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THE NEED FOR LOCAL GOVERNANCE OF GLOBAL COMMONS: THE EXAMPLE OF BLUE CARBON ECOSYSTEMS

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To limit global warming to 1.5°C, vast amounts of CO_2 will have to be removed from the atmosphere via Carbon Dioxide Removal (CDR). Enhancing the CO_2 sequestration of ecosystems will require not just one approach but a portfolio of CDR options, including so-called nature-based approaches alongside CDR options that are perceived as more technical. Creating a CDR "supply curve" would however imply that all carbon removals are considered to be perfect substitutes. The various co-benefits of nature-based CDR approaches militate against this. We discuss this aspect of nature-based solutions in connection with the enhancement of blue carbon ecosystems (BCE) such as mangrove or seagrass habitats. Enhancing BCEs can indeed contribute to CO_2 sequestration, but the value of their carbon storage is low compared to the overall contribution of their ecosystem services to wealth. Furthermore, their property rights are often unclear, i.e. not comprehensively defined or not enforced. Hence, payment schemes that only compensate BCE carbon sequestration could create tradeoffs at the expense of other important, often local, ecosystem services and might not result in socially optimal outcomes. Accordingly, one chance for preserving and restoring BCEs lies in the consideration of all services in potential compensation schemes for local communities. Also, local contexts, management structures, and benefit-sharing rules are crucial factors to be considered when setting up international payment schemes to support the use of BCEs and other nature- or ecosystem-based CDR. However, regarding these options as the only hope of achieving more CDR will very probably not bring about the desired outcome, either for climate mitigation or for ecosystem preservation. Unhalted degradation, in turn, will make matters worse due to the large amounts of stored carbon that would be released. Hence, countries committed to climate mitigation in line with the Paris targets should not hide behind vague pledges to enhance natural sinks for removing atmospheric CO_2 but commit to scaling up engineered CDR.

Keywords: Carbon Dioxide Removal, nature-based solutions, blue carbon ecosystems, common-pool resources, governance, property rights

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1 Introduction

The Intergovernmental Panel on Climate Change (IPCC) estimates that as of 2020 the limit for the emission of CO_2 in net terms is 400 Gt if the increase in global surface air temperature is to be limited to 1.5°C with a probability of 67% (IPCC, 2021). This means that emission scenarios in line with this remaining carbon budget entail a median of 730 Gt atmospheric carbon dioxide removal (CDR) by the end of the century in addition to globally coordinated deep emission cuts (Rogelj et al., 2018). Insufficient greenhouse gas and (especially) non-CO_2 mitigation would require larger amounts of CDR, as would the reduction of net uptake by natural carbon sinks. A distinction has been proposed between engineered, more technical, and nature-based solutions for CDR (Seddon et al., 2020a, 2020b), with current policies favoring atmospheric carbon removal via nature-based solutions (Seddon et al., 2019; Buylov et al., 2021). However, the distinction between engineered and nature-based solutions is by no means clear-cut but subject to societal construction (Bellamy and Osaka, 2020; Bertram and Merk, 2020).

Recent assessments of nature-based and engineered CDR approaches focus primarily on their technological availability and the potential scale of CO_2 removal (Royal Society, 2018; National Academies of Sciences, Engineering, and Medicine, 2019), neglecting the different governance challenges associated with these approaches. On the one hand, engineered solutions such as Direct Air Carbon Capture and Storage with verification of removal, permanent storage, and clear property rights, would seem to be candidates for a decentralized, market-based integration into climate policies by including them, for example, in emissions and offset trading (Rickels et al., 2021). On the other hand, the commons that provide the natural carbon sinks often lack such property rights and are threatened by local, regional and global pressures at the same time, imposing a particular governance challenge on these local commons.

We discuss potential governance challenges arising for nature-based CDR in a local-commons context. While land-based carbon removal approaches, especially forests, have received a lot of attention in the past in this regard (see e.g. Hatcher, 2009), the local commons governance challenges of ocean-based carbon removal approaches have received less attention. We thus focus on different blue carbon ecosystems (BCEs). The first challenge is to avoid emissions by preserving these ecosystems and their carbon stocks. The second challenge is to increase the carbon uptake while not compromising on or neglecting other, more valuable ecosystem services BCEs provide. The third challenge is to incentivize the provision of ecosystem services and govern BCEs in a way that preserves or improves local livelihoods, respects cultural contexts, and reconciles the local, regional, and global demands for the multiple ecosystem services BCEs provide. We argue, that a structured evaluation of local governance schemes similar to the work done by the *International Forestry Resources and Institutions* (IFRI) research network on common-pool forest management is needed to identify best practices for BCE management.

