Carbon-dioxide Removal and Biodiversity: A Threat Identification Framework

Kate Dooley, Ellycia Harrould-Kolieb and Anita Talberg
Climate & Energy College, University of Melbourne

Abstract

Carbon-dioxide removal (CDR) technologies offer the potential to contribute to the restoration and protection of natural ecosystems, the achievement of development goals and the safeguarding of human wellbeing. However, these technologies can also present risks to biodiversity, particularly those techniques that depend on large-scale manipulation of ecosystems and earth-system processes. Debates around the development of these technologies have historically focused on the dichotomy between the need to expand the knowledge base on all options related to emerging technologies, and the concern that research represents a slippery slope to deployment. This paper introduces a new approach to governing CDR research – one based on threat identification. We present a framework for assessing the impacts (positive or negative) on biodiversity and ecosystems from a spectrum of CDR interventions, so as to prioritize research to those CDR options that present minimal threats to biodiversity. Application of the framework indicates that while many CDR interventions present threats to biodiversity, certain options, such as regenerative CDR, may have positive impacts.

Policy implications

- A threat identification perspective that identifies impacts of proposed CDR options on the direct drivers to biodiversity loss suggests some types of CDR should be viewed as potentially harmful mitigation interventions, adding to the imperative for rapid and deep decarbonization to minimize future CDR reliance.
- Strong and effective CDR governance frameworks will be critical in cases where biodiversity impacts are dependent on the CDR implementation method.
- Ultimately, pursuing synergistic activities, such as CDR options that regenerate and restore nature, presents the lowest potential threat to biodiversity.
- The potential impacts of CDR activities on biodiversity need to be at the forefront of decision-making around whether to engage in research and the eventual deployment of these techniques.

Biodiversity and climate change

In May 2019, the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES) global assessment report concluded that 'nature and its vital contributions to people, which together embody biodiversity and ecosystem functions and services, are deteriorating worldwide' (Díaz et al., 2019, p. 3). The acceleration of biodiversity loss is increasingly recognized as a crisis, posing risks to humanity and ecosystems on the same scale as climate change (Díaz et al., 2019). The two are inextricably linked; degraded ecosystems release more carbon thereby increasing the mitigative burden of climate change (Lade et al., 2019). In turn, ecosystems require functional resilience to sequester carbon over the long-term, and to resist, recover or adapt to changing conditions and disturbances that are becoming more rapid and severe under climate change (Seddon et al., 2019). Yet projected scenarios for limiting warming to well below 2°C and 1.5°C rely on large-scale carbon-dioxide removal (CDR) (IPCC, 2018) without serious consideration of any potential threats posed to biodiversity (Díaz et al., 2019).

Assessments of CDR options tend to focus on costs and potential, with limited attention given to constraints and impacts (see, for example: Fuss et al., 2018; McLaren, 2012). The IPCC notes that few studies have specifically addressed the impacts of proposed land-based CDR on ecosystems and land degradation (IPCC, 2019). Even fewer analyses include impacts on marine ecosystems. Analysis of the deployment dynamics of large-scale CDR in a risk-management framework remains a research gap (Fuss et al., 2018). Research identifying CDR options that minimize negative impacts on biodiversity could help to prioritize further research and implementation of those options following no-regret principles. In this paper, we provide a policy-relevant assessment tool to identify the impacts of proposed CDR options on the direct drivers to biodiversity loss. This paper does not provide a broad assessment of a range of CDR limitations and potentials, but rather a specific assessment of the impact of CDR on biodiversity (and, by extension, on ecosystem function).

Assessing the impact of CDR on biodiversity

The Convention on Biodiversity (CBD) defines biodiversity as 'variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems' (CBD, 1992, Art. 2). Biodiversity underpins ecosystem functioning and the provision of all goods and services that are essential to human health and wellbeing (Hooper et al., 2005; Raffaelli, 2006), and is necessary to sustain key ecosystem functions, structure and processes (Morton and Hill, 2014). Biodiversity plays a key role at all levels of the ecosystem services hierarchy; as a regulator underpinning ecosystem processes, as a final ecosystem service and as a good that is subject to valuation (Mace et al., 2012). Biodiversity is measured across multiple levels and dimensions, through indicators such as species richness (number of species) and species abundance (number of individuals in each species) that show the direction in which the key components of biodiversity are heading. The majority of indicators of ecosystems and biodiversity show rapid decline (Díaz et al., 2019). However, as purely quantitative metrics, these indicators are often poor tools for understanding the decline in biodiversity.

CDR can be categorized on the basis of different characteristics, such as geographical (land / marine / technology) or process-based (chemical / mechanical / biophysical) and can be classified into ever-smaller subcategories (Fuss et al., 2018; IPCC, 2013; McLaren, 2012; Smith et al., 2019). For our analysis, we categorize CDR techniques into four distinct processes: those employing either land-use change; regenerative; marine; or chemical-based processes. The distinction is not perfect as there are overlaps between categories, but it allows a clear overview of the various CDR options.

The literature contains conflicting research findings on the environmental and social impacts of CDR. Smith et al. (2019), who presented a comprehensive assessment of the impacts of land-based CDR options on the Sustainable Development Goals (SDGs) and Nature's Contribution to People (NCP), suggested that all land-based CDR options have the potential to contribute positively to NCPs and SDGs. In contrast, the IPBES suggests that all proposed CDR methods included in the IPCC Fifth Assessment Report (AR5) will have negative impacts on biodiversity (with mixed impacts on ecosystem services) (Díaz et al., 2019). We therefore start from the premise that whether CDR contributes to biodiversity loss or enhances biodiversity conservation and protection depends on the specific CDR process, and that consideration of the impacts of each CDR technique on biodiversity must be central to CDR governance and decision-making.

We focus on an upstream measure for biodiversity loss rather than specific ecosystem goods and services and the subjective values of these. We use the drivers of biodiversity loss as measures for assessing the pressures on biodiversity. Drivers of biodiversity loss include direct drivers (land/sea use change, exploitation of ecosystems, pollution, etc.) and indirect drivers such as demographic, economic and technological trends (Balvanera et al., 2019). We focus here on the contribution of CDR options to the direct drivers as a method for prioritizing low-threat interventions and identifying high-threat interventions that should be reconsidered. This method has not been explored in the sustainability literature, and presents a promising option for more detailed analysis.

Method

IPBES classifies the aggregated impacts of human actions on nature into five categories of direct drivers (in order of greatest impact): land-use / sea-use change; resource extraction; pollution; invasive and alien species; and climate change (Balvanera et al., 2019). These direct drivers result from an array of underlying causes – the indirect drivers of change – which are in turn 'underpinned by societal values and behaviors that include production and consumption patterns, human population dynamics and trends, trade, technological innovations and local through global governance' (Díaz et al., 2019, p. 5). Figure 1 summarizes these five categories by providing an example of impact, the extent or severity, influence on other drivers, and factors that promote the driver. All five direct drivers show steady increases over the past five decades globally (Balvanera et al., 2019).

To improve the understanding of CDR impacts, we categorize CDR options broadly to include both land and marine options, and with enough specificity to distinguish ecosystem-based from industrial interventions (Figure 2). This categorization leaves out some CDR approaches such as ocean downwelling, and different end uses for biomass (including CCU) due to a lack of literature surveys that include these approaches, making assessment of their impacts difficult. In addition, our analysis is focused only on CO_2 removal techniques, although there is scope to expand our approach to methane capture methods.

We then constructed an evaluation framework based on the five direct drivers to biodiversity loss. We assessed each CDR option (Figure 2) for its potential impact on each of the five drivers (Figure 1), and ranked these impacts as positive, negative, dependent on implementation (positive-negative), or no impact. This valuation is presented in Figure 3, while the literature-based assessment used to identify the potential threat different types of CDR options pose to biodiversity is summarized in Section 4.

Contributions of CDR options to the drivers of biodiversity loss

In this section, we report on the findings of the threat identification framework, which evaluates the potential impacts of each CDR method against the five drivers of biodiversity loss.

