Assessing the risk of carbon dioxide emissions from blue carbon ecosystems

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"Blue carbon" ecosystems, which include tidal marshes, mangrove forests, and seagrass meadows, have large stocks of organic carbon (C_{org}) in their soils. These carbon stocks are vulnerable to decomposition and – if degraded – can be released to the atmosphere in the form of CO_2. We present a framework to help assess the relative risk of CO_2 emissions from degraded soils, thereby supporting inclusion of soil C_{org} into blue carbon projects and establishing a means to prioritize management for their carbon values. Assessing the risk of CO_2 emissions after various kinds of disturbances can be accomplished through knowledge of both the size of the soil C_{org} stock at a site and the likelihood that the soil C_{org} will decompose to CO_2.

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Destruction and degradation of natural ecosystems accounts for approximately 30% of the CO_2 released to the atmosphere, which helps drive global warming (Houghton 2003). Reducing land-use change is an important component of global strategies to curb atmospheric CO_2 emissions, thereby limiting and enhancing adaptation to anthropogenic climate change. Blue carbon ecosystems such as tidal marshes, mangrove forests, and seagrass meadows store large stocks of organic carbon (C_{org}) in their soils. Organic matter tends to accumulate in these ecosystems, due in part to their high rates of primary productivity, their ability to efficiently trap suspended particles, and their low exposure to wildfire. Moreover, their soils' hypoxic (low oxygen) conditions slow decomposition of organic matter and thus limit remineralization processes by which C_{org} is reconverted to CO_2 (Mcleod *et al.* 2011; Duarte *et al.* 2013). These coastal vegetated ecosystems have rates of C_{org} accumulation (or burial) per hectare that are estimated to be an order of magnitude greater than that of terrestrial forests (Mcleod *et al.* 2011; Duarte *et al.* 2013). Although the total global area of blue carbon ecosystems is two orders of magnitude smaller than that of terrestrial forest ecosystems, their global C_{org} burial capacity is comparable (Mcleod *et al.* 2011).

In a nutshell:

- Blue carbon ecosystems such as tidal marshes, mangroves, and seagrass meadows store large amounts of organic carbon in their soils; when these systems are degraded, some proportion of the organic matter is emitted to the atmosphere as CO_2, contributing to global warming
- A risk assessment framework can reinforce estimates of potential CO_2 emissions when blue carbon ecosystems are degraded
- Such a framework can act as a management tool to help prioritize mitigation or reduction of CO_2 emissions
- Sites with high soil carbon stocks where soil undergoes physical disturbance have the highest risk of CO_2 emissions, and this risk is accentuated in settings that favor the breakdown of organic matter to CO_2

Blue carbon ecosystems and the large C_{org} stocks accumulated in their soils and living biomass are vulnerable to a range of threats (Alongi 2002; Gedan *et al.* 2009; Waycott *et al.* 2009) that can result in their degradation (Figure 1). Approximately 35% of all mangrove forests have already been converted for aquaculture and other developments on tropical coasts (Alongi 2002). Seagrass meadows have experienced comparable losses and their loss rates are accelerating, mainly due to declining coastal water quality (Waycott *et al.* 2009). Tidal marshes have been drained for agriculture and other types of development for centuries and now occupy a small proportion of their original distribution (Gedan *et al.* 2009). These ecosystems are recognized for their large C_{org} stocks and carbon sequestration capacity, as well as for providing a wide range of other ecosystem services, such as supporting fisheries, biodiversity, coastal protection, and climate mitigation (Barbier *et al.* 2011; Duarte *et al.* 2013). This increasing appreciation, combined with the growing threats to these ecosystems, has given rise to the development of blue carbon strategies (Nellemann *et al.* 2009; Mcleod *et al.* 2011).

Blue carbon strategies focus on preserving and enhancing the C_{org} stocks and C_{org} burial capacity of tidal marshes, mangroves, and seagrass meadows, particularly within their soils. These strategies are meant to bolster the ability to adapt to and mitigate climate-change impacts,