2 Conservation, restoration and creation of blue carbon ecosystems

Coastal ecosystems include mangroves, salt marshes, and seagrass meadows. They provide various services, including carbon uptake and storage. This is reflected in the introduction of the term “blue carbon” to heighten public awareness of the importance of the carbon sequestration potential of marine and coastal ecosystems (Nellemann et al., 2009; Macreadie et al., 2019; Lovelock and Duarte, 2019). However, the term blue carbon is neither restricted to the ecosystems listed above nor to only naturally occurring carbon sequestration. Macroalgae ecosystems are also covered by the term and these “may be supporting higher global C burial rates than seagrass, tidal marshes, and mangroves combined” (Macreadie et al., 2019, p. 4). Blue carbon sequestration can also be achieved by utilization of marine biomass. Carbon fixed in marine micro- and macroalgae could be turned into fuels or used for electricity generation and could provide CDR when combined with carbon capture and storage (Krause-Jensen and Duarte, 2016; Williamson et al., 2022).The aggregate annual carbon sequestration of blue carbon ecosystems is rather low, even though mangroves, salt marshes, and seagrass meadows sequester carbon per unit area at significantly higher rates than forests (cf. e.g. Duarte et al., 2010; Mcleod et al., 2011). According to Bertram et al. (2021) mangroves, salt marshes, and seagrass meadows have an annual carbon sequestration of 24.0 [Standard Error of the Mean, (SEM) 3.2] MtC y^{-1}, 13.4 [SEM 1.4] Mt C y^{-1}, and 43.9 [SEM 12.1] MtC y^{-1}, respectively, totaling 81.2 MtC y^{-1} across all these BCEs. This is less than one percent of the annual fossil fuel and industrial CO_2 emissions in 2020 (10.9 GtC, Friedlingstein et al., 2020). Furthermore, the prospects of achieving “extra” CDR via restoration are fairly low even though their marginal carbon removal per area is high since suitable areas are limited. Assuming that 40 percent of historical BCEs were restored, the estimated additional annual removal would be about 50 MtC y^{-1} by the year 2050 (Williamson, 2022). Accordingly, the National Academies of Sciences, Engineering, and Medicine (2019) estimate that in US coastal wetlands an additional cumulative removal of 1,500 MtC by 2100 could be achieved via various approaches to restoration. In comparison, achieving blue carbon sequestration via magroalgae plantations and utilization of marine biomass, feasible annual carbon removal is estimated to range between 0.8 and 1.1 GtC y^{-1} (National Academies of Sciences, Engineering, and Medicine, 2019, Williamson et al., 2022).

The advantage of BCEs over, for example, terrestrial forests is their capability to sequester carbon continuously as “sediments accrete vertically in response to rising sea level” (Mcleod et al., 2011, p. 554). Carbon density is highest for mangroves (502 tC ha^{-1}), followed by marshes (265 tC ha^{-1}) and seagrasses (111 tC ha^{-1}), resulting in estimated global carbon stocks of 7.3 GtC for mangroves, 5.6 GtC for marshes, and 5 GtC for seagrass meadows (Goldstein et al., 2020). In comparison with terrestrial ecosystems, only peatlands (~500 tC ha^{-1}) can match the carbon density of mangrove ecosystems. The share of BCEs’ soil carbon in total carbon (biomass + soil carbon) is significantly larger than the respective share for most types of terrestrial forest. Consequently, the percentage of initial soil-organic carbon typically lost in conversion or ecosystem loss is substantially larger for BCEs (81% for mangroves) than for terrestrial forests (< 30%). For mangroves, for instance, nearly two-thirds of the carbon initially stored in the biomass or the soil is considered irrecoverable (Goldstein et al., 2020). Depending on assumed carbon density and annual hectares lost, emissions from the decline of mangroves alone could lie between 40 and 186 MtC y^{-1} (Howard et al., 2017), the latter being slightly below fossil and industrial CO_2 emissions in Germany in 2019 (192 MtC, Friedlingstein et al., 2020). It is therefore essential to protect existing blue carbon ecosystems to avoid additional emissions.

