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Nonuniform organic carbon stock loss in soils across disturbed blue carbon ecosystems

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Conserving blue carbon ecosystems (BCEs) has gained international attention in climate change mitigation, reflected in United Nations policies and voluntary carbon-offset projects. These efforts assume significant and uniform losses of soil organic carbon (C_{org}) throughout the top meter following disturbances, yet this assumption lacks robust empirical support. Here, we synthesized 239 paired observations of intact and disturbed BCEs globally. Soil C_{org} stock losses in the top meters vary widely: from −68.4% (agricultural conversion, ±13.4%, 95% confidence interval) to +0.8% (harvesting, ±46.2%) in mangroves, −25.9% (climate/hydrological change, ± 30.7%) to +48.6% (grazing, ±78.7%) in saltmarshes, and −34.2% (vegetation cover damage, ±22.4%) to −27.4% (dredging, ±33.6%) in seagrasses. Extensive disturbances deplete C_{org} down to 50–200 cm, while limited disturbances impact only the top 10–30 cm or resulted in negligible losses. This refinement contributes to improved global inventories of greenhouse gas emissions from BCEs, supporting abatement policy settings for nationally determined contributions commitments.

Mangroves, saltmarshes, and seagrasses, the most "actionable" blue carbon ecosystems (BCEs) towards climate mitigation, occupy less than 0.2% of the ocean's surface but store over 30 Pg of organic carbon (C_{org}) in the top meter of soil globally[1,2]. However, when these ecosystems are converted or degraded, an estimated 0.15–1.02 Pg of CO_2 is released into the atmosphere annually[3], accounting for 3–23% of annual emissions from global land use change over the past decade[4]. This release occurs primarily through the decomposition of soil C_{org} stock[2,5,6]. Although rates of BCE area decline are slowing down or even reversing in some regions[7–9], the global annual loss rate remains high at 0.13% for mangroves[10], 0.28% for saltmarshes[11], and 1–2% for seagrasses[12]. As a result, conserving BCEs to avoid greenhouse gas (GHG) emissions has gained global recognition as a cost-effective climate action, supported by major international climate and biodiversity policies such as the Paris Agreement and the Kunming-Montreal Global Biodiversity Framework[13,14].

Financial resources for the conservation of BCEs, aimed at preventing soil C_{org} stock loss and the resulting GHG emissions, can be effectively mobilized through compliance and voluntary carbon markets that recognize and assign carbon credits to blue carbon projects[14–16]. These markets require rigorous accounting and verification of GHG emission reductions to demonstrate "additionality"[14–16]. However, within the science and policy communities, calculations of GHG emission reduction from blue carbon projects often assume that the conversion and degradation of BCEs result in uniform C_{org} stock loss across the entire top meter of soil (e.g., refs. 3,11,17; Supplementary Table 1). This assumption, exemplified by the Intergovernmental Panel on Climate Change (IPCC) Tier 1 standard[18], has been perpetuated across studies estimating of global, national, and regional GHG inventories without thorough examination or verification[19], despite IPCC guidelines advocating for modifying the depth of top meter at higher tiers[18].

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Estimates of GHG emissions from soil C_{org} stock loss following BCEs disturbance have been assumed to range from 22.7% to 100% of the C_{org} stock present in the top meter of soil[3,6,11,17,20,21] (Supplementary Table 1). However, growing empirical evidence suggests that the actual depth and amount of C_{org} loss can vary substantially across ecosystems and disturbance regimes[22,23], indicating substantial uncertainties in GHG emission estimates that rely on untested assumptions. Such problematic GHG emission accounting methodologies risk inflating credit issuances, thereby undermining the true effectiveness of nature-based solutions within nationally determined contributions (NDCs). Indeed, the issuance of carbon credits for emission reductions through terrestrial forest conservation has faced scrutiny due to failure in demonstrating genuine GHG reductions, contributing to skepticism about the effectiveness of voluntary carbon markets[24,25]. Understanding how different ecosystems and disturbance regimes affect soil C_{org} stock loss is therefore critical, especially when estimating GHG emission reductions in carbon market transactions.