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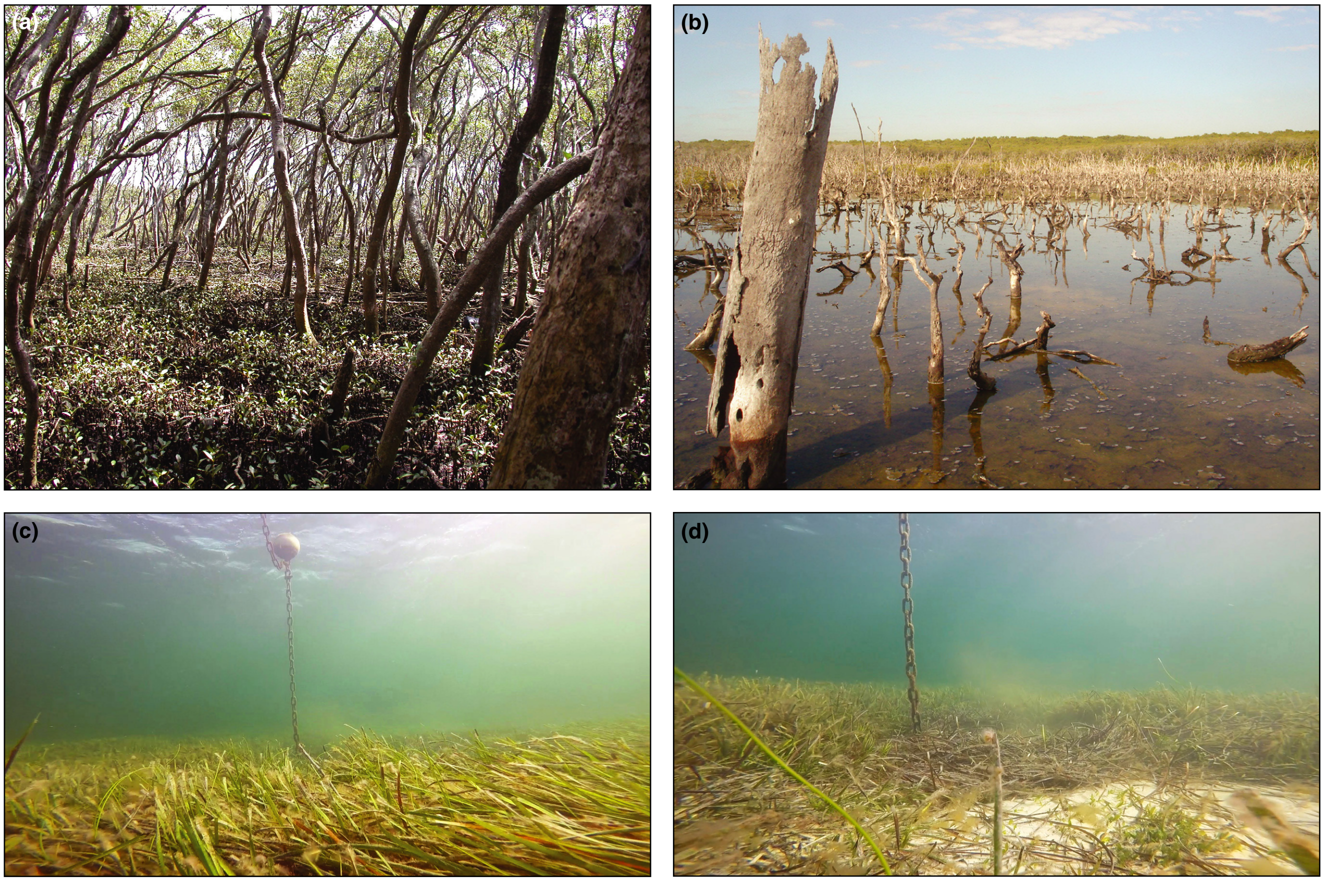

Figure 1. *Examples of healthy (a, c) and degraded (b, d) blue carbon ecosystems that result in CO_2 emissions to the atmosphere. Images are of mangrove forests (a, b) and seagrass meadows (c, d).*

and are based on the assumption that once blue carbon ecosystems are lost or degraded, a large fraction of the C_{org} contained within them will be decomposed and emitted as CO_2 to the atmosphere, thereby contributing to global warming (Pendleton *et al.* 2012). Additionally, loss and degradation of blue carbon ecosystems reduces their future potential role in mitigating greenhouse-gas emissions (Duarte *et al.* 2013). Financing mechanisms – including Reducing Emissions from Deforestation and Forest Degradation (REDD+), other mechanisms linked to the United Nations Framework Convention on Climate Change (UNFCCC), and voluntary markets – have been used (or proposed) to trade avoided CO_2 emissions from conservation and restoration of blue carbon ecosystems. However, to date, soil carbon has often been excluded from carbon credits associated with blue carbon projects, partly because of a lack of guidance for estimating CO_2 emissions from this important source (Wylie *et al.* 2016).