3 Blue carbon ecosystems: ecosystem services and pressures

While blue carbon sequestration recently received a lot of attention, coastal ecosystems actually provide many further benefits and services on different geographical scales (see Table 1) which are more valuable than their carbon storage. They contribute to all the main categories of ecosystem services established in the Millennium Ecosystem Assessment (2005): (1) *supporting services* by sediment formation, nutrient cycling, and as a habitat for aquatic species; (2) *provisioning* of food and materials, like timber or fish stocks; (3) *regulating services,* as BCEs purify the water through their absorption of pollutants and excess nutrients, reduce coastal erosion, offer protection against floods, and sequester significant amounts of carbon; (4) *cultural services* in the form of spiritual or recreational value to residents and tourists (Vegh et al., 2019). The regulating services in particular can play an important role in mitigating and adapting to climate change (Duarte et al., 2013), especially since BCEs are able to self-maintain and even repair after major storms (Gedan et al., 2011) and even hold out prospects for adaptation to future sea-level rise (Williamson et al., 2022).

Costanza et al. (2014) estimate the global value of coastal ecosystems, including all coastal ecosystem services, to be US\$ 31.6 tr yr $^{-1}$. In comparison, Bertram et al. (2021) estimate the value of global BCE carbon storage to be one order of magnitude lower, at around US\$190.7 bn yr $^{-1}$ (± 29.5 bn). Thus, the value of natural, non-enhanced carbon storage is quite low compared to BCEs' overall contribution to wealth. Also accounting for associated non-CO_2 emissions, in particular CH_4, BCE restoration targeted at carbon sequestration only is estimated to cost between 491 USD/tCO_2 and 560 USD/tCO_2 for coastal wetlands and mangrove restoration, respectively (Taillardat et al., 2020). By contrast, considering carbon removal as one of various gains from BCE restoration projects would result in "additional" carbon-removal monitoring costs between only 0.75 and 4 USD/tCO_2 for tidal wetlands and seagrass meadows, respectively (National Academies of Sciences, Engineering, and Medicine, 2019).

Consequently, integrating BCEs into (international) climate policy regimes that operate with carbon-only price signals may actually not only fail to bring about socially optimal outcomes but might even push the whole system into a less desirable state than it was in before (e.g. Atchison, 2019; Song et al., 2021). This means that services ranging from enhancing the local population's livelihood and sustaining biodiversity to providing intangible values enhancing the cultural heritage (Smith et al., 2019) could be underprovided if BCEs are only conserved and restored based on the costs and benefits of carbon sequestration and storage.

Simply creating a CDR "supply curve"[1] where all carbon removals are considered to be perfect substitutes would imply that relatively cheap, technical carbon removal without co-benefits would be preferred over more expensive blue carbon management with very high local, regional, and even global co-benefits. Looking at nature-based solutions maximizing only carbon removal will almost certainly not be an appropriate way of realizing or even maintaining the various other ecosystem services. This applies in particular to their in-situ value and the necessity to avoid their destruction. A reduction in the size of one BCE has non-linear effects: on the one hand, it is likely to induce a reduction in other ecosystem services. In a large-scale restoration project on the other hand, Orth et al. (2020) observe a massive spillover from

[1] A CDR supply curve is created by plotting the different CDR options sorted by abatement costs against their potential to remove carbon from the atmosphere.

restoring seagrass meadows, finding that the seeded area of 213 ha resulted in a total revegetated area of 3612 ha.

The health and existence of BCEs and terrestrial ecosystems are threatened by various pressures that originate from local, regional, and global levels (see Table 1). Examples include deforestation, land-use change, boating or dredging, and climate change (Ahmed & Glaser 2016; Turschwell et al. 2020). In particular, BCEs are impacted by eutrophication caused by agricultural runoff (De los Santos et al., 2019; Lovelock et al., 2019). The sources of pressure can often be characterized as diffuse, especially in connection with agricultural runoff, and it is often difficult to attribute such non-point source pollution to a specific actor. In addition, coastal ecosystems are at the boundary between land an ocean where the management systems of land-use and marine resources meet and potentially clash (O'Hagan et al. 2020). To manage BCEs effectively and to achieve a socially optimal outcome, pressures on, and co-benefits provided by BCEs have to be understood and considered, actors responsible for pressures and ecosystem services should be held accountable and benefit from the conservation and restoration of ecosystems.

