactuals must be explicit |
| Recording gross flows | • All sources of emissions and removals are transparent<br>• Gross flows show the carbon restoration potential from proforestation. |
| Permanence of carbon stocks | • Permanence used as a criterion in accounting<br>• All stock changes reported against a single reference level of the natural condition |
| Natural and anthropogenic disturbances | • All carbon stock changes attributed as additions and reductions in asset accounts |
| Ecosystem service of climate regulation | • The contribution of the magnitude and longevity of carbon stocks in the biosphere to reducing the concentration of CO2 in the atmosphere<br>• The benefit of the carbon stock in the biosphere depends on the ecosystem condition or quality of the reservoir |

**b) Methods and data sources for carbon accounting, including uncertainties: secondary data, field data, remotely sensed data.**

Current approaches for carbon accounting for land-based NBS actions are generally based on the use of a combination of field data, secondary data, remotely sensed data and process and data-based models that are integrated into carbon accounting models (Steininger et al., 2016). Comprehensive accounting for all components of the carbon budget is impossible through measurements alone, either on the ground and/or from remote sensing. However, process or data-based models can be used to extrapolate between observations and these models can, in theory, provide consistent estimates in space and time when constrained by available observations. The use of multiple independent datasets to estimate fluxes and stores is preferable as this provides uncertainty estimates.

Carbon stocks and their changes can be estimated from inventories of different components taken over time (Pan et al., 2011), for example, for live and dead biomass, soil carbon, and transport of biomass derived products such as from harvested crops and wood through supply chains and trade. Individual countries have inventories for some of these components, mostly focused on forest biomass inventories (Andersson et al., 2009) but also soil carbon inventories, and crop and wood products. Some of these are available as snapshots over time, but sometimes only once (Ciais et al., 2022).

At large scales (i.e., national), the focus is generally on bottom-up book-keeping methods, which can be complemented by top down “atmospheric inversion” methods.

Book-keeping methods rely on inventories of carbon stocks, and process and data-based models. There is no common approach to book-keeping and individual methods do not cover all stocks and fluxes. Current research is focused on reconciling the difference between top down and bottom-up approaches at the global or regional scale (Kondo et al., 2019). Local scale inventories may focus on specific components and will generally be based on locally collected field data but supplemented by inventories and secondary data at other scales.

Fossil fuel CO2 emissions are generally developed from national inventories, and these are collated and reported by the Global Carbon Project (GCP). Gridded versions of these are available, such as the Jones et al. (2021) dataset which differentiates fuel sources (oil, coal, gas) and cement production and available at 0.1-deg resolution for 1959-2020. Other anthropogenic and biogenic carbon compounds such as CH4, BVOCs, and CO are generally not taken into account in such inventories but can be significant, and this will tend to bias top-down estimates that are matched against atmospheric measurements.

Emissions from land use activities (land use change and management) can be estimated from remote sensing to identify areas of change and potentially followed up by targeted field work for more detailed inventorying. Often this is integrated in carbon accounting models that estimate changes in the net carbon stock as a result of the balance of different anthropogenic activities and natural processes that act to accumulate or emit carbon (e.g., Kurz et al., 2009; Waterman and Richards, 2008). Accounting models include process-based and empirical sub-models, and parameter values to estimate various stock changes such as allometric relationships, decomposition rates, combustion efficiencies, drainage losses, etc. Generally, a variety of secondary data is required to estimate components of changes in the carbon stock, such as management and harvesting operations, surveys of planting and deforestation, fire incidence data, maps of soil organic content, peat extraction and sales, and process-based model estimates of harder to measure changes such as for soil emissions.

One of the largest challenges is the uncertainty in how carbon is accounted for along supply chains, which are becoming longer, and across countries, which is becoming more prevalent (Bastianoni et al., 2004; Steininger et al., 2016). Emissions may originate in one area and sector (e.g., forestry products), but there are options on how to attribute those emissions along the supply chain (e.g., where a product is made or where it is consumed) and therefore across one or more countries. Essentially this spatial and temporal redistribution of emissions relates to responsibility for emissions. Steininger et al. (2016) propose four main principles that can be applied: 1) extraction-based accounting, reflecting the carbon content of fossil fuels extracted; 2) income-based accounting, allocating emissions along the production chain based on the supply of factors of production; 3) production-based accounting, assigning emissions to the country releasing the pollutant; 4) and consumption-based accounting, attributing emissions to final users of goods and services produced. There is debate on how to frame these in terms of compensation or distributive justice.

**Field measurements**

Field measurement is focused on characterizing vegetative (above- and below-ground biomass) and soil carbon content and can take a variety of approaches. Approaches for forest carbon estimates include “destructive sampling” by harvesting of vegetation to estimate dry weight and then using standard conversion factors to estimate the amount of carbon. Although accurate, this is an infeasible approach for large areas. More acceptable is the use of site and species specific allometric equations which relate direct measurements of height, diameter at breast height (DBH), crown closure and stem density to carbon content as based on statistical relationships derived from many paired measurements of heigh and diameter with destructive sampling (e.g., Chave et al., 2014; Paul et al., 2015). These approaches are generally applicable even in diverse systems if the allometric equations are specific to the broad ecosystem in question, although there is uncertainty in published equations especially for tropical forests (Ramankutty et al., 2007). Many studies have focused on estimates of aboveground biomass, and less on belowground biomass (e.g., roots) as techniques are highly specific and time consuming (Danjon and Reubens, 2008).

Forest inventories are a core component of field-based approaches for national accounts and have a long history in countries like the U.S., where the Department of Agriculture Forest Inventory and Analysis (FIA) has been repeated annually for several decades. Although such datasets reflect comprehensive data collection on individual trees at plot level, and with generally annual repeat, they suffer from a number of drawbacks when used for carbon accounting (Andersson et al. 2009; Sleeter et al., 2022). Firstly, some form of extrapolation is needed to map attributes beyond the sampled plots, which is generally quite uncertain (Marvin and Asner, 2016). This can be exacerbated by poor sampling of plots and unrepresentative plot selection, inaccurate measurements and inappropriate allometric models (Petrokofsky et al., 2012). Secondly, understanding of past changes in carbon stocks is difficult because the drivers and carbon fluxes are not known. Thirdly, future changes in carbon stock are difficult to predict if driven by land use change that has not historically been observed. Fourthly, traditional inventory approaches may not account for dead wood and litter which can be significant in terms of carbon storage, and there is less focus on below-ground biomass as noted above. Finally, implementing in all countries is a challenge for field-based approaches, where access is difficult in some regions and the areal extent is prohibitive. Reliance on remote sensing approaches is therefore key to implementing a broad (national) scale and sustained programme.

Measurement of soil carbon is possible using in-situ methods that involve direct sampling (Smith et al., 2020b). However, this is often done for research purposes to understand soil fertility and impacts of agricultural management, for example, and not necessarily to understand soil carbon changes under land use. Measurements are often limited to the top 30cm or so of the soil profile and may not be carried out over sufficient temporal sampling rates and for long enough to capture decomposition rates and final storage at equilibrium (Petrokofsky et al., 2012) with an understanding of land management that can affect carbon storage. Sampling also needs to be sufficient to capture the spatial variability. This can be complemented by long-term flux measurements using soil chambers or eddy covariance systems that can be used to estimate changes in storage above and below ground (Baldocchi, 2003) and compare with direct measurements of changes in stocks. Spectral sensing methods are becoming more prevalent to infer carbon storage from reflectivity in the infrared range (Nayak et al., 2019). Upscaling of point measurements of plot measurement to landscape or large scale is difficult and costly (Alexander et al., 2015) but can be done using large-sale predictors such as climate, land cover and soil type information including from remote sensing or via combining global measurement networks (e.g., Fluxnet; Balodocchi et al., 2018) with land surface modelling, albeit with additional uncertainties depending on the heterogeneity of the landscape.