In terrestrial forests, the levels of CO_2 emissions from land-use change are relatively certain, since most of the C_{org} stock is found in the tree biomass, which is often combusted (GOFC-GOLD 2009). However, the main C_{org} pool in blue carbon ecosystems is found in the soil or sediment (Dontao *et al.* 2011; Fourqurean *et al.* 2012) so the proportion of C_{org} emitted as CO_2 after disturbance is less certain. While assessments of changes in C_{org} stocks (eg before and after land-use conversion) indicate that CO_2 emissions occur, there is uncertainty about the amount of soil C_{org} that is broken down or remineralized into inorganic forms of carbon. This uncertainty is attributed to the special characteristics of blue carbon ecosystems and is associated with various environmental factors that influence the breakdown of soil C_{org} to CO_2, including variations in the level of inundation with tidal water, exposure to high flow rates and waves, abundance of bioturbating (sediment-disturbing) organisms such as crabs, and the levels of connectivity to adjacent marine and terrestrial habitats. Yet enhancing the current understanding of potential CO_2 emissions resulting from ecosystem loss or degradation will make it easier for managers to include soil C_{org} in blue carbon projects and to make decisions that prioritize management of sites in ways that minimize and reverse CO_2 emissions from blue carbon ecosystems. Here we examine the processes influencing CO_2 emissions from these ecosystems and develop a risk assessment framework for managers to support conservation and restoration measures based on the potential for CO_2 emissions following loss or degradation.

Risk assessment frameworks have been extensively used in environmental impact assessments (Graham *et al.* 1991). These frameworks allow policy makers and regulatory agencies to establish and compare the risks that environmental hazards – including landscape management actions that produce undesirable ecological outcomes – pose for ecological assets (Graham *et al.* 1991). The frameworks have two goals: to provide (1) a systematic way to improve understanding in order to estimate the risks to ecological assets posed by these hazards and (2) a quantitative way of comparing risks associated with environmental problems and management actions. Ecological risk assessment frameworks are particularly useful in the absence of quantitative data when predicting how management actions might perturb ecological assets.

In this assessment, our ecological asset of interest is blue carbon ecosystem soil C_{org}, which may be emitted as CO_2 to the atmosphere (the "endpoint measure" in the ecological risk assessment terminology) under certain hazardous circumstances. Such circumstances include conversion to alternative land use, clearing, nutrient enrichment, storms, and climate change, among others. Below, we present an underlying process-based assessment of the influence of different hazards, component processes (eg excavation during land conversion, which increases exposure of soil C_{org} to oxidation), and environmental conditions affecting the likelihood that soil C_{org} will be emitted as CO_2.

Practicalities of estimating potential CO_2 emissions in blue carbon projects

Identifying the risk of CO_2 emissions following disturbance of blue carbon ecosystems is an important task and justifies the inclusion of soil C_{org} into blue carbon projects. Understanding the risks of CO_2 emissions can also help prioritize sites for conservation (Adame *et al.* 2015) and restoration (Mack *et al.* 2012; Knox *et al.* 2015). The IPCC Wetlands Supplement (http://bit.ly/2oRS9q6; WebTable 1) provides emissions factors for soil carbon for a limited range of project activities, including clearing of mangrove forests and rewetting of soils. These emissions factors are used in newly developed tools (eg FAO Ex-Ante Carbon-balance tool; http://bit.ly/2ohnbty; WebTable 1) and carbon market methodologies (Verified Carbon Standard; http://bit.ly/2pbd86p; WebTable 1) along with their supporting documentation, which includes discussion of critical social and economic challenges (Center for International Forestry Research; http://bit.ly/2pb16Ka; WebTable 1). However, the IPCC emissions factors and earlier methodologies (American Carbon Registry; http://bit.ly/2p3Gx6L; WebTable 1) cover few environmental hazards, because of the limited empirical data available for the wide range of hazards that can degrade blue carbon ecosystems (WebTable 2). The risk assessment framework we describe here provides a complementary approach that highlights the variation in CO_2 emissions among sites (and thus variation in the value of potential blue carbon projects) based on biogeochemical principles.

We propose a general scheme for identifying the key information required for determining how activities resulting in the loss or degradation of blue carbon ecosystems could also produce CO_2 emissions. Obtaining the essential information would necessitate assessing and clarifying:

(1) the extent and magnitude of the C_{org} stocks within the ecosystem;
(2) the nature (eg type and duration) of activities and hazards that may affect C_{org} stocks;
(3) the key physical and biogeochemical factors that act on the soil C_{org} after disturbance;
(4) the likely fate of the soil C_{org} (whether it has a high likelihood of being remineralized); and
(5) the risk of CO_2 emissions.