Figure 1. The five categories of direct drivers to biodiversity loss (based on Balvanera et al., 2019).

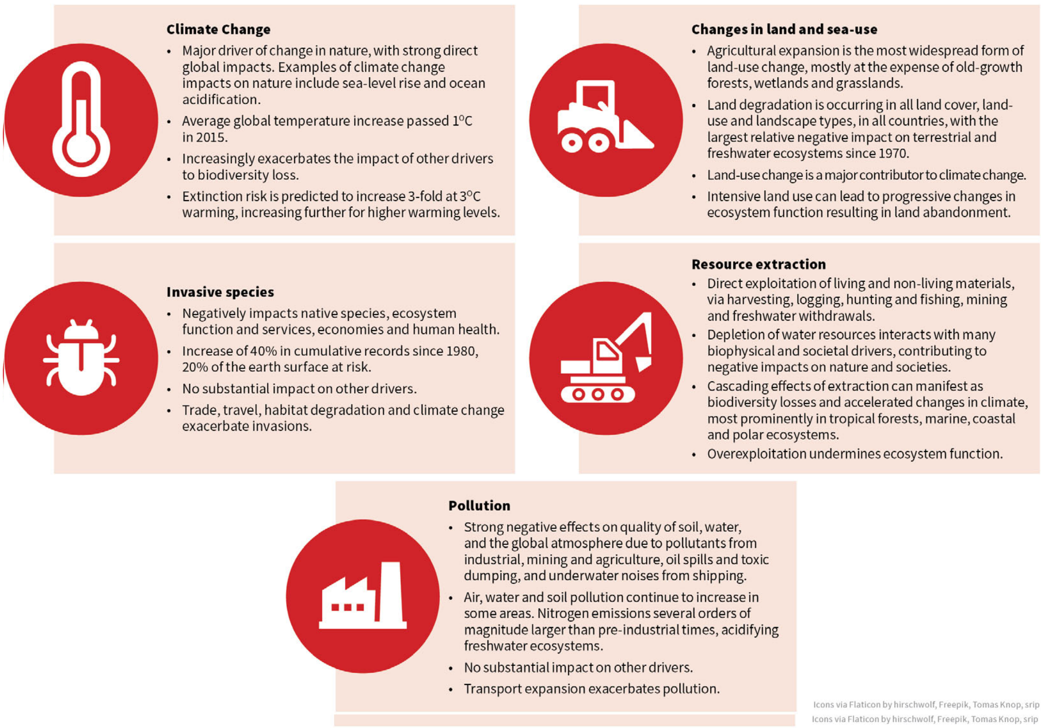
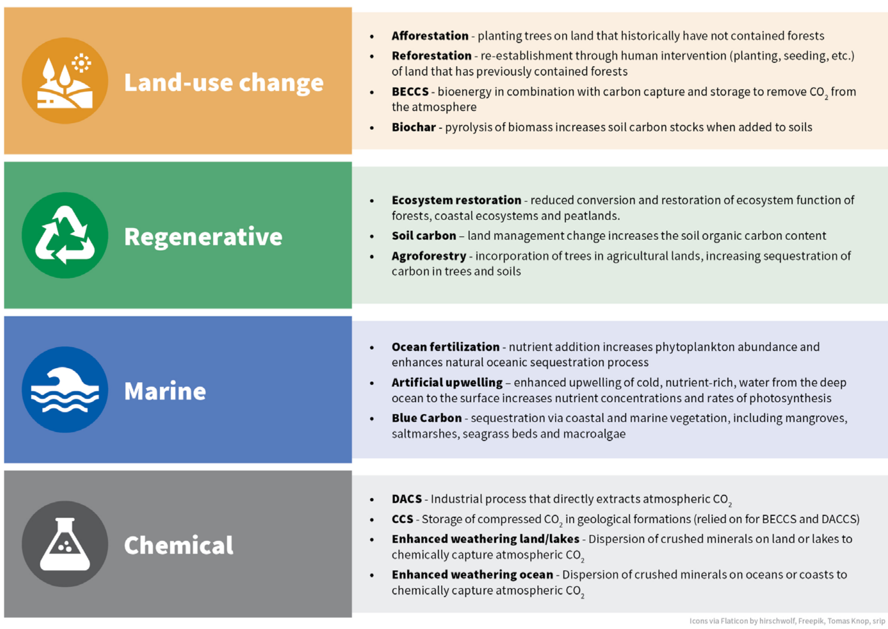


Figure 2. CDR categories

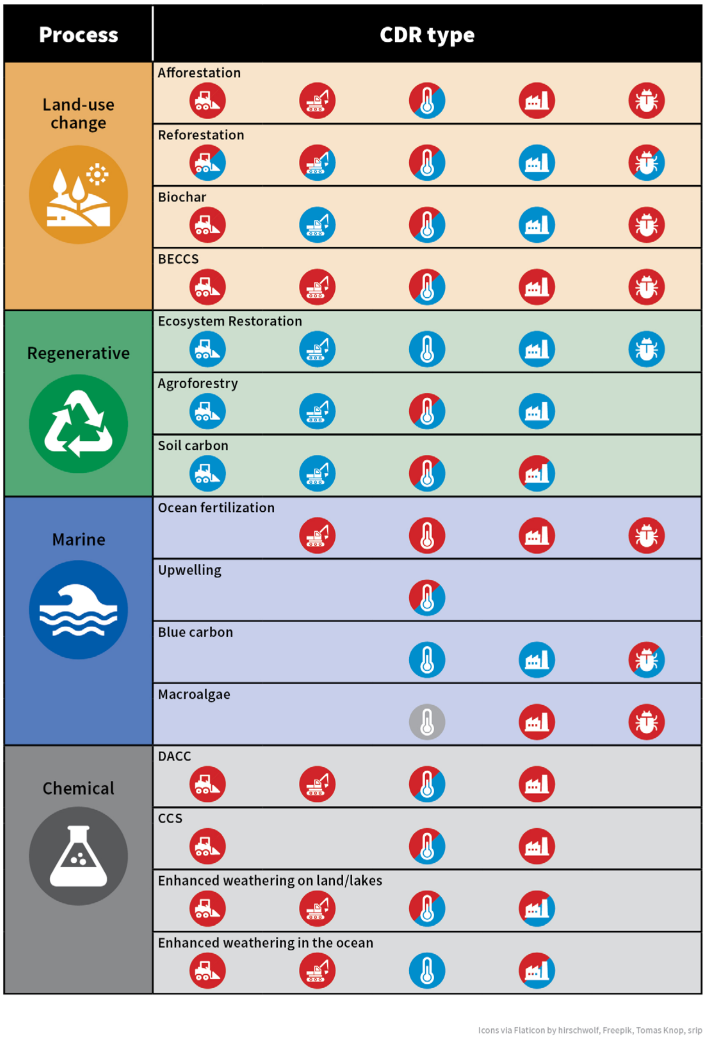


Land-use change

Land-use change, which has had the most impact on terrestrial ecosystems, is primarily driven by agricultural expansion (Balvanera et al., 2019). Land-use change is also inextricably linked with other direct drivers: agriculture and forestry are key introducers of invasive species, lead to increased nutrient and water resource extraction, which in turn leads to surface run-off and pollution.

Afforestation and reforestation differ significantly in their contributions to biodiversity loss, yet CDR reviews often treat these interchangeably (see, for example: Fuss et al., 2018; Smith et al., 2019). Reforestation of natural

Figure 3. Assessment of CDR options against the drivers of biodiversity loss. Red = negative impact; blue = positive impact; red/blue = impacts could be positive or negative depending on implementation; grey = impact unknown. Where there is no direct link between CDR types and drivers to biodiversity loss, icons are not shown. Literature to support the impacts identified here is summarized in Section 4.



forests on previously degraded lands can restore and rehabilitate lands with potential co-benefits (Lamb, 2005). Yet afforestation – planting trees in areas that do not naturally support forests – by definition introduces alien species, a direct drivers of biodiversity loss. Modern agriculture and forestry have introduced invasive and alien species to many areas, intentionally and unintentionally (IPCC, 2019). The impacts of reforestation vary with the method of implementation, and if based on introduced species, engender the same threat to biodiversity as afforestation (Lamb, 2005; Smith and Torn, 2013).