**Remote sensing**

Satellite remote sensing has become critical for a range of monitoring activities, including carbon accounting. It is indispensable in carbon accounting as it can provide a consistent and large-scale perspective on land surface stocks and changes at increasingly higher resolution. This contrasts with field inventories that are expensive and infeasible at large scales and whose data usefulness is dependent on the existence of historic baseline data collected in a consistent way. It also contrasts with remote sensing from airborne platforms (piloted aircraft), which although can be targeted on a particular area and objective, are expensive to operate relative to the scale of data collected. Airborne platforms therefore tend to focus on experimental sensing, and testing of new sensors. Approaches using drones (autonomous aircraft) are much cheaper and becoming more popular for local accounting (e.g., Torresan et al., 2017).

Various technologies can be applied to carbon measurement on satellite and other remote platforms (Goetz et al., 2009). Synthetic aperture radar (SAR) can be used to detect above-ground biomass by using the backscatter from its active microwave radar, which is sensitive to the structure of the canopy. Microwave wavelengths have the advantage that they can penetrate through clouds and so can provide more frequent repeat sampling. Longer wavelengths in the L and P bands are more sensitive to larger structural elements such as trunks and stems, whilst shorter wavelengths in the X and C bands are more sensitive to smaller elements such as leaves and twigs. These can be complemented by estimates of tree height determined using interferometric SAR (inSAR). The upcoming ESA Biomass mission is an example of a dedicated mission focused on measuring global forests (Quegan et al., 2019). Similar to SAR is Light Detection and Ranging (LIDAR), which operates in the optical range and is used to measure the 3-D profile of the canopy and is often focused on retrieving canopy height. LIDAR suffers from not being able to see through clouds, however. LIDAR instruments are generally flown on aircraft for regional mapping, although retrievals are leveraged from non-vegetation missions such as on the ICESAT satellite and more recently the dedicated Global Ecosystem Dynamics Investigation (GEDI) spaceborne mission on the International Space Station, which has been flying since 2018 (Dubayah et al., 2020). Finally passive optical sensors that have been flying on the Landsat satellite series and other platforms for multiple decades have been retrieving above-ground biomass based on relationships between ground observations and optical reflectance measurements (Spawn et al., 2020). Again, they suffer from cloud interference and low repeat times.

In the context of monitoring stocks and flows of carbon, satellite approaches generally focus on tracking land cover change at high resolution and changes in biomass (Ciais et al., 2022), as well as atmospheric CO2 concentrations. Land cover changes can be estimated using land cover maps that are updated on somewhat regular basis, such as those from the European Space Agency (ESA). These include the ESA-CCI 300-m land cover product, available annually from 1992 and 2018 (ESA, 2017), and the Landsat 30 m spatial-resolution land cover change product for forest, short vegetation, and bare soil from 2000 to 2018 (Song et al., 2018). Generally, land cover change tracking has been used for monitoring deforestation with a focus on the state of tropical forests. For example, the EC JRC has been monitoring tropical forests from remote sensing under the TREES project since the early 1990s (Achard et al., 2014). The project has been evaluating forest cover at periodic intervals (1990, 2000, 2010) using systematic sampling based on the 30m Landsat data.

Biomass can be measured using a variety of approaches based on optical, SAR, and LiDAR sensors or a combination of these, and often combined with empirical regressions or machine learning to extrapolate field measurements over large scales (Urbazaev et al., 2018). Current datasets include the ESA GlobBiomass dataset of aboveground biomass data at 100 m spatial resolution (Santoro, 2018). The NASA Carbon Monitoring System program (Hurtt and Kang, 2014) is developing global capability to monitor biomass using combinations of optical MODIS products and lidar approaches using ICESAT-1/2, as well as the GEDI lidar mission. The RECCAP project and its successor (RECCAP2; Ciais et al 2022) have focused on global to regional scale monitoring for carbon accounting and identification of emissions associated with land use change, with heavy reliance on satellite data. New opportunities to monitor at sub-metre resolution are arising from the use of nano-satellites (e.g., Planet Team, 2017) that are most cost effective than traditional sources of such data, and can allow mapping of degraded forests, trees outside of forests, hedgerows and other isolated vegetation (Skole et al., 2021).

Challenges include how to translate remotely sensed images and derived products into inventories of type, extent and carbon content. This is partly dependent on the sensor characteristics, especially as relates to spatial resolution which can require classification of mixed pixels as one category. Sensing of land degradation is difficult, such as selective logging of particular species (Gao et al. 2020). Field data is crucial for validating remotely sensed products, and also for developing data-driven predictive models, e.g., using machine and deep learning methods to predict variables of interest (e.g., above-ground biomass) that are measured on the ground but at sparse locations and times, based on large-scale predictor variables that can be measured by satellites (Odebiri et al., 2021). Remote sensing is also an input into terrestrial carbon models such as DGVMs, and soil biogeochemical models.

**Process based and data driven models**

Process-based models can be used to estimate how carbon stores and associated emissions have changed historically (Janes-Basset, 2021), as well as predicting how they may change in the future under different scenarios, such as for climate or land use change (Huntzinger et al., 2012; Bachelet et al., 2015). They can be applied to monitoring current conditions (Sleeter et al., 2022), and for attribution whereby model experiments are run to understand, for example, the impact of climate variability on the uptake of carbon in forests. Models fall into two main categories: terrestrial biogeochemical models (TBMs) and dynamic global vegetation models (DGVMs), including more specialized models developed for forest, agricultural, savannah, wetland or permafrost environments that differentiate between species groups and represent biome-specific processes. Biogeochemical models are used to represent the carbon cycle and its close coupling to the nitrogen and phosphorus cycles (Achat et al., 2016). They tend to be applied at local scales because of the often site-specific calibration and focus on a single land use, but have been applied at regional to global scales although large scale models tend not to be able to represent detailed agricultural practices (Wang et al., 2010). DGVMs further couple with slower processes associated with competition, growth and mortality of different plant functional types that lead to changes in ecosystem structure and composition (Prentice et al., 2000; Morales et al., 2005). These models can be used to understand the sensitivity of terrestrial ecosystems and their carbon storage to external drivers such as climate change and disturbances (e.g., pests and fire). Because of the complexity of the tight coupling between the carbon cycle and other nutrient cycles, and the significant and accelerating influence of management, land use and other external drivers, such models are the only way to assess changes at large scales.

Versions of DGVMs are incorporated into Earth System Models (ESMs) which simulate the carbon cycle and vegetation feedbacks with climate and other biogeochemical cycles, in addition to the climate as simulated by climate models (GCMs). However, there are large uncertainties in these models, how they represent observed carbon dynamics and the range of their future projections (Luo et al., 2015). Much of this uncertainty stems from uncertainties in model structure and parameter values that are not well constrained by observations.

Data-driven models are different from process-based models in that they represent relationships between components of the system based on empirical relationships derived from observations (or sometimes modelled data). They can take many forms and be applied in many ways. Carbon budget or stock-flow models track fluxes of carbon between a number of carbon pools (Heath et al., 2010; Kurz et al., 2009; Sleeter et al., 2022) but are generally constrained by inventory data in the form of volume curves that relate forest stand age to biomass. Other functional relationships derived from empirical data or process-based models are used to represent connections between biomass and other biomass pools such as roots and branches, between aboveground and belowground biomass, and decomposition and decay rates. Representations of fluxes due to disturbances (land conversion, fire, disease) and climate variability and change (e.g., warming temperatures) are sometimes included making them suitable for looking at future scenarios of carbon stock and flows (Sleeter et al., 2022).

Approaches to integrate measured data from ground observations and remote sensing into models has distinct advantages (Wu et al., 2019). With the ever-increasing wealth of geospatial data from satellite sensors, models, and observations that provide direct and indirect or proxy measurements of carbon storage and fluxes, a large opportunity exists for using AI methods (e.g., deep learning) to integrate disparate data types and provide prediction and uncertainty estimates for carbon related variables (Reichstein et al., 2019; Tao et al., 2020).