Below we provide a conceptual framework and the background needed to support the risk assessment (steps 1–5 above).

The conceptual framework

Remineralization of soil C_{org} to inorganic forms of carbon, which includes CO_2 that can be dissolved in water or liberated as gas to the atmosphere, is facilitated by physical and microbial breakdown of organic matter (Figure 2). Emissions of soil C_{org} to the atmosphere as CO_2 following a hazard or disturbance result from the alteration of the physical and/or biogeochemical environment in which the soil C_{org} was stored. The characteristics of the C_{org} stock, the manner in which the hazard influences the biogeochemical environment of the soil C_{org} stock, and the form in which the C_{org} is released to the environment (ie CO_2 gas, particulate organic carbon [POC], or dissolved organic carbon [DOC]) affect the overall level of CO_2 emissions (Figure 2). There can be direct release of CO_2 from soils to the atmosphere (arrow on far left of Figure 2), but if C_{org} is not immediately released as CO_2, a wide range of biotic and abiotic factors subsequently determine the potential for soil C_{org} to be ultimately converted to CO_2 (Figure 2). Below we briefly review the key variables and pathways influencing remineralization of C_{org} to CO_2, which can be used as a basis for assessing the relative risk of CO_2 emissions from soils exposed to different hazards.

Extent and magnitude of C_{org} stocks

In C_{org}-rich soils, soil depth and soil C_{org} density (grams of C_{org} per cubic centimeter of soil) are among the most important characteristics that determine both the value of the soil C_{org} asset and its vulnerability to emitting CO_2 when disturbed. Detailed guidance on how to quantify soil C_{org} stocks is available (Howard *et al.* 2014; WebTable 1). Typically, assessments of blue C_{org} stocks have been made over vertical soil depths of 1 m or less, mainly because data for deeper

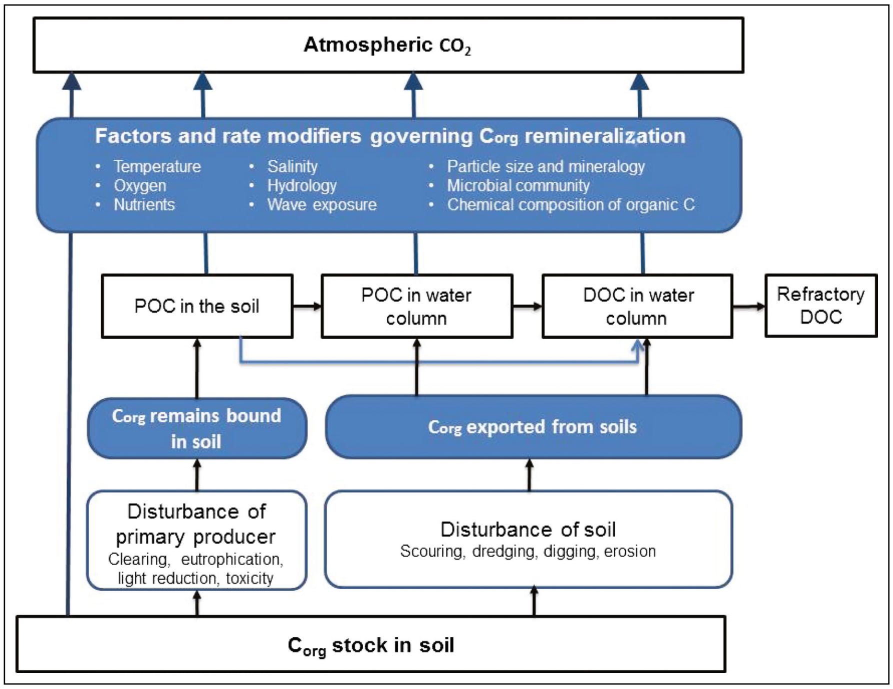

Figure 2. *Conceptual model for the potential remineralization of organic carbon (C_{org}) following disturbance of blue carbon ecosystems. POC = particulate organic carbon; DOC = dissolved organic carbon.*