The various anthropogenic and climate change disturbances affecting BCEs can be broadly classified into two main categories: (1) those that physically remove or alter both the living biomass and the soil, such as conversion of BCEs for aquaculture or agriculture and dredging[5,26,27]; and (2) those that affect BCE plants without directly disturbing the soils, such as harvesting, degradation, or grazing[5,28,29]. For over a decade, it has been hypothesized that the trajectories of soil C_{org} stock loss in BCEs may vary substantially depending on the types of disturbance[5,30] (Fig. 1). Disturbances that physically disrupt the soil are expected to cause greater C_{org} stock loss, penetrating deeper soil layers, while disturbances with minimal soil impact are likely to result in lower C_{org} stock loss, primarily from the shallower soil layers[5,30]. Soil C_{org} stock loss is also anticipated to last longer for disturbances that penetrate deep soil layers relative to those that minimally affect the soil post-disturbance[5]. A comprehensive examination of these hypotheses is essential to understand how soil C_{org} stock loss varies among BCEs subject to different disturbance regimes, which is crucial for accurately estimating GHG emission reductions in blue carbon projects.

Here, we address these knowledge gaps by examining the loss of soil C_{org} stocks across BCEs subject to different disturbance regimes. Our dataset includes paired reports of soil C_{org} stock from both disturbed and intact BCEs from 118 mangrove sites, 82 saltmarsh sites, and 39 seagrass sites worldwide (Supplementary Fig. 1). We categorized disturbances into those causing deep soil disruption—such as the conversion of mangroves and saltmarshes to agriculture or aquaculture and dredging of seagrass meadows—and those causing minimal soil disruption, including mangrove harvesting, saltmarsh grazing, climate/hydrological change in mangroves and saltmarshes, and vegetation cover damage in seagrass meadows (Fig. 1). We first analyzed how soil C_{org} stock loss varied across BCEs and with depth following disturbance to resolve the distribution of C_{org} stock loss both across ecosystems and along the soil depth profile under various disturbance regimes. We then examined the relationship between soil C_{org} stock loss and initial C_{org} stock, as well as its relationship with the duration of disturbance across different disturbance regimes. Our results provide empirical evidence that both soil depth and the extent of C_{org} stock loss differ among BCEs and are controlled by the type of disturbance regimes. The estimates of C_{org} stock loss reported here