Trees require more water and soil nutrients than grassy biomes (Smith et al., 2019). Afforestation of grassland results in substantial resource input and loss of soil carbon from land conversion. Fertilizers can decrease soil nutrient availability, and increase run-off from forest plantations (Smith et al., 2019; Smith and Torn, 2013). Large-scale afforestation can deplete local water resources and disrupt hydrological cycles (Deng et al., 2017; IPCC, 2019; Lamb, 2005; Smith et al., 2019; Smith and Torn, 2013). Reforestation with introduced species will have the same requirement for nutrient and water input as afforestation. The majority of forest expansion occurs through natural succession, which minimizes the need for nutrient and water input to support forest growth (Chazdon et al., 2016). Reforestation of mixed native species, whether assisted or through natural succession, could increase biodiversity and restore waterways, reducing run-off and erosion (Chazdon et al., 2016; Lamb, 2005).

Increasing biomass supply for the scale of **Bioenergy with Carbon Capture and Storage (BECCS)** projected in deep decarbonization scenarios is likely to involve substantial land-use change. This section discusses only the biomass supply aspects of BECCS; CO_2 capture and storage are addressed below.

The IPCC notes that bioenergy can have negative or positive effects depending on the scale of deployment, feedstock and prior land use, and finds impacts to ecosystem services at 500–400 Mha of bioenergy crop extent (IPCC, 2019). Modeled 2°C and 1.5°C mitigation pathways rely on BECCS with median bioenergy crop area at or above this extent (Popp et al., 2017; Rogelj et al., 2018). Limiting deployment to marginal land is often recommended to avoid competition with arable land (Robertson et al., 2017). However, the lower yield output of marginal lands can result in more extensive land area requirement for bioenergy crops, potentially resulting in detrimental impacts on biodiversity through the extent of land conversion (Fuss et al., 2018; IPCC, 2019). Smith and Torn (2013) identify grasslands as particularly vulnerable to conversion for bioenergy crops. Impacts on biodiversity are also influenced by the crop itself, with cellulosic bioenergy crops showing potential to reduce or eliminate negative biodiversity impacts when trade-offs are carefully managed (Robertson et al., 2017).

Resource extraction for bioenergy crops, even at smaller scales, creates demand for fertilizer and water (Boysen et al., 2017; Heck et al., 2018; IPCC, 2019). Smith and Torn (2013) found that sequestering 3.6 Gt of CO_2 via BECCS would require 16–75 per cent of current global nitrogen fertilizer production, in excess of planetary boundaries for nitrogen flows (Boysen et al., 2017). Harvesting bioenergy biomass also depletes soil nutrients, particularly potassium (Smith and Torn, 2013). Intensive land management for large-scale expansion of bioenergy may result in nutrient leakage, with run-off to continental, coastal and marine water bodies causing species decline, eutrophication, toxin formation, and other impacts (Fuss et al., 2018; IPCC, 2019; Smith and Torn, 2013). These impacts are already accelerating from modern agricultural practices, the expansion of which, including for bioenergy crops, also results in the introduction of invasive species in different biomes (Díaz et al., 2019).

Effective climate mitigation via BECCS requires bioenergy to be carbon neutral, and to sequester additional carbon to compensate for process emissions (Fuss et al., 2018; Smith and Torn, 2013). If biomass crops displace existing land carbon stocks, carbon emissions can be higher than that removed via BECCS. This could take decades to centuries to be compensated by fossil fuel substitution or CCS (Davies-Barnard et al., 2015; Harper et al., 2018).

The application of **biochar** to agricultural soils brings productivity and resilience benefits and creates a stable carbon pool. Crop productivity has been shown to increase by 10 per cent on average following biochar soil amendment, while emissions of nitrogen and methane are reduced, and in temperate soils water losses are reduced (Fuss et al., 2018). However, biochar does not always have positive impacts on productivity and results show high variability (Jeffery et al., 2011). When realized, these productivity and other benefits could reduce demand for land, nutrient and water and enhance agroecosystem resilience when using biochar in agricultural soils, with positive benefits for food security, traditional farming and rural livelihoods,. Biochar can also build soil resilience to erosion, degradation, and contamination by absorbing both nutrients and pollutants, contributing to soil remediation and water purification (Smith et al., 2019).

These positive impacts could be tempered by additional land pressure if large quantities of biomass feedstock are required. At scales where biochar would contribute to climate mitigation, the requirement for biomass feedstocks contributes to biodiversity losses through the same drivers as discussed above for BECCS. While the use of agricultural and forestry biomass wastes as feedstock for biochar production would avoid competition for land (Smith et al., 2019), the limited availability of wastes constrains the scale and therefore the CDR potential of biochar.

Regenerative

Regenerative CDR options refer to the restoration of degraded lands, without change of land-use (eg: regenerative agriculture that increases productivity, or restoration of ecosystem function and carbon stocks in degraded forest).

Ecosystem restoration encompasses the restoration of degraded forestlands, peatlands and wetland ecosystems. While CDR research has often focused on afforestation and reforestation of previously cleared land, studies suggest that restoration of degraded natural forests should also be considered when developing terrestrial carbon management options (Asner et al., 2018; Böttcher et al., 2018; Roxburgh et al., 2006).

We refer to forest restoration as practices aimed at regaining ecological integrity in a degraded forest landscape (Smith et al., 2019), while reforestation (discussed above) would refer to the same practices in deforested landscapes. Restoring natural forest landscapes can enhance biodiversity, thereby improving ecosystem function and resilience, which decreases the risk of forest carbon stock reversal (Mackey et al., 2017; Smith et al., 2019). Degraded forests recover naturally if they are not further disturbed by intensive human activities (Grace et al., 2014). Integrating biodiversity considerations can result in resilient and long-term ecosystem restoration (Williamson and Bodle, 2016). Forest restoration may threaten livelihoods and local access to land if subsistence agriculture is targeted (Smith et al., 2019). The CBD definition of primary forest landscapes includes use by indigenous and local communities living traditional lifestyles, which conserve and sustainably use biological diversity (CBD, 2010).

Other degraded natural ecosystems, in particular grasslands, peatlands and wetlands offer carbon sequestration potential through restoration. Grasslands are ecosystems dominated by herbaceous and shrub vegetation, covering approximately 40 per cent of the ice-free land surface (White et al., 2000). Peatlands cover about 3 per cent of the terrestrial surface area, in all climatic regions, and store 21 per cent of the global total soil organic carbon stock (Leifeld and Menichetti, 2018). Intact peatlands contribute to a range of ecosystem functions such as habitat and biodiversity protection, water regulation and carbon sequestration and storage.

Soil carbon sequestration and agroforestry can occur over large land areas, without decreasing land availability for agriculture (Smith et al., 2019; Zomer et al., 2016). Soil carbon is directly linked to soil health. Increasing carbon in degraded soils improves soil structure, regulating water flow and reducing water and wind erosion. It also filters pollutants, which then protects freshwater and coastal waters. Increased ground cover to sequester soil carbon can reduce the vulnerability of soils to degradation and landslides (Smith et al., 2019). Agroforestry also contributes to increasing soil carbon, and provides benefits beyond carbon sequestration such as increased habitat and landscape connectivity, watershed conservation and positive impacts on hydrological cycles (Zomer et al., 2016).

While enhanced soil carbon mostly reduces pollution and improves soil quality, increasing soil carbon requires a commensurate increase in nutrients (nitrogen, phosphorous and potassium). This could be derived from additional organic matter, but if external nutrient addition is required, it adds to resource extraction for synthetic fertilizers, and increases pollution if nutrients are lost to water courses (Fuss et al., 2018). While the theoretical potential for CDR via soil carbon sequestration is high, permanence and saturation are barriers to effectiveness (Smith et al., 2019). Increasing tree cover on agricultural lands provides additional carbon sequestration (Griscom et al., 2017).