layers are usually not available (Fourqurean *et al.* 2012). Although C_{org} stocks can extend deeper than 1 m (Mateo and Romero 1997; Kauffman *et al.* 2014), the stocks of C_{org} within the top meter have been assumed to include those that are most vulnerable to remineralization following disturbance (Pendleton *et al.* 2012; Jardine and Siikamäki 2014). C_{org} stocks vary among blue carbon ecosystems (Mcleod *et al.* 2011), among areas dominated by different species within similar ecosystems (Lavery *et al.* 2013; Saintilan *et al.* 2013), and across environmental gradients for the same species (Donato *et al.* 2011; Serrano *et al.* 2014). For instance, assessments of soil C_{org} in mangrove forests within different geomorphological settings show that levels of soil C_{org} tend to be higher in riverine areas than along ocean coastlines (Donato *et al.* 2011). However, C_{org} stocks in very low nutrient, non-riverine settings can be very large for both mangrove and seagrass ecosystems (Mateo and Romero 1997; McKee *et al.* 2007).

The vulnerability of soil C_{org} to remineralization differs among blue carbon ecosystems (Pedersen *et al.* 2011). Plant-derived C_{org} contains relatively high amounts of recalcitrant (resistant to decay) organic compounds in the plant tissues (eg lignin), which are difficult to remineralize compared to other sources of sedimentary C_{org} (seston or algae) (Marchand *et al.* 2005; Trevathan-Tackett *et al.* 2015). Despite low decay rates of C_{org} in blue carbon soils, changes to soil conditions associated with ecosystem disturbance or degradation may accelerate its remineralization (Moodley *et al.* 2005; Bianchi 2011). Thus, most of the C_{org} stock in blue carbon ecosystems can be assumed to be vulnerable to decomposition and therefore to be a potential source of CO_2 emissions following disturbance.

Activities and hazards that may affect C_{org} stocks

Most disturbances have an impact on the living biomass of blue carbon ecosystems, leading to CO_2 emissions as the source of blue carbon decomposes or is combusted (as in the case of mangrove wood), while reducing their capacity to continue to sequester C_{org} (WebTable 2). Disturbances capable of enhancing the remineralization rates of soil C_{org} stocks or reducing the accumulation rates of C_{org} can be classified into two types: (1) those that affect the primary producers, but do not directly disturb the soils (soils remain intact, following the left-hand pathway of Figure 2), and (2) those that result in the physical removal or alteration of the living biomass together with the soil C_{org} stock (the right-hand pathway of Figure 2). The latter type of disturbance rapidly exposes C_{org} to fundamentally different biogeochemical environments as well as disperses C_{org}, thereby enhancing the risk of CO_2 emissions.

Physical and biogeochemical factors acting on soil C_{org}

Where the soil is not physically altered by the disturbance but the vegetation dies or is removed, the soil C_{org} stock may possibly remain in place under biogeochemical conditions not conducive to remineralization (Macreadie *et al.* 2014). Alternatively, the soil C_{org} may leach to the overlying water (or tidal water) as DOC or dissolved inorganic carbon (DIC) (Bouillon *et al.* 2008; Maher *et al.* 2013), or may be released as gaseous CO_2 (Bianchi 2011). Tides and waves play an important role in pumping soil C_{org} from soils in both seagrass and mangrove habitats (Maher *et al.* 2013; Samper-Villarreal *et al.* 2016). Exposure to strong waves and currents (Marbà *et al.* 2015; Samper-Villarreal *et al.* 2016) or enhanced bioturbation (eg by crabs; Coverdale *et al.* 2014) may make soils more prone to erosion following vegetation loss, which can lead to larger-scale C_{org} losses than at the local scale of disturbance. Losses in C_{org} stocks can also be amplified by post-disturbance biogeochemical changes to soils. For example, draining blue carbon ecosystems for agricultural purposes can result in soil