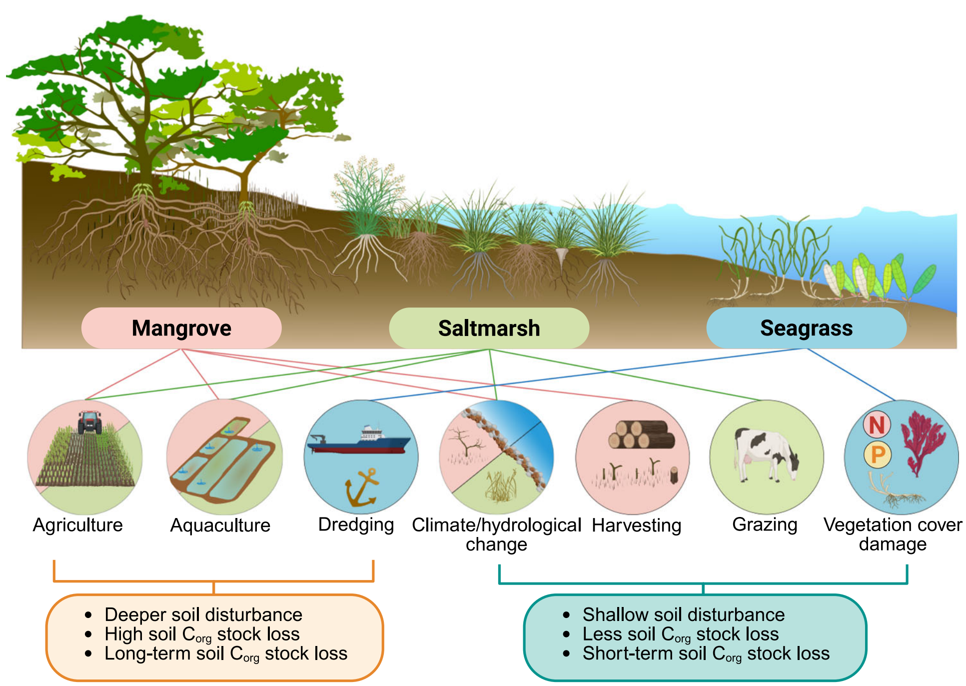

Fig. 1 | Hypothesized impacts of disturbance regimes on soil organic carbon (C_{org}) stocks in blue carbon ecosystems (BCEs). The major disturbance drivers investigated for mangroves include agricultural reclamation for rice fields[36,68–71], pastures[37,72], coconut[40,73,74], rubber[40], or crop plantations[75,76]; aquaculture development for shrimp[34,40,41,72,76–88], or fish[22,88,89], along with salt ponds construction[40] and sand mining[90]; the harvesting[22,28,91–94], clearance[40,95–102], or grazing of trees[103]; and climate impacts such as typhoons[104–108], saltwater intrusion[109], drought[110–113], or strong El Niño-Southern Oscillation[114], as well as hydrological changes like settlement construction[115–129], sediment runoff[130,131], or nutrient effluents from nearby aquaculture ponds[87,132–137]. For saltmarshes, the disturbance drivers include agricultural reclamation for rice[48,49,138–141] or crop fields[49,139,142–147]; aquaculture development for shrimp[41,88,148,149] or fish ponds[41,49,147]; climate impacts such as typhoons[150], as well as hydrological changes like embankment construction[120,148,151–158], or excessive flooding[159,160]; and the grazing[29,161–167], coppicing[168], shading[169], or damage of plants[170]. In this study, aquaculture was combined with agriculture due to the limited and geographically biased data availability for aquaculture, with no significant difference observed between them ($P = 0.217$; see section "Methods"). For seagrasses, the disturbance drivers include unintentional dredging due to mooring[51], boat grounding[26,52], or clam harvesting[171–173], and intentional dredging for sand[174]; and vegetation cover damage due to shading[175–177], removal[178,179], overgrazing[180], loss[23,181–187], or nutrient enrichment[116,188–193]. Created in BioRender. Fu, C. (2025) https://BioRender.com/q55g628.

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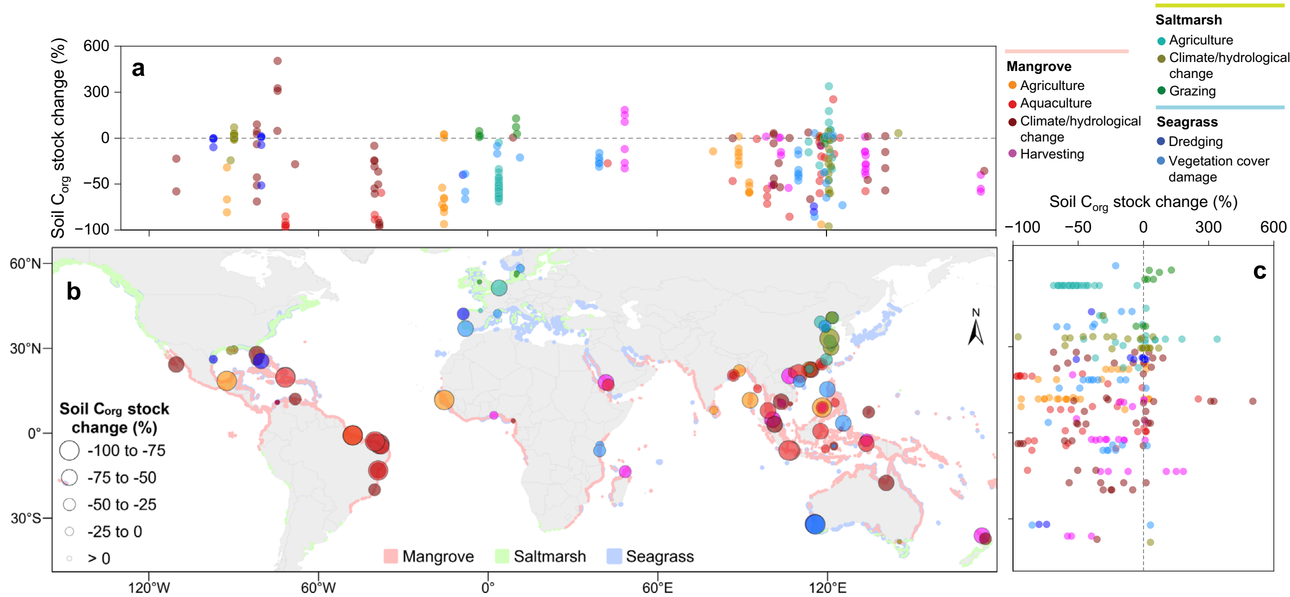