**c) Approaches for estimating co-benefits of land-based NBS for carbon mitigation**

Methods to estimate co-benefits are hampered by the complexities of measuring impacts across different sectors and societal challenges, and potentially different areas and time periods (Raymond et al., 2017). Assessment frameworks have to date mostly been focused on the main purpose of the action (i.e., carbon storage) or have taken a narrow view of co-benefits, for example focusing on a specific framework or assessment approach (Price et al. 2021). As such there is a lack of practical and targeted guidance for how to incorporate co-benefits into assessments (Ürge-Vorsatz et al., 2014; Raymond et al., 2017), as well as simultaneous actions in other sectors by potentially multiple actors (Maes and Jacobs, 2017).

The cost effectiveness of NBS is often unclear and sometimes over-stated, in part because of the non-accounting of the co-benefits and trade-offs, especially as NBS are multi-functional with effects playing out across multiple spatial and temporal scales, and co-benefits that can be non-monetary (Reid et al., 2019; Seddon et al., 2020). This is also due to a lack of consistent metrics to measure co-benefits (and benefits), especially against other options so that institutions can compare and chose the best option (Swann et al., 2021). NBS options can be less prioritized if the co-benefits are non-monetary (Reid et al., 2019). This makes it difficult to communicate clearly to decision- and policymakers, as well as investors, and therefore hampers the process of identifying appropriate NBS, and the development of policies that are cross-sectoral.

A typical cost-benefit analysis based on immediate paybacks is not useful for most NBS where benefits accumulate over time and often not uniformly accrued (Raymond et al., 2017). The effectiveness of NBS actions may also evolve and degrade with time as a result of climate and socio-economic change (Calliari et al. 2019). Work needs to be done to identify how to best to measure and assess multiple direct and co-benefits in such dynamic and complex systems across time and space scales, across sectors, and in monetary and non-monetary terms (Austin et al., 2021). This highlights the dependency of NBS effectiveness on the implementors and stakeholders involved and their viewpoints, which makes assessment inherently a context specific process with the need for context-specific metrics and evaluation approaches (Seddon et al., 2020).

New frameworks for assessment are emerging that attempt to capture the dynamic and multi-scale, multi-sector contexts of NBS (e.g., Raymond et al., 2017; Calliari et al., 2019), which can be applied ex ante to allow the comparison of competing or alternative options, including grey infrastructure. In the context of urban NBS, Raymond et al. (2017) identify four dimensions to be considered in a comprehensive assessment framework: 1) co-benefits for human health and well-being; 2) integrated environmental performance (e.g., the provision of ecosystem services); 3) trade-offs and synergies to biodiversity, health or economy; and 4) potential for citizen's involvement in governance and monitoring. They use these to identify cross-cutting challenges and solutions across spatial and temporal scales of socio-ecological systems that comprise the climate/physical environment, ecosystems, biodiversity, and socio-economic and socio-cultural systems.

Measuring and assessing co-benefits is a critical and evolving part of the framework and the implementation of the NBS, with recommendation for long-term monitoring that can feedback into adaptive management and evolution of future plans for the NBS action. Multiple existing and new indicators of NBS effectiveness are needed to traverse these systems and scales, that could include integrated environmental performance, health and well-being benefits, civil participation and transferability, and economic performance and financial return (Raymond et al., 2017). Indicators can be measured directly, indirectly, or modelled using a range of quantitative, qualitative and mixed methods, as described previously. Raymond et al. (2017) provide examples of methods suited to different direct and co-benefits.

Calliari et al. (2019) evaluate this and other similar frameworks and take them further to propose a framework that considers the constituent elements of NBS (multifunctionality; simultaneous delivery of economic, environmental and social benefits; multi-stakeholder engagement), and addresses the impacts of future climate change on the NBS ecosystems and ecosystem services that are generally missing in previous frameworks. Their framework is based on system analysis to provide a set of logical steps for identifying the best solutions to a problem (choosing between options), and then visioning and backcasting, which co-develops visions of the future and pathways to reach these. Crucial to this, as noted above, is mapping of the direct and indirect effects of the NBS and quantifying these based on a set of identified indicators. Calculating co-benefits using SDG indicators is likely the most efficient and logical approach, linking clearly to the SDG monitoring process.

# Implementing Land-based nature-based solutions: Case studies of Land-based nature-based solutions projects around the world

**Peatland Restoration in Indonesia**

Given the large amount of carbon storage in peatlands and their degradation worldwide, there is large scope for restoration to sequester CO2. Indonesia is one of the countries with the highest potential (see section 5) (Brancalion et al., 2019), containing the largest amount of tropical peatland carbon storage of 57 GtC (55% of tropical peatland carbon) (Page et al., 2011), but with some of the largest threats from conversion to agricultural land (for oil palm and pulpwood plantations), illegal logging and mining (Warren et al., 2017). There is also the threat of peatland fires which are locally devastating but have broader impacts. Degraded peatland covers about 23% of Peninsular Malaysia, Sumatra, and Kalimantan (Hansson and Dagusch, 2018) and is a legacy of long-term agricultural conversion, such as the Central Kalimantan Peatland Development Project initiated in 1995, which later became known as the Mega Rice Project (MRP). More than half of the 21M ha of Indonesia's total peatland was drained by the end of the 2000s mainly for large agricultural plantations (Mietinnen et al., 2012).

A number of small scale, pilot restoration projects have been ongoing since the early 2000s (Dohong et al., 2018), but with little impact on the overall loss of peatland and the associated carbon storage. Large-scale action on peatland restoration was prompted by the extensive fires in 2015/16 driven by the very dry El Niño conditions and land use change (Adrianto, et al., 2019), which not only degraded the existing peatlands but had a number of other important impacts such as reduced air quality affecting across the region beyond Indonesia. The fires burnt 2.6M ha of peatland, generating 1.5 billion Mt CO2 emissions in 2015 (Heymann et al., 2017) and affecting 69M people and with costs to the Indonesian government of $16B (World Bank, 2016; NatureNow, 2022). This is part of trend of higher frequency of fires in recent decades compared to the very long-term (Dommain et al., 2014).

The fires and their impacts drove the government to instigate a moratorium on peatland use and a restoration programme for the peatlands of the order of 2.5 Mha over five years across seven provinces, through the National Peatland Restoration Agency (BRG), to reduce the risk of fires and as part of its NDCs. The regions most impacted by the fires were Central Kalimantan and Riau. Rewetting of peatland areas in these regions has entailed the removal or damming of drainage canals and was complemented by forest revegetation and livelihood revitalization (BRG, 2016). Progress overall has been mixed with about 45% completed by 2020, likely hampered by the spatial complexities of land tenure and jurisdiction. An extension to the BRG's mandate was granted with a goal of 1.2 Mha from 2021-2025 and has reached 300,000 ha restored by 2021, which is about 25% of the target (Jong, 2022).

Restoration approaches, however, can be at odds with local community needs, as these regions have also been cleared for agriculture for local livelihoods and income, and the drainage canals are important transport corridors for exporting agriculture products including timber. So careful integration of stakeholder needs and concerns (e.g., land tenure, alternative livelihoods including from paludiculture) is required to make such restoration schemes acceptable and sustainable

(Jewitt et al., 2014; Schaafsma, et al., 2017). The total cost may be severely underestimated, and significantly higher than the investment by the Indonesian government and international donors, with estimates of US$4.6 billion required (Hansson and Dagusch, 2018), although there are large uncertainties in cost estimates (Glenk and Martin-Ortega, 2018). However, this compares to estimated cost savings of US$8.4 billion from the restoration (Kiely et al., 2021). Further work is needed to identify more cost-effective methods for restoration, more diversified funding sources that move away from international donors and towards more market-based initiatives (Puspitaloka et al., 2021), improved land use policy and governance (Dohong et al., 2018).