Panel 1. CO_2 emissions from impounding coastal wetlands

In 1976, 110 hectares of mangrove forest, tidal salt marsh, and salt flats in the East Trinity Inlet near Cairns, Australia, were drained for sugar cane production (Figure 3). The draining of these highly organic soils exposed soil C_{org} to oxygen. This resulted in the production of 34 metric tons of sulfuric acid per hectare per year and an extremely acidic porewater (water within the sediment) pH of 3.2; the highly acidic water then drained into the adjacent estuary (Hicks *et al.* 2003). In addition to the production of acid and the associated environmental hazard, much of the site lost 1.3 m of soil elevation, from 0.9 m above sea level to 0.4 m below sea level. This loss of soil elevation was associated with an estimated loss of 680 metric tons of soil carbon per hectare (in both organic and inorganic forms) over 23 years (until 1999). CO_2 emissions associated with soil C_{org} remineralization during the disaster were estimated at 0.27 million metric tons of CO_2 (0.012 million metric tons of CO_2 per year) (Hicks *et al.* 2003). The total CO_2 emissions of the state of Queensland, Australia, in 1999 were about 100 Tg, making this 110 ha disaster a substantial component of the state's CO_2 emissions despite the small area of the site. There are 30,000 km^2 of potential acidic sulfate soils within estuaries in Australia alone, making them a potentially major source of CO_2 emissions if disturbed. The Queensland Government purchased the site in 2000 and began the process of restoration by reintroducing tidal flow and liming. Avoided CO_2 emissions and soil C_{org} gains associated with restoration have yet to be assessed.

Queensland Government

Figure 3. *Trinity Inlet, Cairns, Queensland, Australia. The levee wall used to impound the wetlands is on the right. Degradation of vegetation is evident on both sides of the levee wall. Acid was released, soil C_{org} was remineralized, and there was a 1.3 m loss in soil elevation (Hicks* et al. *2003).*

subsidence (Rojstaczer and Deverel 1995) as well as the formation and emission of CO_2 and sulfuric acid (Panel 1). In disturbed coastal ecosystems that remain unvegetated, there is an increased likelihood that these factors could lead to greater C_{org} losses over time.

If the soil is disturbed through excavation, for instance, once-buried C_{org} can be released to the environment in various forms, the nature of which depends partly on the characteristics of the disturbance. Following a physical disturbance (eg excavation during dredging or during construction of aquaculture ponds and pond walls), soil C_{org} can be released into the water column as POC, which can be removed from the site and experience several fates, including consumption, leaching of DOC, microbial remineralization, or photo-oxidation (Baldock *et al.* 2004) (Figure 2). The POC may be exported and physically degraded into smaller particles, which may become distributed throughout adjacent habitats and be buried again, or may release recalcitrant DOC (Blair and Aller 2012). However, ultimately, a large fraction of the POC transported to the oceans is likely to be remineralized and released to the ocean/atmosphere in the form of CO_2 (Cai 2011; Blair and Aller 2012).

Restoration of blue carbon ecosystems following disturbance can help to minimize or cap CO_2 emissions from disturbed and degraded sites (Duarte *et al.* 2013; Marbá *et al.* 2015) and also serve as an ecosystem-based measure to mitigate CO_2 emissions (Sutton-Grier *et al.* 2014). Creation and restoration of blue carbon ecosystems has mainly focused on mangroves and tidal marsh habitats; however, initiatives to restore seagrass beds are becoming more common and increasingly successful (van Katwijk *et al.* 2015). Restoration reinstates the sedimentary biogeochemical conditions and the soil stability in disturbed sites. It also enhances C_{org} storage by increasing the living biomass and its capacity to sequester CO_2 and by trapping organic material that is delivered in tidal flows (Panel 2).

Risk of soil C_{org} remineralization and CO_2 emissions

Qualitative tools can help to assess the relative importance of the factors influencing soil C_{org} remineralization at any given site, to direct and focus further analyses and corrective measures, and to develop our understanding of the risks of CO_2 emissions and, therefore, guide conservation strategies. Based on published evidence, Bayesian-based qualitative tables (AS/NZS 2004, a and b) can be used to rank the most important factors influencing the risk of CO_2 emissions for any given site or project. Bayesian approaches, where hypotheses are informed by existing evidence and understanding to formulate working models, allow the assignment of risk classes (Jones 2001). These models must be revised regularly as more evidence becomes available. For example, the availability of oxygen is important in determining rates of C_{org} remineralization of soil C_{org} (Moodley *et al.* 2005). Therefore, rates of C_{org} remineralization would be expected to be low with low-to-moderate C_{org} stocks under anoxic conditions (ie strongly reducing, redox potential $E_H < -100$ mV), which are typically associated with undisturbed mangrove, tidal marsh, and seagrass soils (scoring 1–3 in

Panel 2. Restoring blue carbon ecosystems: capping CO_2 emissions and restoring carbon accumulation