Marine

More than 65 per cent of the ocean is affected by changes in sea temperature, by-catch, habitat transformation, ocean acidification, and ocean pollution (Balvanera et al., 2019). Marine CDR options can add to these drivers or may contribute to and enhance ecosystem function.

While it is proposed that the process of fertilizing the ocean would draw-down atmospheric carbon and lock it away in deep ocean sediments, there are a variety of ways in which it can exacerbate climate change. The impacts of ocean fertilization begin with the upstream effects of resource extraction from the mining of substances to be added to the ocean, such as phosphate-bearing rocks. This is an energy intensive process that also requires purification (Lampitt et al., 2008). Manufacturing and transportation processes result in carbon emissions, with a footprint potentially larger than that of the carbon sequestered (Lampitt et al., 2008). The production of substantial algal blooms could increase heat transfer to the ocean surface via respiration, thereby increasing regional sea surface temperatures (Lawrence, 2002). The additional biomass produced that does not reach the ocean floor will release the captured carbon via remineralization (Lampitt et al., 2008). The movement of carbon dioxide from the upper to deep ocean will likely also shift ocean acidification from the surface to deeper waters (Cao and Caldeira, 2010).

The addition of nutrients can also cause eutrophication, lowering oxygen levels and potentially triggering fluxes of greenhouse gases with high warming potential, including methane and nitrogen (Law, 2008). This could negate any intended climate benefit (Jin et al., 2008). Eutrophication could also affect the abundance of phytoplankton species, including shifts towards harmful algal blooms and lowering biodiversity (Chisholm et al., 2001). Extended exposure to anoxia also leads to mortality, especially in sessile species, which can reduce the capacity of a system to support commercial fisheries and other ecosystem goods and services.

A redistribution of nutrients on a global scale may result from the substantial addition of nutrients, with some areas experiencing a decrease in nutrient supply and a subsequent decrease in biological productivity (Tripathy and Jena, 2019). This can result in a redistribution of phytoplankton, with colonizing and nuisance species moving into new areas (Lucas et al., 2007).

Artificial upwelling, which brings nutrient-rich waters to the surface to stimulate phytoplankton and draw down CO_2, also brings with it carbon-rich waters and will therefore not necessarily lead to a reduction in the concentration of carbon dioxide at the ocean's surface, which is a necessary condition for the enhanced uptake of CO_2 from the atmosphere (Oschlies et al., 2010). Uplifting large amounts of deep ocean water is highly energy intensive. Research into the use of renewables to power this uplift is ongoing, but these are not yet viable in the long-term (Pan et al., 2016). Upwelling of carbon-rich waters is also likely to exacerbate ocean acidification, which has been documented at natural upwelling sites along the west coast of the United States (Feely et al., 2008). It has also been suggested that deploying an array of ocean pipes bringing cooler waters to the surface could reduce sea-surface temperatures, although a cessation of the use of these pipes would likely result in an increase in sea-surface temperatures and atmospheric carbon to levels higher than would have occurred without the artificial upwelling (Oschlies et al., 2010). This transitory alleviation of climate change impacts is relevant to other forms of artificial upwelling as well. Moreover, while these activities will increase phytoplankton abundance in the upper ocean, most of the carbon taken up in their growth could be released back to the atmosphere within a year, with only a small proportion sequestered in the deep ocean (Shepherd et al., 2007). Similar to other forms of fertilization, artificial upwelling could shift species composition and therefore alter ecosystem function (Dutreuil et al., 2009).

Coastal **blue carbon** bears closer similarity to terrestrial CDR than open-ocean approaches of fertilization, upwelling and alkalinization. Tidal wetlands and seagrasses are some of the most productive vegetation types, but some of the most degraded natural systems globally. Revegetation and conservation are needed to realize their climate mitigating potential. Tidal wetlands sequester more carbon per unit than seagrasses; however, seagrasses have the potential for higher total carbon sequestration rates due to greater area coverage. Overall, through restoration and the creation of coastal wetlands, blue carbon (excluding macroalgae) has the potential to more than double the current rate of carbon dioxide removal (National Academies of Science, 2019). These systems continue to decline, with drainage and excavation estimated to release 450 million tons of carbon dioxide globally each year (Pendleton et al., 2012). This trend can be reversed through better management, increased restoration and wetland creation (National Academies of Science, 2019). Such efforts have the added benefits of providing coastal protection from storms and wave attenuation, habitat provision for wildlife and commercially important species, improvements in water quality and offsetting hypoxia (Barbier et al., 2011). These systems can also moderate local water chemistry, thereby alleviating ocean acidification in proximate waters (Sippo et al., 2016, 2019).

Macroalgae need to be considered separately to other coastal systems as they do not have root systems and soils to accumulate carbon; to ensure sequestration on a large scale, exportation and storage or use is required (Howard et al., 2017; Krause-Jensen et al., 2018). Most carbon from macroalgae is thought to remain within the carbon cycle due to herbivory and is not sequestered (Howard et al., 2017). Seaweed aquaculture could sequester carbon through technology to facilitate the export of seaweed to the deep ocean, the impacts of which are unknown (Froehlich et al., 2019). Unlike restoration and conservation of other blue carbon sites, the cultivation of macroalgae could change local hydrodynamics, increase disease and invasive species risks, and divert nutrients away from natural food webs (Campbell et al., 2019).

Chemical

The chemical process of removing atmospheric carbon can employ large machines that scrub the air as it blows past, can accelerate the natural process of weathering that usually takes place over millennia, and includes the geological storage of sequestered carbon.

Proposed **Direct Air Capture (DAC)** methods include adsorption, absorption, membrane or chemical looping (Al-Mamoori et al., 2017). The most developed techniques are absorption either in a highly or moderately alkaline solution (CBD Secretariat, 2012), and adsorption methods onto solid sorbents (Kulkarni and Sholl, 2012). Once the carbon dioxide has been extracted, it must be compressed and then used or stored. This section discusses the capture aspects only; storage is covered below.

Techno-economic analyses of DAC methods generally suggest that environmental impacts are minor, especially compared to other CDR methods (Williamson and Bodle, 2016). However, when assessed against the drivers of biodiversity loss, potential negative impacts appear. The land area required for DAC is less than that for biomass-based CDR methods, but because of the low concentration of carbon dioxide in the air and the need therefore to maximize air contact area, large machines are likely to be needed; and if siting is remote, access roads would be needed (CBD Secretariat, 2012; Royal Society, 2009). In relation to resource extraction, impacts depend on the type of sorbent used; for example, amine-based solvents require large amounts of water (CBD Secretariat, 2012; Royal Society, 2009). The various stages of a DAC process – including heating, cooling, running fans, and pumping chemicals – are highly energy intensive (Zeman, 2007); although, renewable energy driven methods have been proposed for both adsorption and absorption methods (Brethomé et al., 2018; Breyer et al., 2020; Wohland et al., 2018). The pollutive aspect of DAC depends again on solvent. Some chemical solvents may pollute the local water or air in their production process, and there is concern that air downwind of a DAC installation may have toxic elements and could affect vegetation because of their low carbon content (Socolow et al., 2011) – although environmentally friendly alternatives have been proposed (Brethomé et al., 2018).

Both DAC and BECCS rely on some form of carbon storage, the most common of which is **Carbon Capture and Storage (CCS)**. The environmental impacts of CCS include the land-use change resulting from drilling wells and pipelines and dust pollution from this. The CCS process also calls for substantial energy use (EEA, 2011). The potential for carbon-dioxide leakage is a major concern that could affect both aquifers and the local airspace and therefore threatens all lifeforms (Benson et al., 2012; Damen et al., 2006; EEA, 2011; IPCC, 2005).