Fig. 2 | Spatial distribution of reported and extrapolated top-meter soil organic carbon (C_{org}) stock changes in blue carbon ecosystems (BCEs) under different disturbance regimes. a The longitudinal distribution of soil C_{org} stock change (%) in BCEs; **b** global distribution of soil C_{org} stock change (%) in BCEs, **c** the latitudinal distribution of soil C_{org} stock change (%) in BCEs. Shapefile of the world obtained from www.naturalearthdata.com. Source data are provided as a Source Data file.

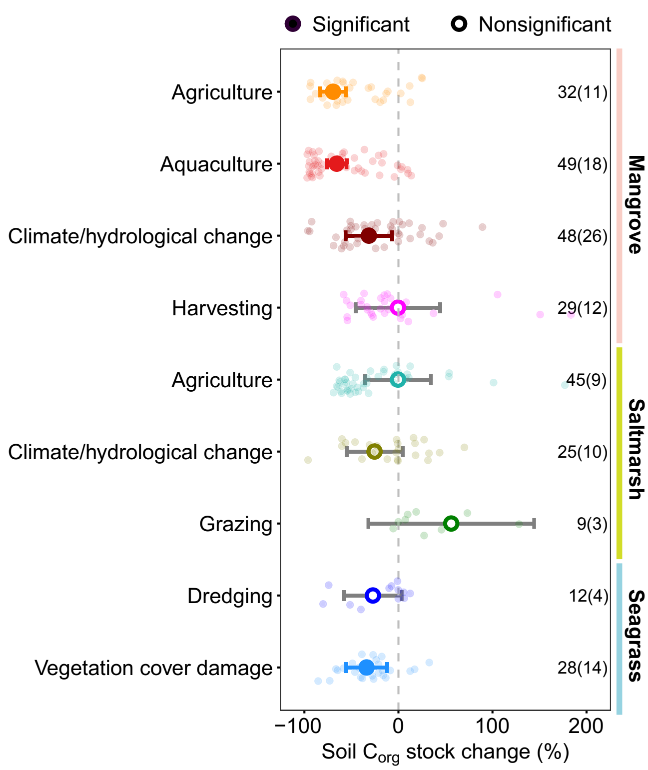

Fig. 3 | Responses of top-meter soil organic carbon (C_{org}) stocks in blue carbon ecosystems (BCEs) to different disturbance regimes. Bars around the means denote 95% CIs. The first and second numbers in parentheses indicate, respectively, the number of observations and the number of studies included in each calculation. Filled circles indicate significant effects of soil C_{org} stock loss, while circles indicate nonsignificant effects (95% CIs overlapping zero). Mangrove soil C_{org} stock changes resulted from agricultural and aquaculture conversions, climate/hydrological change, and harvesting; saltmarsh soil C_{org} stock changes resulting from agricultural conversions, climate/hydrological change, and grazing; seagrass soil C_{org} stock changes resulting from dredging and vegetation cover damage. Source data are provided as a Source Data file.

address a knowledge gap and enable more accurate and robust estimates of GHG reduction in blue carbon projects as a nature-based climate solution.