Farmer managed natural restoration in West AfricaForest restoration projects fall under a range of global and regional initiatives such as the UN Decade on Ecosystem Restoration (2021–2030), Bonn Challenge, and AFR100. The latter has ambitious targets for sub-Saharan African countries to restore 100 million ha by 2030. However, the cost and time to develop and implement these large-scale initiatives may be significant (Holl and Brancalion, 2020), whilst deforestation continues (FAO, 2020). Natural regeneration is potentially more appealing because of the lower costs (Catterall, 2020), but the time required and the context-specific success is a concern. Restoration on actively managed farmland has potential to address this challenge by integrating tree planting and management as part of agriculture livelihoods, across the millions of smallholder farmers (Bayala et al., 2019). This is known as Farmer Managed Natural Restoration (FNMR; Haglund et al., 2011) and is “the practice of actively managing and protecting non-planted trees and shrubs with the goal of increasing the value or quantity of woody vegetation on farmlands” (van Haren et al., 2019). This is a form of agroforestry, which has a wide range of potential benefits, including land restoration, improvements in agricultural productivity, and impacts on livelihood, include and wellbeing, and improvements in biodiversity and other environmental quality dimensions. It is therefore primarily implemented for these benefits, rather than for carbon sequestration, and has success rates far higher than for large-scale tree planting with survival rates of just 20% (Chomba et al. 2020).

FNMR has been promoted across the Western Africa Sudano-Sahel since the droughts of the 1970s and 80s, mostly focused on regions of Niger, Chad, Mali, Burkina Faso, northern Ghana, Nigeria and Senegal (Tougiani et al., 2009; Casey et al., 2020) with 5–6 Mha restored, particularly in the Maradi and Zinder regions of Niger (Smale et al., 2018). A number of studies have documented increases in tree and shrub cover as well as more soil organic matter (Reij et al., 2009; Weston et al., 2015; Stith et al., 2016; Bayala, et al., 2019). Yet the evidence base overall is small and scattered, and the attribution of documented benefits to FNMR is uncertain, with overreliance on a few particular studies (Chomba et al., 2020). The potential to scale has nevertheless been promoted based on such limited evidence, often funded and enabled by external donors and NGOs, yet much more work is required on the social and governance barriers and opportunities of restoration (Kandal et al., 2021; Elias et al., 2021), including alignment with NBS principles of equity and social inclusion (see section 3c).

**a) Frameworks for carbon valuation (market and societal) and credits**

Carbon valuation is used in government policy and international agreements for valuing the impacts of interventions and actions on GHG emissions. As such they represent the monetary value that society puts on carbon as expressed in $ per tonne of CO2 equivalent (Stechemesser and Guenther, 2012). A carbon price puts a value on the social cost of the impacts of GHG emissions (e.g., damage to crops from droughts, health impact of heatwaves, or damage to buildings from flooding) and is used to tell the emitter how much they need to change practice, lower emissions or pay the price (Nordhaus et al., 2014). The price can also be calculated in terms of the mitigation cost, by either estimating the cost to achieve a (net) emissions reduction target or as the expected price in a carbon market (UKGOV, 2021).

Carbon pricing can help mobilize financing to drive innovations in low-carbon technologies and practices, and an overall transition to a decarbonized world. It can also help governments as one element of a portfolio of climate policies and where best to focus interventions; it can help corporations to identify risks in their business models and possible revenue opportunities; and help investors identify risks in their portfolios and possibly how to invest in low-carbon activities (Baranzini et al., 2017).

Well-constructed schemes for carbon pricing and support for mitigation are required to incentivise business and society to scale up mitigation actions, as well as demand-side reductions. However, there is limited research literature on how to value carbon in monetary, and especially in social terms, although there is best practice to be gleaned from existing frameworks (UKGOV, 2021). Some initial framing of the problem specifically for NBS has been done (e.g., Seddon et al., 2020) and how to allow for the externalities of NBS benefits that create problems of ownership. Different models of financing are being proposed through multilateral consortia across a diverse range of partners, where risks are shared and funding is provided through equity in the programme to be undertaken. Large uncertainties remain in how best to value carbon and the literature is evolving (e.g., Pindyck, 2019; Wang et al., 2019; Huang et al., 2016).

It is estimated that about 15% of global emissions are covered by some form of carbon price implemented through a taxation or trading scheme (World Bank, 2017). Costs have generally been too low to incentivise investment and actions at scale to have any chance of meeting the Paris Agreement targets. However, at the time of writing (May 2022), a flurry of recent EU legislation to lower emissions in Europe to reach 2030 targets, as well as increases in gas prices leading to more use of coal and higher demand for carbon permits, has driven the price up to about €100 in the EU emissions trading system (ETS) (Reuters, 2022). Prices of this order may now be high enough to enable investment to upscale NBS, and also to invest in green technologies (e.g., for cement and steel production, or for carbon capture and storage). As public and policy attitudes continue to favour climate action, the hope is that prices will increase further to attain levels of investment required without government support.

**b) Standards and best practice, including transparency, accuracy/precision**

Standards for NBS are required to ensure that they adhere to widely agreed principles (e.g., IUCN, 2020; EC, 2021) and are designed and implemented in ways that provide robustness and quality. This will ensure that projects, whatever the scale, are not subject to criticism and uncertainty, and quality is safeguarded (NetworkNature, 2022) so that investments can be made and credits traded. Certification standards for NBS projects are emerging (and need to meet rapidly increasing demand for NBS projects) but have a basis in the longer-term development and implementation of standards focused on REDD+ forest projects.

**i. Project standards**

Independent standards have been developed and applied over the past two decades aimed generally at specific projects at small scale. They are commonly developed by private and non-governmental institutions according to different methods and procedures and are aimed at providing quality assurance of projects. Ultimately, they provide trust in carbon markets so that certified emission reduction and removal can be translated into tradable carbon credits (UNDP, 2021). A range of carbon standards include the Verified Carbon Standard (VCS; also referred to as the Voluntary Carbon Standard), the Climate Action Reserve, and the Gold Standard. The VCS is the most widely used voluntary programme globally, with of the order of 1,800 certified projects associated with reductions or removals of almost 922 MtC and other GHGs (VCS, 2022). The VCS was developed through a collaboration between the Climate Group, the International Emissions Trading Association, the World Economic Forum and the World Business Council for Sustainable Development. Such standards offer standardized methods on which to base methods for certification, with the goal of enabling and streamlining the process for specific projects.

Analysis of a selected range of REDD+ standards by Schmidt and Gerber (2016) indicate that there is large diversity in how they meet a range of carbon and co-benefit criteria (climate integrity; biodiversity conservation; human and community rights, stakeholder participation and sustainable community development; long-term project viability and compatibility with UNFCCC and jurisdictional approaches) with no standard providing a satisfactory overall performance. Combinations of purely carbon focused standards and co-benefit focused standards performed the best overall but potentially may not be sufficient to meet institutional criteria.

Co-benefits and trade-offs are increasingly being incorporated into these standards, either to ensure that co-benefits are materialized but also to provide safeguards to ensure that negative impacts are minimized. For example, the Gold Standard requires projects to have impacts on two additional SDGs, beyond SDG13 on climate action. This is done through specific methodologies focused on, for example, gender equity, water access, or health benefits. Alternatively, this can be achieved by attaching separate standards to the carbon standard. For example, the Sustainable Development Verified Impact Standard (SD VISta, 2022) has been implemented since 2019 to verify sustainable development benefits associated with carbon mitigation projects. Trade-offs are similarly being tackled in standards through safeguarding (e.g., requirements of the proportion of native species in afforestation and reforestation; or protecting / enhancing biodiversity), ensuring positive impacts on communities, and crucially by requiring stakeholder inclusion in project design (e.g., Plan Vivo standard).