4 Governance of blue carbon ecosystems

In a stylized management approach, the various services and disservices, plus pressures on the BCEs, would be quantified to derive optimal payments and compensations between the various stakeholders to achieve socially optimal BCE conditions.[2] Figure 1 illustrates this approach, considering the internalization of further services like water purification and depicting an idealized payment scheme with the local community as property-rights holder. The management approaches related to CDR that are currently under discussion are often based on the idea that regional and global stakeholders should compensate a local community providing access to a BCE for services like carbon storage by way of some international payment scheme (e.g. similar to REDD+) through international institutions (Ahmed & Glaser 2016; Herr et al. 2012; ICUN 2021). Figure 1 illustrates this situation by showing CO_2-based monetary payments in a different color. If compensation for CDR alone were paid, other ecosystem (dis)services and pressures on BCEs would be ignored, thus leading to a non-optimal outcome.

[2] One could argue that "pressures" and "services" are two sides of the same coin, as the services of the ecosystem may be to absorb pressure, i.e. eutrophication as a "pressure" may be counterbalanced by the purification "service" of the ecosystem. However, a pressure (negative externality) may be so strong as to actually destroy the ecosystem. Accordingly, we explicitly include both "pressures" and "services".

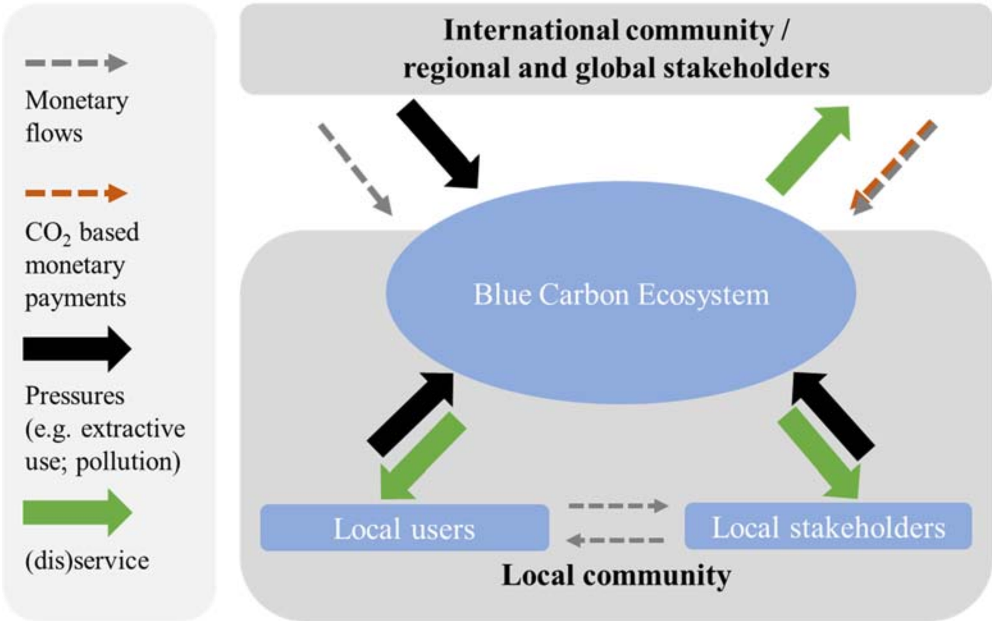

Figure 1: Idealized payment scheme for internalizing externalities (pressures and (dis)services) on different scales, assuming that the local community – consisting of users and other stakeholders - is the holder of the property rights.[3]

This approach, however, hinges on the central assumptions that all services and disservices can be measured and that the property rights can be clearly defined and enforced. On the local level, where most nature-based solutions would be realized, this is often not the case. Clearly defined and measurable management targets are often absent (Domínguez-Tejo & Metternicht 2018) and more importantly, BCEs can often be categorized as commons that are affected by the actions of a broad variety of user groups since they provide various ecosystem services for coastal communities. Without any (formal) assignment of property rights, an open access situation prevails, usually resulting in overuse of the resource and its respective services. The current situation related to coastal ecosystem services is far from ideal, e.g. as their direct and indirect benefits are rarely considered in decision-making (European Commission 2022), suggesting that property rights assignment or enforcement is mostly lacking. Even if property rights are assigned, this might only be the case for some but not all services. Furthermore, enforcement is often difficult, leading to de-facto open access regimes (Miteva et al. 2015). Just as for the other services, property rights frameworks for carbon removal are missing. In theory, for example a country or the developer of a project could own the carbon removal, but in practice, a related legal framework has been defined by only very few countries (IUCN 2021).