Results

Top-meter soil C_{org} stock loss under different disturbance regimes

When extrapolating soil C_{org} stock to the top one meter globally, we found that disturbances in BCEs resulted in significantly higher losses of C_{org} stock in mangroves (−54.4 ± 8.7%, mean ± 95% confidence interval [CI]) and seagrasses (−36.4 ± 27.6%) relative to saltmarshes (−4.4 ± 36.3%, $P < 0.001$), though substantial variability was observed across all ecosystems. Soil C_{org} stock losses in mangroves were more pronounced and less variable in areas reported as mangrove loss hotspots, including Southeast Asia, and Central and South America[10] (Fig. 2). In saltmarshes, soil C_{org} stock losses were similarly less variable and more prominent in China relative to Europe and USA, despite historical declines in saltmarsh area across these regions[11,31]. However, no clear geographic pattern of C_{org} stock loss was observed in seagrasses under disturbance. We could not detect an influence of plant species on soil C_{org} stock loss under disturbance across BCEs ($P > 0.05$; Supplementary Table 2), except in mangroves, where species interacted with disturbance regimes (disturbance × species, $P = 0.003$).

Disturbance regimes exerted significant control on soil C_{org} stock loss in mangroves ($P < 0.001$), but their effects were less pronounced in saltmarshes and seagrasses ($P > 0.05$). Major soil C_{org} loss in mangroves were caused by extensive soil disruption, in contrast to the variability in soil C_{org} stock loss under limited soil disturbance. Specifically, the conversion of mangroves to agriculture and aquaculture led to significant reductions in top one-meter soil C_{org} stocks, averaging −68.4 ± 13.4% and −65.4 ± 10.6%, respectively (Fig. 3). It should be noted, however, that the estimated C_{org} stock loss from converting mangroves to aquaculture might be conservative if soil excavation was involved during the initial "construction" phase[18]. In contrast, climate/hydrological change (−32.1 ± 23.0%) and harvesting activities (+0.8 ± 46.2%) in mangroves, which cause limited soil disturbance, result in less or nonsignificant C_{org} stock changes. In saltmarshes, no significant changes in soil C_{org} stock were detected in the top meter, regardless of the disturbance regime (agriculture, −1.1 ± 36.1%; climate/