**ii. Jurisdictional approaches**

Jurisdictional approaches can play a large role in promoting and supporting NBS activities, especially around deforestation, and may play a future role for other forms of NBS where sustainable practices can be assured. These approaches can address a number of challenges found in VCS. Firstly, they alleviate the burden on smaller entities who want to reduce their reliance on carbon emitting activities such as use of products that are associated with deforestation (e.g., soy, palm oil or timber), but have trouble attaining certification under VCS approaches. The onus is also put on the company to invest to meet the certification standards which can be expensive. The certification standards may be uncertain in the eyes of the consumer especially when products are derived from multiple sources with variable certification, and so traceability is an issue (Lambin et al., 2018). There may also be challenges in confidence in the monitoring and auditing processes (see section 6). The demand for sustainably certified goods is still small but expanding, and mostly focused on western companies, and this can hamper progress based on market forces. There has also been a recognition that governments should play a larger role in certification because of their vested interest in regulating and maintain natural resources through policy, regulation and engagement, which can also help reduce costs.

Jurisdictional approaches bring together government and larger companies, and multiple key stakeholders, to implement integrated landscape management with financial and technical support to a range of smaller producers, including through mapping and monitoring (Essen and Lambin, 2021). They have the potential to provide a more joined-up approach across sectors and therefore a larger-scale ambition. Although they are still in development in many areas, progress has been made in recent years in regional initiatives such as pilots and developing schemes in the Indonesian and Malaysian palm oil and Brazilian soy sectors. They can be defined as “an integrated landscape approach which aims to reconcile competing social, economic and environmental objectives through participation by a full range of stakeholders across sectors, implemented within government administrative boundaries, and with a form of government involvement” (EII, 2017). They aim for sustainable practices across the full jurisdiction and the full range of actors, chains and ecosystems. They can bring together the various approaches from farm and production level certification, domestic public policy approaches and corporate commitments to align on tackling more ambitious larger-scale goals (Mallet et al., 2016).

**c) Financing options and policies for carbon mitigation projects and credit schemes from international, national and public/private perspectives.**

A range of possible instruments are available for financing NBS at project and large scale. These include carbon pricing (carbon taxes or emissions trading schemes (ETS) and carbon markets), environmental or green bonds, and payment for ecosystem services (PES). Carbon pricing has been in use since the 1990s to incentivise reductions in GHG emission, and usually through cap and trade ETS. This allows participants to buy and sell allowances set by government or other jurisdiction, with the cap gradually reduced over time to meet national or regional emission reduction targets. Entities covered by the scheme (e.g., an industrial plant) can try to reduce emissions and sell their allowances or if this is not possible can buy allowances from other entities to cover their emissions. Other schemes include emissions reduction funds (ERF) by which governments buy credits created by emissions reduction schemes, and carbon taxing whereby a price signal is given around fossil fuel usage which will drive economy wide reductions in emissions. There are also hybrid schemes that combine elements of these.

Over the past few years, several nations and sub-national entities have set up ETS and carbon tax schemes, and there is a move towards joining these up internationally to address more ambitious goals (ADB, 2016; Beurmann et al., 2017). Schemes currently running or being implemented cover more than 50 jurisdictions at national and sub-national level with > $40B in revenues (WB, 2019). The EU ETS is currently the largest such scheme globally. It covers all 27 EU countries (the UK now runs its own separate scheme, with some remaining connections to the EU ETS) plus Norway, Iceland and Liechtenstein, and limits emissions from power stations and industrial plants, and airlines operating between member countries. It currently covers around 45% of total GHG emissions and is operating a reduction in the cap on total emissions by 2.2% per year to meet EU mitigation targets. The recently launched (mid-2021) ETS scheme in China will eventually be the largest in the world covering 4B tCOe2, which is about 40% of national emissions, although it does not have a firm cap on emissions. Emissions trading worldwide is estimated to cover about 15% of global emissions.

On the other hand, Voluntary Carbon Markets (VCM) sit outside of jurisdictional mandatory reductions under compliance or regulated markets (UNDP, 2021). Businesses purchase credits on a voluntary basis to meet their own commitments, including meeting goals of carbon neutrality. Credits can be used to compensate for or offset emissions that occur elsewhere, outside of national or regional cap and trade systems.

This is important as it can be used to direct climate financing to the developing world where the majority of carbon credits can be sourced (Streck, 2021) to provide multiple co-benefits around sustainable development. VCMs tend to be less burdensome and cheaper than regulated markets and so can be targeted at smaller projects that can reach local communities, and can direct financing to projects that would not develop otherwise, including supporting the development of new technologies. There are serious concerns, however, about whether such flows of finance actually benefit local communities and that this constitutes exporting the mitigation problem from developed to developing countries (Howard et al., 2015).

VCM credits have more than doubled since 2017 with 104 MtCO2e traded in 2019 for a cumulative market value of US$320 million (Kreibich and Hermwille, 2021). Estimates for the demand for carbon credits on the voluntary market are estimated to increase by a factor of 15 or more by 2030 and worth $50B, and up to 100 times by 2050 (McKinsey, 2021). However, the current VCM is very diverse and fragmented, with evaluations carried out on a voluntary basis and in-house using different standards and systems which creates uncertainty, and provides potential for mis-counting and green washing. To meet expected demand and scale up, the VCM needs to be larger, but also needs to be more transparent and more robust to ensure reductions are environmentally sound (Kreibich and Hermwille, 2021). Improvements in how actions are labelled can help the financial sector identify risks and opportunities and provides a more sound basis for strategic investments. Verification approaches need to be strengthened and made more streamlined. Higher quality credits using well-defined standards and common verification methods and accounting for co-benefits would help. Reducing the variability amongst credits would be a useful first step.

Suppliers of carbon mitigation projects would benefit from clearer price signals that stem from more quantitative and stable data that comes with scale and from more standardized labelling of attributes with common features. Inconsistent and variable labelling can provide for highly variable pricing. The lack of consistent pricing data means that buyers and sellers do not know whether they are paying a fair price, and for suppliers to manage risks of initiating projects when they may not know what price a buyer is willing to pay. The supply of projects also needs to ramp up quickly, whilst addressing the challenge of credits not being sold until the projects have verified negative net emissions. Temporary or forward credits and buffer reserves may help. Bundling of similar credits based on common standards would enable trading of larger volumes. Matching buyers with sellers can be time-consuming and so streamlining, standardizing and stabilizing the process overall will help.

Scaling up of the voluntary market to an overarching, transparent and robust scheme is a possible way to achieve this (McKinsey et al., 2021). Key to implementation of sustainable financing at scale, is to ensure that there are clear, consistent and transparent definitions of what is sustainable, its attributes and how it is verified (Swann et al., 2021). As discussed in section 5, it is vitally important that this is clarified for land-based NBS, for which such definitions are only just being consolidated and arguably more work is required to be of use in finance mobilization. Such an approach in effect requires a sustainable classification system or taxonomy including commonly agreed principles and metrics, or in the case of NBS, a taxonomy of NBS measures. Such a taxonomy provides standard definitions of measures that provide carbon mitigation and may include definitions of co-benefits and dis-benefits. Such clarity in definitions enables proper measurement of the implementation and outcomes of such actions and therefore tracking of progress and reporting to investors and stakeholders. Currently, the financial sector does not differentiate at this level of detail and therefore it is difficult to track the flows of finance in the sustainability realm.

The EU has led the way with the initial development of the “EU green finance taxonomy” (EU, 2019), which builds on the SDGs. This taxonomy is aimed at determining whether an economic activity is environmentally sustainable, to provide: an EU Green Bond Standard; benchmarks for low-carbon investment strategies; and guidance to improve corporate disclosure of climate-related information. All of which are essential to allowing the financial sector to confidently reorient investments to meet net-zero emissions targets and other sustainability goals. China has developed its own taxonomy for Green Bonds guidelines including attempts to align with European guidelines. Such a taxonomy needs to be applied to NBS.

Green bonds (sometimes referred to as climate bonds) are a more recent development since the mid-2000s, as originally developed by multi-lateral development banks, and more recently via the Climate Bonds Initiative and Green Bond Principles, and the FTSE Green Revenues Classification System. Green bonds are a useful way of providing funding for NBS type mitigation projects where other sources of funding (e.g., loans) are not possible. The issuance of green bonds has increased rapidly since about 2012, with bonds now issued by governments, institutions and business. Suggestions to provide hybrid schemes that combine carbon credits and bonds have been proposed (e.g., World Bank, 2019).