Coastal communities in developing countries often lack these resources and the capacities (Turschwell et al. 2020; Williams et al. 2020). While formal property rights might be missing in such contexts, there are implicit systems of de-facto-use rights of marine resources (Bennett 2016). But these customary tenure schemes do not automatically lead to sustainable outcomes (Ferse et al. 2010). For example, in the case of marine protected areas the environmental outcomes, such as an increase in the abundance of fish biomass and species or mangrove densities, do not only depend on the size, age, geographical isolation, and degree of

[3] We note that the division into 'services and 'disservices' can be a matter of subjectivity. While beach wrack is often seen as disservice polluting beaches, the wrack could also be used and would then count as 'service'.

fragmentation of an area, but also on the rules for access to it and the enforcement of protection (Edgar et al., 2014; Miteva et al. 2015; Turschwell et al. 2020). When the rules are forced upon communities this can lead to a boycott (Ferse et al. 2010). They will oppose the total closure of the area especially when the local population depends on extractive uses like timber or fodder from mangrove forests to sustain their livelihoods, (e.g. Badola et al., 2012; Roy, 2016). At the same time, the provision of other services like storm protection or nursery habitats can increase local support for mangrove conservation and restoration (Stone et al., 2008; López-Medellín et al., 2011; Badola et al., 2012). Even though it has long been recognized that community involvement is a crucial factor in the successful management of mangrove forests or marine protected areas (for an overview on mangroves see Arumugam et al. 2020), most marine protected areas are still implemented top-down (Ferse et al. 2010).

However, the mere inclusion of local stakeholders is not sufficient in itself. Trust in local management and clear benefit-sharing schemes are equally important (Arumugam et al., 2020; Ha et al., 2012; Wever et al. 2012). The rise of blue carbon and the growing attention to the ecosystems' carbon sequestration and storage service puts a spotlight on a before neglected service and raises interest among international actors. These new, external actors whose goal it is to create offsets for international carbon markets may be perceived as intruders by local communities (Gannon and Hulme, 2018). Furthermore, the focus on payments for CDR-only may lead to local governance geared to international money flows, thus leading to a relative neglect of local co-benefits and moving further away from the social optimum. If the local benefit-sharing rules are not well established, this can exacerbate inequalities in the local community and lead to continued exploitation of BCEs as poorer households continue to depend on it. This has similarly been observed for aquaculture, where the benefits accrue only for a few managers, while the pressures on the coastal ecosystems affect the entire community (van Oudenhoven et al. 2015).

Ferse et al. (2010) concluded in their review of studies evaluating the implementation of marine protected areas that a systematic survey and evaluation of local management practices was still absent. A more recent cross-country evaluation of the impact of anthropogenic pressures and regulatory strength on mangrove loss revealed no systematic patterns but highlighted the importance of national contexts (Turschwell et al. 2020). While in the meantime there are studies analyzing the local governance of mangrove forests in countries like Brazil and Indonesia (e.g. Miteva et al. 2015; van Oudenhoven et al. 2015; Wever et al. 2012), there is to the best of our knowledge no comparative study of best-practices on a community level looking at schemes to govern various BCEs especially when the global perspective of carbon storage is added. We, therefore, take a look at the experiences with forests as common-pool resources, the empirical evaluation of community forest management (CFM) and the role of the international REDD and REDD+ frameworks.

5 Property-rights regimes and lessons to be learned from the literature on common-pool resources

The net emissions from forestry and other land-uses including deforestation accounted for 11% on average of annual GHG emissions between 2007 and 2016 (IPCC 2019). This being so, international schemes have been discussed to compensate for (local) income losses resulting from refraining from deforestation (Hatcher, 2009). The idea is to internalize global benefits via payment schemes so that incentives to avoid deforestation are in place. Reducing emissions from deforestation and forest degradation (REDD) in combination with conservation of forest carbon stocks, sustainable management of forests, and enhancement of forest carbon stocks (REDD+) are frameworks developed under the UNFCCC to provide such incentives. However, REDD or REDD+ have been criticized for their failure to reduce deforestation and the fact that so far their contribution to climate change mitigation has been overestimated (West et al., 2020). One reason brought forward for the poor performance of this framework is that local communities and stakeholders have not been properly involved.