Enhanced weathering on land or lakes could have some positive but also some negative impacts on the drivers of biodiversity loss. Mining the required minerals, for example limestone, could cause both land-use change and resource extraction impacts (CBD Secretariat, 2012; Royal Society, 2009). However, Rau (2011) proposes that waste stocks of limestone could be used to minimize the impact. Nonetheless, the energy costs of preparing the minerals are likely to be high (Williamson and Bodle, 2016). The application of the mineral could have additional beneficial climate change impacts. First, run-off into the oceans may reduce ocean acidity in the medium-term – although the initial spike in alkalinity could negatively impact ecosystems (CBD Secretariat, 2012; Hartmann et al., 2013). Second, more research is required to understand whether the lighter color of the minerals could increase soil albedo and thereby reduce localized warming (CBD Secretariat, 2012). Studies also suggest that the increased alkalinity of soils could benefit some species and vegetation productivity, but this is location- and species-dependent (Bronick and Lal, 2005; Haynes and Naidu, 1998; Holland et al., 2018). Similarly, increasing the pH of rivers and lakes could be considered a positive outcome, particularly if these had been previously acidified (CBD Secretariat, 2012; Köhler et al., 2010). However, a concern of adding olivine to soils relates to the risk of nickel accumulation, which could negatively impact on biodiversity (Berge et al., 2012).

Proposals for **enhanced weathering in the oceans** include adding lime, carbonate minerals, olivine or other silicate minerals to the ocean or coast (GESAMP Working Group, 41, 2019). As with enhanced weathering on land, there are potential land-use change and resource extraction implications from the mining of the chosen mineral (CBD Secretariat, 2012). Reduced ocean acidity is one likely impact; however, the sudden spike in alkalinity could have deleterious effects locally (Henderson et al., 2008; Renforth and Henderson, 2017). How the mineral is transported will determine whether there are associated ship emissions, for example, which have known polluting effects (Renforth and Henderson, 2017); the Royal Society (2009) have proposed pipelines instead. The activities of enhanced weathering could promote coral growth (Marubini and Thake, 1999; Renforth and Henderson, 2017), but additional research is needed to understand the wider marine ecosystem effects (Renforth and Henderson, 2017). There has also been the suggestion that the dissolution of iron and silica from the use of silicate minerals in enhanced ocean weathering could produce ocean fertilization effects (Renforth and Henderson, 2017), the impacts of which are detailed above.

Discussion

Land and sea-use change are the leading direct drivers of biodiversity loss (Díaz et al., 2019). CDR options requiring such changes at any scale (but particularly at the gigatons scale projected in climate scenarios) will add to already severe pressures on biodiversity. Land-use change is often linked to the other four drivers of biodiversity loss: resource extraction (for nutrients), pollution (from nutrient run-off), invasive species and climate change itself. The latter is because carbon is released through land-use change, which in the case of high-carbon ecosystems will not be recaptured on timescales relevant to climate mitigation objectives.

Our analysis (Figure 3) finds that some CDR methods have an overwhelmingly positive impact on biodiversity, ameliorating all drivers of biodiversity loss, while others have overwhelmingly negative impacts. The beneficial methods are regenerative CDR options. The exception being their impact on climate change where the reversible nature of terrestrial carbon storage presents a risk of failed mitigation. However, the restoration of degraded ecosystems and their ongoing protection improves resilience to external stressors and reduces the risk of reversal (Seddon et al., 2019). CDR options with negative impacts are those reliant on land-use change (in particular afforestation and BECCS), marine interventions with the exception of blue carbon (sharing traits with ecosystem restoration), and chemical interventions in terms of DAC, CCS and enhanced weathering in oceans, which raise many of the same impacts as ocean fertilization. In general, marine interventions that are not focused on restoration tend to cause pollution and invasive species, and in the case of ocean fertilization, resource extraction for mining phosphate-rich rocks (Lampitt et al., 2008). The effectiveness of ocean fertilization and enhanced weathering as climate mitigation strategies are questionable (Chisholm et al., 2001; Lampitt et al., 2008). DAC and CCS have negative impacts on resource extraction and pollution through the use of chemical sorbents (McCormack et al., 2016) and the risk of leakage from storage (Damen et al., 2006).

Some of the CDR methods analyzed could be either positively or negatively linked to the drivers of biodiversity loss, depending on how they are implemented. In these cases, strong governance frameworks are important. For example, whereas afforestation by definition represents an introduction of exotic species (risking invasive species, nutrient requirements and resultant pollution), reforestation could be either negative – if implemented via monoculture plantations thus carrying the same risks as afforestation; or positive – if implemented with native mixed species in a manner that buffers and reconnects primary forests. Similarly, enhanced weathering over land could provide positive outcomes if the minerals are drawn from waste stocks rather than freshly mined (Rau, 2011), if all processing is powered from renewable energy sources (Brethomé et al., 2018) and if the activities are targeted to restore the pH of acidified lakes (Köhler et al., 2010). Blue carbon encompasses a range of CDR methods that can positively impact biodiversity (such as restoration and conservation of coastal ecosystems (National Academies of Science, 2019)) or negatively impact (such as seaweed aquaculture which could alter local hydrodynamics, increase invasive species risks, and divert nutrients (Campbell et al., 2019)). Other interventions, such as biochar, can be positive at small scales (Smith et al., 2019), but at large scale bring the same negative impacts as land-use change reliant CDR options.

These CDR options represent strong governance challenges at global to local scales. Assessing the drivers of biodiversity loss in governance frameworks is one way to link both local and global ecosystem considerations. The threat identification framework presented here, by highlighting the impacts of CDR options on the drivers to biodiversity loss, can help to identify and eliminate dangerous CDR methods early, but can also guide decision-making around how, when and at what scale to implement various methods. Ultimately, pursuing synergistic activities, such as regenerative CDR options, presents the lowest threat to biodiversity. Nonetheless, policy-relevant assessment of individual CDR options requires context-based assessment with consideration of timeframe, scale, extent and reversibility.

Our framework offers a broad approach to guide CDR research efforts and to highlight positive options, while cautioning against progressing options that clearly contribute to known drivers of biodiversity loss. A similar framework could also be applied to research on options for other types of Greenhouse Gas Removal options (such as methane), to facilitate assessment of potential negative and positive impacts on biodiversity of such techniques. However, to ensure sustainable reliance of CDR methods, our framework alone is insufficient. Broad assessment frameworks must also be accompanied by quantitative and qualitative methods to assess and manage the timing, scale and cumulative effects of CDR interventions. It is widely accepted that the risks of CDR increase with the scale of intervention (Fuss et al., 2018). The IPCC reports a cumulative range of 150–1200 Gt CO_2 removal over the century (IPCC, 2018). Yet, recent research shows that the scale of required CDR depends on speed and urgency in reducing emissions (Strefler et al., 2018) and endogenous model assumptions such as discount rates (Emmerling et al., 2019), meaning that CDR at the top end of the predicted range cannot be taken as a given for 1.5°C compatible pathways.

Conclusions

The IPBES warns that the negative trends in biodiversity and ecosystem function are projected to continue or worsen, and that nature-friendly climate adaptation and mitigation will play important roles in achieving future societal and environmental objectives (Díaz et al., 2019). Our threat identification exercise suggests that, on the whole, reliance on CDR does not represent a nature-friendly mitigation strategy. Many CDR options drive the same processes that contribute to climate change and environmental degradation: land-use change and intensive agriculture for bioenergy crops; mining and resource extraction for DAC; pollution including greenhouse gases from ocean fertilization and upwelling. Any delay in mitigation that shifts the burden to future CDR options has a double-negative impact – first by increasing pressure on biodiversity through the processes that cause greenhouse gas emissions, and then again by increasing the drivers to biodiversity loss through the implementation of CDR options. With the exception of removal options that restore and regenerate nature, our analysis suggests that CDR should be viewed as an intervention with likely negative impacts on biodiversity loss, adding to the imperative for rapid and deep decarbonization to minimize future CDR reliance.