Overall, current carbon pricing schemes and bonds fall far short of the estimated billions of dollars per year required to provide the necessary contribution to the overall mitigation problem. Up to now, the integration of NBS in carbon markets has been mainly focused on the forest sector in terms of measures to conserve, restore and afforest. Yet this has largely been aimed at diverse and smaller projects, whilst there are large concerns about the lack of progress on REDD+, which represents the only global strategy for forest preservation, with deforestation likely increasing in recent years (NYT, 2020). This has been in part due to the low cost of forest carbon credits (as low as $5 per tCO2) compared with investments needed in new technologies to reduce emissions, in addition to generic concerns over monitoring and verification. Results based funding via the Green Climate Fund (GCF) $500M pilot programme begun in 2017 is showing promise as a way of funding at larger scale (GCF, 2017), as is the World Bank's $900M Carbon Fund. But concerns linger on the permanency of these conservation schemes. Other land use sectors have more recently begun to be incorporated such as agricultural and urban, for which there is large scope to scale up and use carbon markets to incentivise. Much of the focus has also been on certain parts of the world, particularly South America, with less well-coordinated approaches in southeast and east Asia (Lechner et al., 2020).

**d) Leveraging financing and co-benefits to enhance uptake**

To move NBS forward and to upscale to the scope necessary for meaningful impact on climate mitigation, additional financing is required (IUCN, 2021). The need is very large, and potentially of the order of trillions of dollars by end of this century (Royal Society, 2018). Current funding for NBS comes from a variety of sources including public and private, national and international funds (e.g., Green Climate Fund, the Adaptation Fund, Global Environmental Facility). For forestry, financing is delivered mainly through set-aside funds such as PES and carbon credits, and via the private sector for individual projects using voluntary markets. However, currently less than 5% of climate finance goes towards land-based mitigation (Buchner et al., 2015; Griscom et al., 2017; Seddon et al., 2020), despite a significant body of evidence of its potential. This is likely primarily related to uncertainties about the potential and its costs but may also relate to the social and political factors that can prevent uptake.

The required additional financing will necessitate diversifying funding sources and developing strong partnerships between the public and private sector. The necessary investment will flow if NBS activities are presented with returns that are adjusted to the risks and requires policy interventions to reduce risks and incentivise returns. As actions are required urgently this may necessitate government intervention in markets to encourage investments where start-up costs are high or technologies are less well developed at scale. Often NBS implementation requires investment in actions that will not materialize returns for many years or even decades that can cause uncertainty. There are also uncertainties about whether increased carbon stock are maintained in the long-term, if not indefinitely. NBS are increasingly promoted to address a range of social and environmental problems at scale, which presents uncertainties to investors. The multiple co-benefits can deter investment because of the perceived complexity especially around working with multiple stakeholders, when applied at scale. This inevitably will lead to uncertainties in how to manage such projects, and the large-scale ambition may present perceived risks.

Nevertheless, leveraging additional financing is likely to be enabled by drawing from the inherent co-benefits that NBS presents, such as links to climate adaptation, SDGs, biodiversity goals, etc. This will also require close cooperation between the public and private sectors, and innovations in how financing is mobilized and used with a focus on “sustainable finance” that is targeted at sustainability goals including climate mitigation (Gehrig-Fasel et al., 2021). For example, private sector investment in offsets delivered by NBS will be key to reaching net-zero (EU, 2019). But this will require strong governance to ensure that the co-benefits of NBS are materialized and social and environmental safeguards are ensured. There is vast potential to increase the NBS in NDCs in terms of the diversity of NBS that they incorporate, which to date is mostly focused on forestry, and there are large gaps in the specificities of the targets in terms of the scope, sectors, approaches and measurability. This can attract NBS funding through climate finance. Further, it needs to sit within a broader set of national and regional policies and international initiatives to transition to a decarbonized world.

Carbon markets offer great potential for NBS for mitigation, primarily because they offer opportunities for private investment to bridge the existing gap in climate finance. NBS have been financed for a while in such schemes based on voluntary standards and well-evidenced methodologies for monitoring and verification, but often at the project level. To upscale from credits provided at the project level to country level means overcoming a range of barriers. There are opportunities to leverage the increasingly fuzzy line between compliance at jurisdictional level and credit markets and go further to align them. As carbon markets evolve and merge this creates more demand for NBS. Carbon markets have now evolved from national schemes to new blended markets. There are also new schemes being developed that are aimed at specific sectors (e.g., the CORSIA air transport sector). Corporate schemes and initiatives could provide opportunities, which would require credits from NBS applications such as removals of carbon to attain net-zero. Regulatory systems need to be updated to allow NBS implementation and market-based climate finance (Lederer, 2012). Forward crediting is needed to bridge the gaps between up-front investments and lagged returns. Trade-offs between different NBS activities need to be managed as do the potentially complex interactions between a wide range of interested stakeholders, which likely requires system wide frameworks to understand and communicate these across different NBS activities. Common policies and regulatory requirements and incentives across schemes would be required. The increasing requirement for the use of NBS for carbon removals can help. Risks of double counting across different markets, supply chains, countries and with the NDCs need to be reduced as NBS implementations are scaled up.

# Risks around Land-based nature-based solutions projects

Despite their large mitigation potential, as well as protection from the impacts of climate change, there are risks associated with NBS implementation. A central issue that can lead to multiple risks is the lack of clear definition and guidelines on what is NBS and how it can be designed, implemented, monitored, verified and evaluated, despite recent progress (IUCN, 2021). In some respects, a single clear definition and guidelines may be useful, yet many have been developed and applied by a range of agencies and institutions. This provides choice on how to implement which is welcomed, but also risks, especially if there are inequitable benefits or disadvantages for sets of stakeholders because of particular choice of definition and guidance. A major distinction is between the IUCN framework which emphasizes biodiversity and well-being, whilst the EU sets priorities around economy and social benefits with an emphasis on urban NBS. Gaps or lack of clarity in any framework will lead to a number of risks.

**a) Risks associated with uptake, financing and sustainability**

A fundamental risk is lack of uptake of NBS. Despite the growing interest and theoretical impact, and evidence from a wide range of projects and demonstrations, there is potential that uptake at scale may come up against a variety of barriers. For example, in the context of NBS in urban areas, Kabisch et al. (2016) highlight "fear of the unknowns, the disconnect between short-term actions and long-term goals, the discontinuity between short-term actions and long-term plans, sectoral silos, and the paradigm of growth." Risks of financial under-investment are apparent and have been seen in diverse expectations and progress on deforestation under REDD+ (Turnhout et al., 2017) and in associated efforts on climate mitigation to date, such as the low implementation of carbon capture and sequestration at scale (Martin-Roberts, 2021). There are risks around the cost-effectiveness of NBS. As NBS are generally multi-functional with a range of co-benefits (and disbenefits), it is difficult to estimate the worth of these in monetary terms (see section 7). Further, these co-benefits may be inherently difficult to cost, if they relate to intangible impacts such as mental health and wellbeing. This can deter investment. Comparisons of cost-effectiveness of NBS showed that investors were less inclined when there were high up-front costs, where benefits accrued over time or where many co-benefits were non-monetary (Reid et al. 2019). An increasing focus on NBS may divert funding from other and often related or similar activities labelled, for example, as ecosystem approaches. Projects are already happening which are attracting climate co-funding (e.g., Arbaro Fund that invests in sustainable plantation forestry projects in emerging markets of Latin America and Sub-Saharan Africa; GCF, 2022). There will also be opportunity costs as the implementation of one NBS approach (e.g., biochar) decreases the opportunity to use the biomass for another (e.g., for biofuels). The necessary investments may be difficult to materialize because the benefits may emerge over the long term and be spread across a range of beneficiaries, making it very difficult to coordinate funding across a range of investors. As mentioned above, consortia of diverse funders are likely necessary. There are also risks around green-washing, as corporations, institutions and governments leverage the increasing interest in NBS to recast dubious practices that are causing more harm than good, especially if it allows business-as-usual fossil fuel exploitation packaged as natural solutions (Gałecka-Drozda et al., 2021).