Several empirical studies investigating the performance of common-pool resource governance focus on forest management and thus provide potential insights for the management of nature-based CDR in general. Drawing upon data from the *International Forestry Resources and Institutions* (IFRI) research network, Hayes (2006) finds no statistically significant difference in forest conditions between legally protected areas (e.g., national parks or wilderness areas) and areas governed by local communities, suggesting that the outcomes of installing protected areas are not necessarily superior to the outcomes of management regimes with appropriate rules and the involvement of locals in common-pool forests.

Also referring to the IFRI project database, Chhatre and Agrawal (2008) demonstrate that local enforcement is strongly associated with forest regeneration. Furthermore, Chhatre and Agrwal (2009) find that local rule-making autonomy facilitates "win-win outcomes," i.e., improvements in livelihoods *and* carbon storage. From the same data, Coleman (2009) concludes that local sanctioning and monitoring seems to be a more decisive factor than the form of ownership. Investigating the determinants of local monitoring and sanctioning, Coleman and Steed (2009) find that the right to extract resources from the common-pool resource increases users' willingness to engage in monitoring and sanctioning. In a meta-analysis, Porter-Bolland et al. (2012) show that in their sample of 73 case studies the annual deforestation rates under CFM are lower and less variable than in protected areas, although deforestation occurs in both categories.

Brooks et al. (2012) use four dimensions (attitudinal, behavioral, ecological, economic) to analyze community-based conservation efforts regarding forests, grasslands, fisheries, and wildlife. Their main finding indicates that local capacity building, i.e., investments in human capital, is generally associated with positive outcomes. Oldekop et al. (2019) examine the effect of CFM in more than 3800 municipalities in Nepal, finding that CFM reduces both deforestation and poverty, although the initial poverty level moderates this effect. In a more recent meta-analysis, Hajjar et al. (2021) provide more nuanced insights into the effects of CFM on environmental outcomes, income, and resource access rights. In the majority of up to 524 cases, environmental indicators and livelihoods are positively affected by CFM; however, the commercial or subsistence access rights decreased in more than 50% of the 249 studies providing information on this category. This led to serious trade-offs between income and access rights, especially when elites benefited from formalized CFM while poor or marginalized groups lost their subsistence access. This shows that the design of benefit-sharing rules is important and should be included in any analysis, as only half of the observed sharing schemes resulted in more equitable outcomes. The conclusion is that the distribution of rights within local communities is an important success factor for CFM. Overall, the empirical findings described above are not conclusive in identifying the *one* perfect approach to managing forests or common-pool resources; however, they are conclusive in the sense that while there is no panacea, the inclusion of the local community and particular attention to local property rights are likely to favor positive environmental outcomes. As Ostrom (2010), the initiator of the IFRI research program, notes:

> "Our research shows that forests under different property regimes—government, private, communal—sometimes meet enhanced social goals such as biodiversity protection, carbon storage, or improved livelihoods. At other times, these property regimes fail to provide such goals. […] Thus, it is not the general type of forest governance that is crucial in explaining forest conditions; rather, it is how a particular governance arrangement fits the local ecology, how specific rules are developed and adapted over time, and whether users consider the system to be legitimate and equitable" (Ostrom, 2010, p. 658).

For the management of BCEs, one can conclude from the empirical studies that they can indeed, under the right circumstances and governance framework, contribute to positive ecological and socio-economic outcomes. The quote from Ostrom above also indicates the need for further research on the governance of BCEs, as the regime needs to fit the local ecology. Unlike in forest systems, where research has shown that extractive use increases the willingness to engage in monitoring efforts, extractive use may be less relevant for many BCEs as typical services include coastal protection and nutrient and pollution uptake (see Table 1). However, the range of regional and global pressures that affect BCEs directly or indirectly by changing their environment (see Table 1) demonstrates that even if local institutions prevent overuse of resources and foster sustainable outcomes, achievement of such outcomes cannot be taken for granted. For instance, a seagrass meadow may be protected locally, but nutrient spillovers into the sea could threaten its preservation. These aspects show that insights from the literature on the communal management of terrestrial forests can only be drawn upon in part for the governance of BCEs.