References

Al-Mamoori, A., Krishnamurthy, A., Rownaghi, A. A., and Rezaei, F. (2017) 'Carbon Capture and Utilization Update' *Energy Technology*, 5 (6), pp. 834–849.

Asner, G. P., Brodrick, P. G., Philipson, C., Vaughn, N. R., Martin, R. E., Knapp, D. E. et al. (2018) 'Mapped Aboveground Carbon Stocks to Advance Forest Conservation and Recovery in Malaysian Borneo', *Biological Conservation*, 217, pp. 289–310.

Balvanera, P., Pfaff, A., Viña, A., Frapolli, E. G., Hussain, S.A. & Merino, L. (2019) 'Status and Trends – Drivers of Change', in *Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Bonn: IPBES secretariat, p. 245.

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C. and Silliman, B. R. (2011) 'The Value of Estuarine and Coastal Ecosystem Services', *Ecological Monographs*, 81 (2), pp. 169–193.

Benson, S. M., Hepple, R., Apps, J., Tsang, C.-F. and Lippmann, M. (2012) Chapter 13: 'Carbon Capture and Storage', in *Global Energy Assessment - Toward a Sustainable Future*. Vienna: International Institute for Applied Systems Analysis; Cambridge and New York, NY: Cambridge University Press. pp. 993-1068.

ten Berge, H. F. M., van der Meer, H. G., Steenhuizen, J. W., Goedhart, P. W., Knops, P. and Verhagen, J. (2012) 'Olivine Weathering in Soil, and Its Effects on Growth and Nutrient Uptake in Ryegrass (Lolium perenne L.): A Pot Experiment', *PLOS ONE*, 7 (8), e42098. https://doi.org/10.1371/journal.pone.0042098

Böttcher, H., Hennenberg, K. and Winger, C. (2018) *Forest Vision Germany*. Berlin: Oko Institut.https://www.greenpeace.de/sites/www.greenpeace.de/files/publications/20180228-greenpeace-oekoinstitut-forest-vision-methods-results.pdf [Accessed 19 May 2019].

Boysen, L. R., Lucht, W., Gerten, D., Heck, V., Lenton, T. M. and Schellnhuber, H. J. (2017) 'The Limits to Global-Warming Mitigation by Terrestrial Carbon Removal', *Earths Future*, 5 (5), pp. 463–474.

Brethomé, F. M., Williams, N. J., Seipp, C. A. et al. (2018) 'Direct Air Capture of CO2 via Aqueous-phase Absorption and Crystalline-phase Release Using Concentrated Solar Power', *Nature Energy*, 3 (7), pp. 553–559.

Breyer, C., Fasihi, M. and Aghahosseini, A. (2020) 'Carbon Dioxide Direct Air Capture for Effective Climate Change Mitigation Based on Renewable Electricity: A New Type of Energy System Sector Coupling', *Mitigation and Adaptation Strategies for Global Change*, 25, pp. 43–65. https://doi.org/10.1007/s11027-019-9847-y

Bronick, C. J. and Lal, R. (2005) 'Soil Structure and Management: A Review', *Geoderma*, 124 (1–2), pp. 3–22.

Campbell, I., Macleod, A., Sahlmann, C., Neves, L., Funderud, J., Øverland, M., et al. (2019) 'The Environmental Risks Associated With the Development of Seaweed Farming in Europe - Prioritizing Key Knowledge Gaps', *Frontiers in Marine Science*, 6, pp. 1–22. https://doi.org/10.3389/fmars.2019.00107

Cao, L. and Caldeira, K. (2010) 'Can Ocean Iron Fertilization Mitigate Ocean Acidification?', *Climatic Change*, 99 (1–2), pp. 303–311.

CBD (1992) Convention on Biological Diversity. Entered Force 1992.

CBD (2010) Forest Biodiversity Definitions. Convention on Biological Diversity. Available from https://www.cbd.int/forest/definitions.shtml [Accessed 20 September 2018].

CBD Secretariat (2012) Geoengineering in Relation to the Convention on Biological Diversity: Technical and Regulatory Matters (No. 66), CBD Technical Series. Montreal.

Chazdon, R. L., Broadbent, E. N., Rozendaal, D. M. A., Bongers, Frans, Zambrano, A. M. A., Aide, T. M. et al. (2016) 'Carbon Sequestration Potential Of Second-growth Forest Regeneration in the Latin American Tropics', *Science Advances*, 2 (5), pp. 1–10. https://doi.org/10.1126/sciadv.1501639

Chisholm, S. W., Falkowski, P. G. and Cullen, J. J. (2001) 'Dis-Crediting Ocean Fertilization', *Science*, 294, pp. 309–310.

Damen, K., Faaij, A. and Turkenburg, W. (2006) 'Health, Safety and Environmental Risks of Underground Co2 Storage – Overview of Mechanisms and Current Knowledge', *Climatic Change*, 74 (1–3), pp. 289–318.

Davies-Barnard, T., Valdes, P. J., Singarayer, J. S., Wiltshire, A. J. and Jones, C. D. (2015) 'Quantifying the Relative Importance of Land Cover Change from Climate and Land Use in the Representative Concentration Pathways: Land Cover Change: Climate Versus Land Use', *Global Biogeochemical Cycles*, 29 (6), pp. 842–853.

Deng, Q., McMahon, D. E., Xiang, Y., Yu, C.-L., Jackson, R. B. and Hui, D. (2017) 'A global meta-analysis of soil phosphorus dynamics after afforestation', *New Phytologist*, 213 (1), pp. 181–192.

Díaz, S.Settele J., Brondízio, E., Ngo, H. T., Guèze, M. & Agard, J. et al. (2019) 'Summary for Policymakers of the Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services', pp. 44

Dutreuil, S., Bopp, L. and Tagliabue, A. (2009) 'Impact of Enhanced Vertical Mixing on Marine Biogeochemistry: Lessons for Geo-engineering and Natural Variability', *Biogeosciences*, 6 (5), pp. 901–912.

EEA (2011) Air Pollution Impacts from Carbon Capture and Storage (Technical report No. 14/2011). Copenhagen: European Environment Agency.

Emmerling, J., Drouet, L., van der Wijst, K.-I., Vuuren, Detlef van, Bosetti, Valentina and Tavoni, Massimo (2019) 'The Role of the Discount rate for Emission Pathways and Negative Emissions', *Environmental Research Letters*, 14 (10), pp. 1–11. https://doi.org/10.1088/1748-9326/ab3cc9

Feely, R. A., Sabine, C. L., Hernandez-Ayon, J. M., Ianson, D. and Hales, B. (2008) 'Evidence for Upwelling of Corrosive "Acidified" Water onto the Continental Shelf', *Science*, 320 (5882), pp. 1490–1492..

Froehlich, H. E., Afflerbach, J. C., Frazier, M., and Halpern, B.S. (2019) 'Blue Growth Potential to Mitigate Climate Change through Seaweed Offsetting', *Current Biology*, 29 (18), pp. 3087–3093.e3.

Fuss, S., Lamb, W. F., Callaghan, M. W., Hilaire, J., Creutzig, F., Amann, T. et al. (2018) 'Negative emissions—Part 2: Costs, potentials and side effects', *Environmental Research Letters*, 13 (6), pp. 1–47. https://doi.org/10.1088/1748-9326/aabf9f

GESAMP Working Group 41 (2019) High level review of a wide range of proposed marine geoengineering techniques (Report No. 98), Reports and studies. IMO.

Grace, J., Mitchard, E. and Gloor, E. (2014) 'Perturbations in the Carbon Budget of the Tropics', *Global Change Biology*, 20 (10), pp. 3238–3255.

Griscom, B. W., Adams, J., Ellis, P. W., Houghton, R. A., Lomax, Guy, Miteva, D. A. et al. (2017) 'Natural Climate Solutions', *Proceedings of the National Academy of Sciences*, 114 (44), pp. 11645–11650.