## Risks around Land-based nature-based solutions projects

There are also risks related to governance: NBS are often implemented in complex socio-environmental contexts, spanning jurisdictional boundaries with impacts that reach far downstream of the NBS action, which requires cooperation and consensus of a variety of stakeholders. This can impinge on the rights of local communities who may be impacted unintentionally or left out of design and discussions on benefits. There are therefore higher risks that mitigation will be reversed and “leakage” into unintended outcomes in another area (Doelman et al., 2020; Chagas et al., 2020). For example, protection or reforestation can lead to unmet demand for agricultural products and an increase in prices, subsequently increasing deforestation in another area to meet that demand (Popp et al., 2014). Leakage can also happen over time, if efforts to uptake carbon are not permanent and just delay the emissions.

There are large assumptions about the sustainability of NBS actions because of the often long time scales (Seddon et al., 2020). For example, short-term investments in monocultures can lead to diminished storage within a short period depending on effectiveness of management. In some cases this may return the extra stored carbon back to the atmosphere and in the worst case may lead to land degradation and further emissions. For naturally regenerating forests, the timescale of payback in net carbon emissions could be over decades, during which net emissions could increase because of required land use change or implementation of management or technologies (e.g., biochar production) before a net reduction is achieved. NBS can then reach saturation point whereby the uptake of carbon is balanced by emissions. This is not necessarily problematic in itself, but does set a limit on the additionality of the action and a stronger emphasis on the preservation and management of the system, including policy to protect such systems.

The prevalence of monocultures for tree planting is already generating problems. Tree plantations are the most popular commitment in government pledges under the Bonn Challenge and are also a risk. Ecologically, large plantations of monocultures present multiple disbenefits in terms of their impact on biodiversity, erosion and water quality and many other ecosystem services. They can also incentivise commercial exploitation of forestation and can lead to land grabbing with impacts on land rights including indigenous rights to land and their ecosystem services. Monocultures can displace other land uses such as agriculture with impacts on local food security and land clearance for plantations can negate any gains made in the tree carbon storage. Furthermore, they can detract from other forms of NBS such as wetlands that should be part of a broader portfolio of actions across landscapes. Fundamentally they provide quick carbon mitigation gains but relatively lower longer-term sustainability of carbon storage, and store less carbon than more diverse forests and much less than natural forests. Large-scale initiatives such as the Bonn Challenge to restore 350M ha of degraded and deforested land by 2030, and the World Economic Forums 2019 “1 Trillion Trees” initiative could be contributing to mono-culture planting.

There are risks that NBS will incentivise the status quo of fossil fuel use. As mentioned previously, implementation of NBS at scale is a key element of a portfolio of measures needed to reduce global warming to safe levels which includes rapid and drastic cuts in the use of fossil fuels. Continued use of fossil fuels could be facilitated by the wider availability of NBS-based carbon storage to offset the emissions and lessen the incentives (Seddon et al., 2021). Similarly, sector-based carbon offset schemes such as CORSIA for the aviation industry may disincentivise the industry to innovate in their use of energy sources. In turn, this may reduce incentives for reducing consumption and demand for energy, most of which is in the developed world; carbon credits tends to be focused on offsetting consumption in the developed world based on solutions implemented in the developing world, creating a disconnect in a global problem. Many of these risks are articulated in a recent open letter from a set of concerned NGOs to the CoP26 presidency and the UNFCC/CBD (NGOs, 2021).

**b) Climate resilience of projects**

NBS may also become less effective in the future (see next subsection) because of how changes in climate may reduce their effectiveness in slowing down the rate of change in climate. There are risks about the resilience of land-based NBS projects because of the tight relationship between vegetation productivity and health and climate variability, and the sensitivity to extreme climate-related events. Some of these have been mentioned previously in relation to specific approaches. In summary, the largest risks are because of climate change leading to more frequent, long, and severe droughts that can hamper vegetation growth, and in some circumstances lead to die-off. This has been seen in forest ecosystems across the western U.S. and elsewhere (Clark et al., 2016) and is a concern for the future (Anderegg et al., 2020). Often this is exacerbated by risks from pests and disease, which in turn can be exacerbated by climate change. Monocultures and mono-plantations are a relatively easy way of storing carbon quickly but are much more susceptible to climate-related disturbances that can reduce their carbon uptake abilities and their longer-term resilience, such as disease, pests, drought and wildfires. Low biodiversity value in such plantations can reduce their resilience in a changing climate, and essentially be maladaptive (Turner et al., 2020).

**c) Climate variability and emissions pathways**

Uncertainty in future climate is driven by uncertainty about which emissions pathway is taken by humanity and how climate evolves regionally, and uncertainty in the year-to-year and decade-to-decade variations in climate that occur naturally. Global climate models can provide projections of how climate may change in the future including the likelihood of climate related hazards that may affect NBS actions, but this is rarely taken into account in the design of NBS and assessment of their future effectiveness (Martín et al., 2021). Similarly, the future development of socio-economic environments in which the NBS sit can have a large bearing on their effectiveness. To reduce these risks, ex ante assessment needs to be done of how the NBS performs within its socio-ecological under a range of future scenarios using participatory systems modeling based.

**d) Disbenefits**

Much has been touted about the significant potential of NBS to contribute to solutions to the carbon mitigation problem. Much also has been noted and demonstrated on the co-benefits of NBS, which is a major selling point for their wider support and implementation. Less is discussed and researched on the potential disbenefits and unintended consequences. These can manifest in the form of acceptable trade-offs when considering a NBS project, but can also present unacceptable trade-offs if the disbenefit is too great. Like co-benefits, disbenefits can be in the form of ecological, socio-cultural and economic impacts or often some combination as these are often linked, and so strict safeguards are required to avoid these (IUCN, 2021).

In the context of disaster risk reduction, a wide number of co-benefits have been identified

## Risks around Land-based  nature-based solutions projects

(Ommer et al., 2022) as well as smaller number of disbenefits. In urban areas, improvements through greening can improve wellbeing and social inclusion, but can also increase house prices as demand from improved spaces increases, resulting in exclusivity of some parts of society (Bockarjova et al., 2020). High density of plantings aimed at sequestration can also provide a barrier for air pollution making conditions worse. Increased vegetation and wet areas can increase the prevalence of disease-vectors such as mosquitos and can increase pollen counts (Lyytimäki and Sipilä, 2009). In terms of land use change and management, such as afforestation or rewetting of wetlands, the benefits are generally clear and well-evidenced, but there are high risks of land use leakage as demand for the original land use is moved elsewhere.

Opportunity costs may arise, such as when the resources required for NBS, including ongoing management and monitoring, are redirected from or could have been used for other activities that could have other benefits, and may result in net-disbenefits including exclusion of some groups. There may be time dependent variations in costs and prices related to supply chains of NBS derived products as NBS is scaled up, such as wood products from forests becoming scarcer and driving up prices, or the converse when surplus wood builds up with impacts on the profits and livelihoods of smaller producers.

# Useful resources and links

**a) Literature on carbon science and nature-based solutions for carbon mitigation**

(Key sources; also see full reference list)

Friedlingstein, P., et al., Global carbon budget 2019. Earth Syst. Sci. Data 11, 1783–1838 (2019). 10.5194/essd-11-1783-2019

Girardin, C.A.J., Jenkins, S., Seddon, N., Allen, M., Lewis, S.L., Wheeler, C.E., et al. (2021). Nature-based solutions can help cool the planet — if we act now. Nature, 593, 191-194. doi: 10.1038/d41586- 021-01241-2.

Griscom, B. W. et al. Natural climate solutions. Proc. Natl Acad. Sci. USA 114, 11645–11650 (2017).

International Union for Conservation of Nature (2020). IUCN Global Standard for Nature-based Solutions: A User-Friendly Framework for the Verification, Design and Scaling Up of NbS. Gland. doi: 10.2305/iucn.ch.2020.08.en.