Tidal marshes

Restoration programs in tidal marsh habitats were some of the first initiatives to recognize degradation-associated soil C_{org} losses as a serious problem, and established the goal of increasing C_{org} accumulation in soils to mitigate climate change (Craft and Reader 1999; Connor *et al.* 2001; Mack *et al.* 2012). Reported C_{org} accumulation rates varied from 0.18 to 1.25 metric tons C_{org} ha^{-1} yr^{-1}, with an average value around 0.90 metric tons C_{org} ha^{-1} yr^{-1}. The time it takes for C_{org} levels to reach those of undisturbed marshes is variable and often slow (Burden *et al.* 2013). Craft and Reader (1999) showed that after 25 years, soil C_{org} stocks in a restored marsh were still lower than in a 2000-year-old undisturbed marsh. Similarly, Craft *et al.* (2003) reported that although most ecological attributes of restored marshes achieved equivalence to those of natural marshes in 5 to 15 years, the soil C_{org} content was still significantly lower in constructed marshes after 28 years, suggesting that at least 70 years were needed to fully recover soil C_{org} stocks. Burden *et al.* (2013) estimated it would take 100 years for restored marsh sites to achieve the level of C_{org} stocks of natural marshes.

Mangroves

Mangrove reforestation programs, largely focused on the recovery of the lost aboveground biomass, have proliferated worldwide over the past 50 years. However, studies initiated in the last few years have indicated significant losses in soil C_{org} also occurred with forest degradation and thus have begun to monitor the recovery of soil C_{org}, comparing soil C_{org} stocks among natural and restored or created mangrove forests (Osland *et al.* 2012; Salmo *et al.* 2013; Lunstrum and Chen 2014). Soil C_{org} accumulation rates reported for restored mangroves varied between 1.5–2.0 metric tons C_{org} ha^{-1} yr^{-1} and the time it takes for C_{org} in the upper soil layers (about 10 cm) to match that of undisturbed mangroves is estimated to occur 20–25 years after restoration (Osland *et al.* 2012; Salmo *et al.* 2013).

Seagrasses

About 80% of the *Posidonia* seagrass meadows of Oyster Harbour (southwest Australia) were lost between the mid-1960s and 1988 due to lower water quality associated with clearing of the catchment and application of fertilizers. ^{210}Pb dating of sediment cores revealed the erosion of the C_{org} deposit corresponding to 60 years of soil C_{org} sequestration (Marbà *et al.* 2015). In Virginia more than 1700 ha of *Zostera* seagrass were lost in 1933 due to wasting disease, with a hurricane also contributing to loss of soil C_{org} (Greiner *et al.* 2013). At both sites, restoration enhanced soil C_{org} sequestration over time due to increased plant biomass and shoot density contributing to C_{org} deposition and burial. Recovery of C_{org} burial rates in the restored *Posidonia* meadows was comparable to those of continuously vegetated sites (ie 0.25 metric tons C_{org} ha^{-1} yr^{-1}) within two decades and was estimated to reach those of continuously vegetated sites within 12 years in the *Zostera* sites. However, longer periods of time are required to achieve the levels of soil C_{org} stocks in natural meadows. Similar studies are not yet available for tropical seagrasses.

WebTable 1–3). In contrast, when large soil C_{org} stocks are exposed to oxic conditions (redox potential E_H > 400 mV), C_{org} remineralization rates are likely to be very high (scores of > 20; WebTable 3). Similar Bayesian-based tables could be constructed for other relevant environmental variables (eg temperature, salinity) based on published evidence or expert knowledge.

Determining the risk of CO_2 emissions

Risk tables, which establish the relative likelihood of soil C_{org} remineralization as a function of key environmental variables, are used in a subsequent step to estimate the relative risk of CO_2 emissions (Table 1). In Table 1, for instance, the scores of the risk of C_{org} remineralization are combined with an assessment of the size of the C_{org} stock to provide a relative estimate of the risk of CO_2 emissions.