6 Discussion and conclusion

To limit global warming to 1.5°C, natural carbon sinks will have to be preserved, enhanced, and extended. Blue carbon ecosystems (BCE) can contribute to the achievement of this goal. However, it is still unclear how the pressures on these ecosystems from local, regional, and global actions can be reduced and their contribution to global carbon storage governed at the local level. We identify three main challenges pertaining to the conservation, restoration, and enhancement of BCEs for carbon storage.

The first challenge is that the potential for extending BCEs is low because the area available is limited. Thus, while prospects for achieving "extra" CDR via restoration of mangroves, saltmarshes, and seagrass meadows are unpromising, the destruction of BCEs would release a relatively large amount of carbon. New incentive schemes are needed that take this feature into account, and restorative measures should rather be assessed in connection with the aim of protecting existing meadows, i.e. keeping blue carbon stocks intact instead of considering them primarily as methods for carbon removal.

The second challenge relates to the fact that the wealth BCEs generate through carbon storage is small compared to their contribution via other provisioning, regulating, supporting, and cultural ecosystem services. A carbon-only price signal might not result in a socially optimal outcome as a "CDR" supply curve would imply that all units of carbon removed are perfect substitutes neglecting differences in co-benefits and in the case of ecosystem-based solutions underproviding other ecosystem services. Still, not all natural sink enhancement methods are equal. Accounting for CO_2 storage only could, for example, favor fast-growing monoculture reforestation over mangrove preservation despite the differences in additional services. Accordingly, there is a chance of preserving and restoring BCEs by considering all services in potential compensation schemes for local communities. However, instead of defining CDR targets, policies should first adopt precautionary management strategies for these ecosystems that take the various ecosystem services and their climate adaptation potential into account. The second step would be to look at carbon sequestration and storage in these ecosystems mainly as a co-benefit, not their main benefit.

The third challenge is how to govern natural sink enhancement. Since the services these ecosystems provide often display characteristics typical of common-pool resources or public goods, this poses a special management challenge. This is true in particular of marine ecosystems because high monitoring and enforcement costs for ocean environments imply that many BCEs are *de facto* open-access. Furthermore, ecosystem services accrue at different geographical levels. The major (ecosystem) services like coastal protection are provided at the local and regional level, while co-benefits like carbon sequestration are relevant at the global level. Similarly, a range of regional to global pressures affect BCEs directly or indirectly. Even if local institutions prevent overuse of resources and foster sustainable management – and strengthening local institutions seems necessary for the bid to achieve CDR and enhance co-benefits –, there is still no guarantee that the outcome will be good in environmental terms. In many cases, pressures are diffuse and difficult to attribute to a particular firm or person, e.g. in the case of non-point source pollution.

As an advance in the attempt to set up adequate governance systems for CDR by BCEs and to reverse the trend of destroying BCEs, we suggest taking detailed stock of current property rights allocations and local governance structures, focusing on the methods they use to incorporate BCEs' ecosystem services and hence how they interact with regional or international institutions. Local structures are particularly important here, as good management can lead to a trend reversal of destroying BCEs (De los Santos, 2019; O'Connor et al., 2020; Orth et al., 2020), but the other levels must not be left out of account. A modified IFRI-like questionnaire for BCEs should be devised to identify the factors that at various levels constitute successful governance regimes for BCEs. The current IFRI questionnaire collects information on a wide range of social predictor variables and ecological outcome variables, the latter including forest vegetation density and species diversity. Although there are several biophysical predictors such as elevation or temperature, most predictors relate to potential *local* user groups and associations and the activities and institutions related to forest use (Huntington et al., 2016). Only a few questions investigate the extent to which higher-level authorities supervise or interact with forest user groups. Given the variety of ecosystem (dis)services provided by BCEs to users at various geographical levels and the range of non-local pressures and benefits affecting BCEs (see Table 1), a suitable questionnaire would inquire into the ecosystem-specific circumstances in this connection and gather information concerning the property rights holders involved (Sikor et al., 2017). Also of interest are the regulatory, institutional, or organizational changes in the past that may have affected the BCE's ecological outcomes, e.g., regulations on the agricultural use of fertilizers or integration into an international offset regime. Based on results from such a survey, barriers to sustainable ecosystem management and best-practices could be identified, which would lay the base to improve governance.