Harper, A. B., Powell, T., Cox, P. M., House, J., Huntingford, C., Lenton, T. M. et al. (2018) 'Land-use Emissions Play a Critical Role in Land-Based Mitigation for Paris Climate Targets', *Nature Communications*, 9 (1), pp. 1–13. https://doi.org/10.1038/s41467-018-05340-z

Hartmann, J., West, A. J., Renforth, P., Köhler, Peter, De La Rocha, C. L., Wolf-Gladrow, D. A., et al. (2013) 'Enhanced Chemical Weathering as a Geoengineering Strategy to Reduce Atmospheric Carbon Dioxide, Supply Nutrients, and Mitigate Ocean Acidification', *Reviews of Geophysics*, 51 (2), pp. 113–149.

Haynes, R. J. and Naidu, R. (1998) 'Influence of Lime, Fertilizer and Manure Applications on Soil Organic Matter Content and Soil Physical Conditions: A Review', *Nutrient Cycling in Agroecosystems*, 51, pp. 123–137.

Heck, V., Gerten, D., Lucht, W. and Popp, A. (2018) 'Biomass-Based Negative Emissions Difficult to Reconcile with Planetary Boundaries', *Nature Climate Change*, 8 (2), pp. 151–155.

Henderson, G., Rickaby, R. and Bouman, H. (2008) Decreasing atmosphere CO2 by increasing ocean alkalinity. The ocean dimension: would the concept work and what would be the environmental consequences? University of Oxford, Department of Earth Sciences and The James Martin 21st Century Ocean Institute.

Holland, J. E., Bennett, A. E., Newton, A. C.,White, P.J., McKenzie, B. M, George, T.S., et al. (2018) 'Liming Impacts on Soils, Crops and Biodiversity in the UK: A Review', *Science of the Total Environment*, 610–611, pp. 316–332. https://doi.org/10.1016/j.scitotenv.2017.08.020

Hooper, D. U., Chapin, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., et al. (2005) 'Effects of Biodiversity on Ecosystem Functioning: A Consensus of Current Knowledge', *Ecological Monographs*, 75 (1), pp. 3–35.

Howard, J., Sutton-Grier, A., Herr, D., Kleypas, J., Landis, E., Mcleod, E., et al. (2017) 'Clarifying the Role of Coastal and Marine Systems in Climate Mitigation', *Frontiers in Ecology and the Environment*, 15 (1), pp. 42–50.

IPCC (2005) 'IPCC Special Report on Carbon Dioxide Capture and Storage', Prepared by Working Group III of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press.

IPCC & (2013) 'Chapter 6: Carbon and Other Biogeochemical Cycles', in T. F. Stocker, D. Qin, G. K. Plattner, M. Tignor, S. K. Allen, J. Boschung, et al. (eds.), *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge: Cambridge University Press, pp. 465–570.

IPCC. (2018) 'Summary for Policymakers — Global Warming of 1.5 °C', in V. Masson-Delmotte, P. Zhai, H. O. Pörtner, D. Roberts, J. Skea and P. R. Shukla et al. (eds.), *Global Warming of 1.5°C. An IPCC Special Report on the Impacts of Global Warming of 1.5°C above Pre-Industrial Levels and Related Global Greenhouse Gas Emission Pathways, in the Context of Strengthening the Global Response to the Threat of Climate Change, Sustainable Development, and Efforts to Eradicate Poverty*. Geneva, Switzerland: World Meteorological Organization, pp. 1–24.

IPCC (2019) IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse gas fluxes in Terrestrial Ecosystems. Full report. Geneva: Intergovernmental Panel on Climate Change.

Jeffery, S., Verheijen, F. G. A., van der Velde, M. and Bastos, A. C. (2011) 'A Quantitative Review of the Effects of Biochar Application to Soils on Crop Productivity Using Meta-analysis', *Agriculture, Ecosystems & Environment*, 144 (1), pp. 175–187.

Jin, X., Gruber, N., Frenzel, H., Doney, S. C. and McWilliams, J. C. (2008) 'The Impact on Atmospheric CO_2 of Iron Fertilization Induced Changes in the Ocean's Biological Pump', *Biogeosciences*, 5, pp. 385–406.

Köhler, P., Hartmann, J. and Wolf-Gladrow, D. A. (2010) 'Geoengineering Potential of Artificially Enhanced Silicate Weathering of Olivine', *Proceedings of the National Academy of Sciences*, 107 (47), pp. 20228–20233.

Krause-Jensen, D., Lavery, P., Serrano, O., Marbà, Núria, Masque, Pere and Duarte, Carlos M. (2018) 'Sequestration of Macroalgal Carbon: The Elephant in the Blue Carbon Room', *Biology Letters*, 14 (6), pp. 2–6. https://doi.org/10.1098/rsbl.2018.0236

Kulkarni, A. R. and Sholl, D. S. (2012) 'Analysis of Equilibrium-Based TSA Processes for Direct Capture of CO2 from Air', *Industrial and Engineering Chemistry Research*, 51 (25), pp. 8631–8645. https://doi.org/10.1021/ie300691c

Lade, S. J., Norberg, J., Anderies, . M., Beer, C., Cornell, . E., Donges, J. F. et al.(2019) 'Potential Feedbacks Between Loss of Biosphere Integrity and Climate Change', *Global Sustainability*, 2,

Lamb, D. (2005) 'Restoration of Degraded Tropical Forest Landscapes', *Science*, 310 (5754), pp. 1628–1632.

Lampitt, R. S., Achterberg, E. P., Anderson, T. R., Hughes, J. A., Iglesias-Rodriguez, M. D., Kelly-Gerreyn, B. A., et al. (2008) 'Ocean Fertilization: A Potential Means of Geoengineering?', *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, 366 (1882), pp. 3919–3945.

Law, C. (2008) 'Predicting and Monitoring the Effects of Large-scale Ocean Iron Fertilization on Marine Trace Gas Emissions', *Marine Ecology Progress Series*, 364, pp. 283–288.

Lawrence, M. G. (2002) 'Side effects of oceanic iron fertilization', *Science*, 297, pp. 1993–1993.

Leifeld, J. and Menichetti, L. (2018) 'The Underappreciated Potential of Peatlands in Global Climate Change Mitigation Strategies', *Nature Communications*, 9 (1), pp. 1–7. https://doi.org/10.1038/s41467-018-03406-6

Lucas, M., Seeyave, S., Sanders, R., Mark Moore, C., Williamson, R. and Stinchcombe, M. (2007) 'Nitrogen Uptake Responses to A Naturally Fe-fertilised Phytoplankton Bloom During the 2004/2005 CROZEX Study', *Deep Sea Research Part II: Topical Studies in Oceanography*, 54 (18-20), pp. 2138–2173.

Mace, G. M., Norris, K. and Fitter, A. H. (2012) 'Biodiversity and Ecosystem Services: A Multilayered Relationship', *Trends in Ecology & Evolution*, 27 (1), pp. 19–26.

Mackey, B., Cadman, S., Rogers, N. and Hugh, S. (2017) 'Assessing the Risk to the Conservation Status of Temperate Rainforest from Exposure to Mining, Commercial Logging, and Climate change: A Tasmanian Case Study', *Biological Conservation*, 215, pp. 19–29.

Marubini, F. and Thake, B. (1999) 'Bicarbonate Addition Promotes Coral Growth', *Limnology and Oceanography*, 44 (3), pp. 716–720. https://doi.org/10.4319/lo.1999.44.3.0716

McCormack, C. G., Born, W., Irvine, P. J., Achterberg, E. P., Amano, T., Ardron, J., et al. (2016) 'Key Impacts of Climate Engineering on Biodiversity and Ecosystems, with Priorities for Future Research', *Journal of Integrative Environmental Sciences*, pp. 1–26.

McLaren, D. (2012) 'A comparative global assessment of potential negative emissions technologies. *Process Safety and Environmental Protection*, 90 (6), pp. 489–500.