Roe, S., Streck, C., Beach, R., Busch, J., Chapman, M., Daioglou, V., et al. (2021). Land-based measures to mitigate climate change: Potential and feasibility by country. Global Change Biology [early view]. doi: 10.1111/GCB.15873.

Seddon, N., Chausson, A., Berry, P., Girardin, C.A.J., Smith, A. and Turner, B. (2020). Understanding the value and limits of nature-based solutions to climate change and other global challenges. Philosophical Transactions of the Royal Society B: Biological Sciences. 375(1794). doi: 10.1098/rstb.2019.0120.

Smith, P. et al. Biophysical and economic limits to negative CO2 emissions. Nat. Clim. Change 6, 42–50 (2016).

**b) Links to carbon assessment standards, guides and toolkits**

https://www.iucn.org/theme/nature-based-solutions/resources/iucn-global-standard-nbs - IUCN Gold Standard:

https://redd.unfccc.int - Reducing emissions from deforestation and forest degradation in developing countries (REDD+) web platform.

https://data.europa.eu/doi/10.2777/244577 - Evaluating the impact of nature-based solutions - A handbook for practitioners (EC, 2021).

https://data.globalforestwatch.org - FAO forest resource watch.

https://infoflr.org/ - IUCN InfoFLR web portal provides country-based information on country's forest landscape restoration targets and activities.

https://carbonpricingdashboard.worldbank.org - Carbon Pricing provides information on existing and emerging carbon pricing initiatives around the world.

https://maap.worldbank.org/#/homepage - World Bank Mitigation Action Assessment Protocol (MAAP) on performance and risks of climate actions.

https://klimalog.die-gdi.de/ndc - German Development Institute (DIE) NDC Explorer database and online visualization tool  to analyze and compare the quantitative and qualitative content of all (I)NDCs.

https://www.climatewatchdata.org – WRI/WB online platform with open climate data, visualizations and resources to gather insights on national and global progress on climate change focused on NDCs.

https://www.climatewatchdata.org/ndcs-sdg - WRI platform for identifying potential alignment between the targets, actions, policy measures and needs in countries' NDCs and the targets of the SDGs.

https://www.ndcs.undp.org/content/ndc-support-programme/en/home/impact-and-learning/library/nature-based-solutions-for-ndcs-toolkit-.html - NBS Toolkit captures more than 100 tools and resources on NBS that can support national decision makers as they enhance their NDCs.

https://naturebasedsolutions.org/about-us - WB Global Program on NBS for Climate Resilience to increase investments in solutions that integrate and strengthen natural systems across regions and sectors.

https://taan-adaptationdata.org - The Tool for Assessing Adaptation in the NDCs (TAAN) is a GIZ interactive knowledge platform that aims to provide an overview of and detailed information on the adaptation content of countries' NDCs.

https://ambitiontoaction.net/scan_tool/ - the SDG Climate Action Nexus tool (SCAN-tool) is designed to provide high-level guidance on how climate actions can impact achievement of the SDGs.

https://www.sei.org/projects-and-tools/tools/ndc-sdg-connections/ - NDC-SDG Connections tool analyses connections between climate change and the SDGs.

https://ec.europa.eu/info/business-economy-euro/banking-and-finance/sustainable-finance/international-platform-sustainable-finance_en - International Platform on Sustainable Finance (IPSF) is a forum for dialogue between policymakers, with the overall aim of increasing the amount of private capital being invested in environmentally sustainable investments.

https://www.naturebasedenterprise.eu – Global platform developed to connect market demand with the supply of NBS by organisations and enterprises, and to support the nature-based economy.

https://ec.europa.eu/info/news/evaluating-impact-nature-based-solutions-handbook-practitioners-2021-may-06_en - Evaluating the impact of NBS: a handbook for practitioners (EU).

http://www.eklipse-mechanism.eu – EU funded EKLIPSE impact evaluation framework for decision making on better-informed decisions on biodiversity and ecosystem services in Europe.

https://oppla.eu - EU funded knowledge marketplace showcasing the latest thinking on ecosystem services, natural capital and NBS, plus case studies.

https://networknature.eu – EU funded resource for the NBS community, creating opportunities for local, regional and international cooperation to maximise the impact and spread of NBS.

https://www.think-nature.eu – EU funded platform that supports the understanding and the promotion of Nature-Based Solutions (NBS)

https://naturvation.eu – EU funded project on NBS in urban environments.

https://una.city – NATURVATION web atlas of European city NBS case studies.

**c) Links to carbon certification schemes and critical analysis**

https://www.offsetguide.org - lists of “compliance” programmes run by governmental bodies and “voluntary” programmes run by NGOs. Provides critical analysis of schemes including highlights of high-quality and low-quality programmes.

https://cdm.unfccc.int/about/index.html - Certified Emission Reduction (CER) of the Clean Development Mechanism (CDM) (Developing countries) https://unfccc.int/process/the-kyoto-protocol/mechanisms/joint-implementation - Emission Reduction Unit (ERU) of the Joint Implementation (JI) (Developing countries)

https://www.rggi.org/allowance-tracking/offsets - RGGI CO2 Offset Allowance (ROA) of the Regional Greenhouse Gas Initiative (RGGI) (Northeast United States).

https://americancarbonregistry.org - Emission Reduction Tonne (ERT) of the American Carbon Registry (US)

https://www.climateactionreserve.org/how/offsets-marketplace/ - Climate Reserve Tonne (CRT) of the Climate Action Reserve (CAR) (US, Mexico)

https://www.goldstandard.org/articles/gold-standard-emission-reductions - Verified Emission Reduction (VER) of the Gold Standard (International)

https://www.planvivo.org/pvcs - Plan Vivo Certificate (PVC) of the Plan Vivo (International)

https://verra.org/project/vcs-program/verified-carbon-units-vcus/ - Verified Carbon Unit (VCU) of the Verified Carbon Standard (International)

**d) Links to relevant policies**

- International

https://unfccc.int/process-and-meetings/the-paris-agreement/the-paris-agreement - UNFCCC Paris Agreement

https://habitat3.org/the-new-urban-agenda/ - United Nations Conference on Housing and Sustainable Urban Development (Habitat III) New Urban Agenda

https://www.un.org/en/climatechange/2019-climate-action-summit - UN Climate Action Summit

https://www.cbd.int – Convention on Biological Diversity.

https://www.cbd.int/sp/targets/ - Aichi Biodiversity Targets

https://www.nbspolicyplatform.org - Nature-based Solutions Policy Platform – focused on climate change adaptation planning and policy.

- EU policy and strategies

https://ec.europa.eu/info/research-and-innovation/research-area/environment/nature-based-solutions/research-policy_en - EU NBS research policy.

## Useful resources and links

https://op.europa.eu/en/publication-detail/-/publication/31e4609f-b91e-11eb-8aca-01aa75ed71a1 - EU Biodiversity Strategy

https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal_en - EU Green Deal

https://www.eea.europa.eu/policy-documents/the-eu-forest-strategy-com - EU Forest Strategy

https://futurium.ec.europa.eu/en/urban-agenda/sustainable-land-use - Partnership on Sustainable Land Use and Nature-based Solutions of the Urban Agenda for the EU

https://www.gcca.eu - Global Climate Change Alliance Plus (GCCA+), an EU initiative to help vulnerable countries address climate change, largely based on nature-based solutions.

- UK policy

https://www.gov.uk/government/publications/25-year-environment-plan - UK Government's 25-year Environment Plan for England.

https://www.gov.uk/government/publications/budget-2020-documents/budget-2020 - UK budget 2020 includes the Nature for Climate Fund, the Nature Recovery Network Fund, and Natural Environment Impact Fund.

https://www.naturebasedsolutionsinitiative.org/wp-content/uploads/2020/12/NbSinUKPolicy_Dec2020.pdf - The Role of Nature-based Solutions for Climate Change Adaptation in UK Policy.

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# List of Boxes, Figures and Tables