Thus, the focus for supporting the use of BCEs and other nature- or ecosystem-based approaches for CDR may not be on international payment schemes like REDD+ but rather on international support for setting up suitable local management structures and on a better understanding of the governance of natural commons across different levels. However, focusing on setting up suitable local management structures also requires that countries committed to climate mitigation in line with the Paris targets do not hide the required amounts of CDR behind ill-defined approaches to the enhancement of natural sinks, but commit themselves to scaling up engineered CDR. Pinning excessive hopes on ecosystems to provide extra CDR will most likely not result in the desired outcome—neither for climate mitigation nor for ecosystem preservation. But ignoring their role in storing carbon is not an option either.

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Level / Ecosystem	Local		Regional / National		International / Global	
	Pressures	Ecosystem (Dis-)Service	Pressures	Ecosystem (Dis-)Service	Pressures	Ecosystem (Dis-)Service
Mangroves	Deforestation[19]: o Overharvesting of timber o Conversion to other land use Pollution, nutrients[19]	*Timber, fish stocks, other materials*[1] [P] Nutrient/pollution uptake[1] [R] Protection against storms, erosion control[1] [R] Provision of habitats[1] [S] Recreation, tourism[1] [C]	Nutrient spillover (e.g., from agriculture)	*Migrating fish stocks*[13] [P] Protection against storms, erosion control[1] [R]	Climate change impacts[15]	Carbon storage and sequestration[1,16,17] [R] N_2O and CH_4 emissions[8] [R]
Seagrass	Mussel beds[2,3] Pollution, nutrients[19] Boating, dredging[19] Loss of other ecosystems[19]	*Seagrass biomass* [P] Nutrient/pollution uptake[1] [R] Protection against storms, erosion control[1] [R] Habitat for aquatic species[1] [S] Recreation, tourism[1] [C] Wrack on beaches[12] [C]	Nutrient spillover (e.g., from agriculture)	*Migrating fish stocks*[13] [P] Protection against storms, erosion control[1] [R] Wrack on beaches[12] [C]	Climate change impacts[15] Ocean acidification[9,15]	Carbon storage and sequestration[1,16,17] [R] N_2O and CH_4 emissions[8] [R]
Salt Marshes	Conversion to other land use[19] Pollution, nutrients[19]	Nutrient/pollution uptake[1] [R] Protection against storms, erosion control[1] [R] Recreation, tourism[1] [C]	Nutrient spillover (e.g., from agriculture)	Protection against storms, erosion control[1] [R]	Climate change impacts[15]	Carbon storage and sequestration[1,16,17] [R] N_2O and CH_4 emissions[8] [R]
Macroalgae		*Seaweed biomass*[14] [P] Nutrient/pollution uptake[14] [R] Habitat for aquatic species[14] [S] Esthetic effects (buoys), noise (boats)[14] [C]	Nutrient spillover (e.g., from agriculture)		Climate change impacts [4,5,6,15] Ocean acidification[4,5]	Carbon storage and sequestration[10] [R] Halocarbon emissions[14] [R]
Terrestrial Forests	Deforestation[18]: o Overharvesting of timber o Conversion to other land use	*Timber, plants, game, freshwater* [P] Stabilizing the hydrological cycle, cooling effect[11] [R] Protection against soil erosion, avalanches, etc. [R] Provision of habitats [S] Recreation, tourism [C] Potential fuel for wildfires		Stabilizing the hydrological cycle, cooling effect[11] [R] Protection against soil erosion, avalanches, etc. [R]	Climate change impacts	Stabilizing the hydrological cycle, cooling effect[11] [R] Carbon storage and sequestration[16,17] [R]

[1] Vegh et al. (2019); [2] Peterson and Heck (2001) [3] Wall et al. (2008) [4] Koch et al. (2013) [5] Provost et al. (2017) [6] Smale (2020) [7] Thom and Seidl (2016) [8] Keller (2019) [9] Garrard and Beaumont (2014) [10] Krause-Jensen and Duarte (2016) [11] : Ellison et al. (2017) [12] Kirkman and Kendrick (1997) [13] Orth et al. (2020) [14] Hasselström et al. (2018) [15] Macreadie et al. (2019) [16] Mcleod et al. (2011) [17] Goldstein et al. (2020) [18] Skutsch and Turnhout (2020) [19] Lovelock et al. (2019)

Blue: Service / positive effect; Red: Disservice / negative effect; Black: Unclear

[R], [S], [P], [C]: Regulating, supporting, provisioning, cultural ecosystem service according to the Millennium Ecosystem Assessment (2005)

Table 1: Overview of services, disservices, and pressures for blue carbon ecosystems and terrestrial forests.

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