Morton, S. and Hill, R. (2014) Chapter 1: 'What is Biodiversity, and Why is it Important?', in *Biodiversity: Science and Solutions for Australia*. Collingwood: CSIRO publishing, pp. 55–68.

National Academies of Science (2019) *Negative Emissions Technologies and Reliable Sequestration: A Research Agenda*. Washington, DC: National Academies Press.

Oschlies, A., Pahlow, M., Yool, A., and Matear, R. j (2010) 'Climate Engineering by Artificial Ocean Upwelling: Channelling the Sorcerer's Apprentice', *Geophysical Research Letters*, 37 (4), pp. 1–5. https://doi.org/10.1029/2009GL041961

Pan, Y., Fan, W., Zhang, D., Chen, J. W., Huang, H. C., Liu, S. X., et al. (2016) 'Research Progress in Artificial Upwelling and Its Potential Environmental Effects', *Science China Earth Sciences*, 59 (2), pp. 236–248.

Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., et al. (2012) 'Estimating Global "Blue Carbon" Emissions from Conversion and Degradation of Vegetated Coastal Ecosystems', *PLoS ONE*, 7 (9), e43542. https://doi.org/10.1371/journal.pone.0043542

Popp, A., Calvin, K., Fujimori, S., Havlik, P., Humpenöder, F., Stehfest, E., et al. (2017) 'Land-use Futures in the Shared Socio-economic Pathways', *Global Environmental Change*, 42, pp. 331–345. https://doi.org/10.1016/j.gloenvcha.2016.10.002

Raffaelli, D. (2006) 'Biodiversity and Ecosystem Functioning: Issues of Scale and Trophic Complexity', *Marine Ecology Progress Series*, 311, pp. 285–294.

Rau, G. H. (2011) 'CO2 Mitigation via Capture and Chemical Conversion in Seawater', *Environmental Science and Technology*, 45, pp. 1088–1092.

Renforth, P. and Henderson, G. (2017) 'Assessing Ocean Alkalinity for Carbon Sequestration', *Reviews of Geophysics*, 55 (3), pp. 636–674.

Robertson, G. P., Hamilton, S. K., Barham, B. L., Dale, B. E., Izaurralde, R. C., Jackson, R. D., et al. (2017) 'Cellulosic Biofuel Contributions to a Sustainable Energy Future: Choices and Outcomes', *Science*, 356 (6345), pp. 1–9. https://doi.org/10.1126/science.aal2324

Rogelj, J., Popp, A., Calvin, K. V., Luderer, G., Emmerling, J., Gernaat, D., et al.(2018) 'Scenarios Towards Limiting Global Mean Temperature Increase Below 1.5 °C', *Nature Climate Change*, 8 (4), pp. 325–332.

Roxburgh, S. H., Wood, S. W., Mackey, B. G., Woldendorp, G. and Gibbons, P. (2006) 'Assessing the Carbon Sequestration Potential of Managed Forests: A Case Study From Temperate Australia', *Journal of Applied Ecology*, 43 (6), pp. 1149–1159.

Royal Society (2009) *Geoengineering the Climate: Science, Governance and Uncertainty (Policy document)*. London: Royal Society.

Seddon, N., Turner, B., Berry, P., Chausson, A. & Girardin, C. (2019) 'Grounding Nature-Based Climate Solutions in Sound Biodiversity Science', *Nature Climate Change*, 9 (2), pp. 84–87.

Shepherd, J., Iglesias-Rodriguez, D. and Yool, A. (2007) 'Geo-Engineering Might Cause, Not Cure, Problems', *Nature*, 449 (7164), pp. 781–781.

Sippo, J. Z., Maher, D. T., Tait, D. R., Holloway, Ceylena and Santos, Isaac R. (2016) 'Are Mangroves Drivers or Buffers of Coastal Acidification? Insights from Alkalinity and Dissolved Inorganic Carbon Export Estimates Across a Latitudinal Transect', *Global Biogeochemical Cycles*, 30 (5), pp. 753–766.

Sippo, J. Z., Maher, D. T., Schulz, K. G., Sanders, C. J., McMahon, A., Tucker, J. and Santos, I. R. (2019) 'Carbon Outwelling Across the Shelf Following a Massive Mangrove Dieback in Australia: Insights from Radium Isotopes', *Geochimica et Cosmochimica Acta*, 253, pp. 142–158.

Smith, L. J. and Torn, M. S. (2013) 'Ecological Limits to Terrestrial Biological Carbon Dioxide Removal', *Climatic Change*, 118 (1), pp. 89–103.

Smith, P., Adams, J., Beerling, D. J.,Beringer, T., Calvin, K. V. and Fuss, S. et al. (2019) 'Impacts of Land-Based Greenhouse Gas Removal Options on Ecosystem Services and the United Nations Sustainable Development Goals', *Annual Review of Environment and Resources*, 44, 4.1–4.32. . https://doi.org/10.1146/annurev-environ-101718-033129

Socolow, R., Desmond, M., Aines, R. , Blackstock, J., Bolland, O., Kaarsberg, T. et al. (2011) Direct air capture of CO2 with chemicals: a technology assessment for the APS Panel on Public Affairs. American Physical Society, pp. 9

Strefler, J., Bauer, N., Kriegler, E., Popp, A., Giannousakis, A. and Edenhofer, O. (2018) 'Between Scylla and Charybdis: Delayed mitigation narrows the passage between large-scale CDR and high costs', *Environmental Research Letters*, 13 (4), 044015.

Tripathy, S. C. and Jena, B. (2019) 'Iron-Stimulated Phytoplankton Blooms in the Southern Ocean: a Brief Review', *Remote Sensing in Earth System Sciences*, 2, pp. 64–77. https://doi.org/10.1007/s41976-019-00012-y

White, R., Murray, S. and Rohweder, M. (2000) *Grassland Ecosystems: Pilot Analysis of Global Ecosystems*. Washington DC: World Resources Institute.

Williamson, P. (2016) 'Emissions reduction: Scrutinize CO2 removal methods', *Nature*, 530 (7589), pp. 153–155.

Williamson, P. and Bodle, R. (2016) *Update on Climate Geoengineering in Relation to the Convention on Biological Diversity: Potential Impacts and Regulatory Framework (No. 84), Technical Series. Secretariat of the Convention on*. Montreal: Biological Diversity.

Wohland, J., Witthaut, D. and Schleussner, C.-F. (2018) 'Negative Emission Potential of Direct Air Capture Powered by Renewable Excess Electricity in Europe', *Earths Future*, 6 (10), pp. 1380–1384.

Zeman, F. (2007) 'Energy and Material Balance of CO2 Capture from Ambient Air', *Environmental Science and Technology*, 41 (21), pp. 7558–7563. https://doi.org/10.1021/es070874m

Zomer, R. J., Neufeldt, H., Xu, J., Ahrends, A., Bossio, D., Trabucco, A., et al. (2016) 'Global Tree Cover and Biomass Carbon on Agricultural Land: The contribution of agroforestry to global and national carbon budgets', *Scientific Reports*, 6 (1), pp. 1–12. https://doi.org/10.1038/srep29987

Author Information

Kate Dooley is a Research Fellow at the University of Melbourne, currently looking at the potential for 'rights-based' approaches to restoration of natural ecosystems to remove and lock-up atmospheric carbon. Kate has a PhD in international climate politics from the University of Melbourne and a Masters in Environmental Technology from Imperial College London.

Ellycia Harrould-Kolieb has a PhD in the international governance of ocean acidification from the University of Melbourne. She also has a Masters of Environment from the University of Melbourne and Arts and Science degrees from Monash University. Previously Ellycia worked as Marine Scientist for Oceana, where she provided science-based policy advice for the Climate Change & Clean Energy campaign.

Anita Talberg has a PhD in climate governance from the University of Melbourne, plus a Masters in climate change and an engineering degree from the Australian National University. She previously worked for the Australian Parliamentary Library providing research and analysis to Members and Senators of the Australian Parliament on climate change and renewable energy issues.