The scores in Table 1 can be compared with existing case studies to assess the robustness of the risk scores. Some of the highest levels of soil C_{org} losses have been reported following the conversion of mangrove forests to aquaculture ponds in the Dominican Republic and within Southeast Asia, where soils containing large C_{org} stocks (> 500 Mg ha^{-1}) were excavated (Kauffman *et al.* 2014; see WebTable 4 for a list of case studies). Direct measures of CO_2 efflux from cleared mangrove soils in Belize showed high levels of CO_2 emissions (Lovelock *et al.* 2011), but lower CO_2 effluxes were observed in Indonesian mangroves, where soils had lower levels of C_{org} (Sidik and Lovelock 2013). Moderate-to-high C_{org} losses were reported from tidal marshes in regions of the US where soil had been eroded and dispersed due to intense bioturbation (13–54 metric tons CO_2 ha^{-1} yr^{-1}; Coverdale *et al.* 2014). In Kenya, where the soil C_{org} of mangrove forests was high but the mangrove trees were killed without disturbing the soils, moderate C_{org} losses and CO_2 emissions (25–36 metric tons ha^{-1} yr^{-1}) were documented (Lang'at *et al.* 2014). Moderate C_{org} losses were also recorded for tidal marshes in the US and mangroves in the Honduras during natural disturbances where the vegetation died but the soil remained intact, although subsided (Cahoon *et al.* 2003; Macreadie *et al.* 2013; Lane *et al.* 2016). For seagrass beds with low-to-moderate stocks of soil C_{org}, low-to-moderate levels of C_{org} loss have been reported with vegetation declines resulting from eutrophication, seismic testing, and damage by boat mooring chains in southern Australia (Macreadie *et al.* 2015; Serrano *et al.* 2016). Losses in soil C_{org} stocks were not detected after small patches of vegetation were cleared in Australian seagrass meadows, which had low levels of soil C_{org} (Macreadie *et al.* 2014). Land reclamation of tidal marshes in China – where soil C_{org} stocks were relatively low and soils were not

Table 1. Risk matrix of CO_2 emissions with varying size of the soil C_{org} stock and relative rate of C_{org} remineralization (derived from WebTable 3)

		Soil carbon stock				
		Low C_{org} stock (< 50 mt ha^{-1})	*Low–moderate C_{org} stock (50–100 mt ha^{-1})*	*Moderate C_{org} stock (100–250 mt ha^{-1})*	*Moderate–high C_{org} stock (250–500 mt ha^{-1})*	*High C_{org} stock (> 500 mt ha^{-1})*
Description of potential for remineralization	***Relative scores***	***1***	***2***	***3***	***4***	***5***
Low	**1**	**1 (Low)**	**2 (Low)**	**3 (Low)**	**4 (Low)**	**5 (Mod)**
Moderate	**2**	**2 (Low)**	**4 (Low)**	**6 (Mod)**	**8 (Mod)**	**10 (Mod-High)**
Moderate–high	**3**	**3 (Low)**	**6 (Mod)**	**9 (Mod)**	**12 (Mod-High)**	**15 (High)**
High	**4**	**4 (Low)**	**8 (Mod)**	**12 (Mod-High)**	**16 (High)**	**20 (Very High)**
Very high	**5**	**5 (Mod)**	**10 (Mod-High)**	**15 (High)**	**20 (Very High)**	**25 (Very High)**

Notes: mt = metric tons. The relative risk of CO_2 emissions varies from low (blue, scores 1–4), moderate (green, 5–9), moderately high (yellow, 10–12), high (orange, 15–16), to very high (red, 20–25). Final scores (from 1, low likelihood to 25, very high likelihood) were obtained by multiplying the scores related to likelihood of remineralization and the magnitude of C_{org} stocks.

disturbed but overlaid with sediment, thereby remaining anoxic – also resulted in relatively small losses of C_{org} (Bu *et al.* 2015). These case studies emphasize the importance of assessing the size of the soil C_{org} stock, the disturbance to the soil, and the specific environmental conditions affecting oxidation regimes after the disturbance. These factors therefore represent the main pillars underpinning estimates of the risk of CO_2 emissions.

Conclusions

Many schemes to reduce CO_2 emissions from land-use change are based on calculating the likelihood of emissions after ecosystem loss or degradation. Yet to date, the risks of CO_2 emissions from the soils of degraded or destroyed blue carbon ecosystems have received little attention; resultant data gaps have contributed to the limited accounting of soil C_{org} within blue carbon projects and to the low level of financing of those projects. The variation in CO_2 emissions from soils could be large, depending on the size of the soil C_{org} stocks and likely rates of C_{org} remineralization. Clear articulation of the risks of CO_2 emissions may help to incorporate soil C_{org} into emerging blue carbon projects as well as to determine priorities for conservation or restoration. Our framework for assessing the risk of CO_2 emissions is qualitative, informed by assumptions based on existing evidence and understanding of the drivers of CO_2 emissions. This approach could be refined and extended to a quantitative framework as additional evidence becomes available. Our framework is based on the size of soil C_{org} stocks, about which global knowledge is increasing rapidly. Combined with assessments of the likelihood of soil C_{org} remineralization, this framework provides a structured pathway that can help establish estimates of CO_2 emissions to support the valuation of soil C_{org} stocks and the implementation of blue carbon projects.

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