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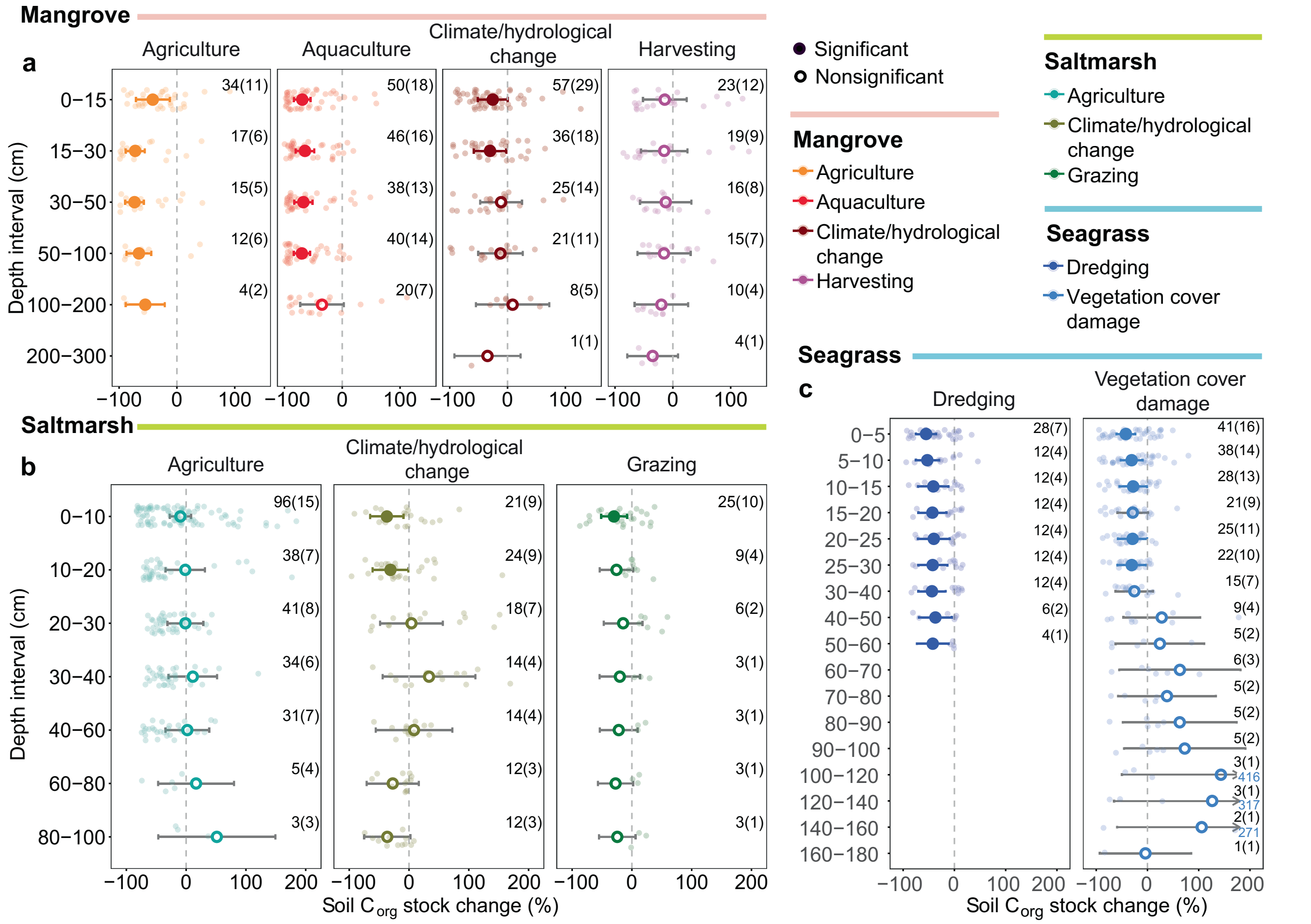

Fig. 4 | Responses of soil organic carbon (C_{org}) stocks to different disturbance regimes across various depth intervals in blue carbon ecosystems (BCEs). **a** Mangrove soil C_{org} stock changes due to agricultural and aquaculture conversions, climate/hydrological change, and harvesting, **b** Saltmarsh soil C_{org} stock changes due to agricultural conversions, climate/hydrological change, and grazing, **c** Seagrass soil C_{org} stock changes due to dredging and vegetation cover damage. Bars around the means denote 95% CIs. The bars with arrows denote 95% CIs that extend beyond the X-axis range. The numbers alongside the arrows indicate the maximum values of the 95% CIs. The first and second numbers in parentheses indicate, respectively, the number of observations and the number of studies included in each calculation. Filled circles indicate significant effects of soil C_{org} stock loss, while circles indicate nonsignificant effects (95% CIs overlapping zero). Source data are provided as a Source Data file.

hydrological change, −25.9 ± 30.7%; grazing, +48.6 ± 78.7%). Similarly, in seagrass ecosystems, soil C_{org} stock losses driven by dredging (−27.4 ± 33.6%) were comparable to those from damage to the vegetation cover of seagrass meadows (−34.2 ± 22.4%).

Depth patterns of soil C_{org} stock loss under different disturbance regimes

The depth patterns of soil C_{org} stock loss varied among different disturbance regimes. Extensive soil disruptions, such as the conversion of mangroves to agriculture or aquaculture and the dredging of seagrass meadows, led to significant C_{org} loss reaching soil depths of 100–200 cm and at least 50 cm, respectively (Fig. 4). Specifically, for mangroves converted to aquaculture, soil was excavated to a mean depth of 66 ± 24 cm (± 95% CI, Supplementary Fig. 2), potentially extending soil C_{org} stock loss to approximately 166 cm. In contrast, saltmarsh conversion to agriculture showed no significant soil C_{org} stock loss across depth increments (Fig. 4b). For disturbance regimes with limited soil impact, soil C_{org} stock loss was either negligible across the depth profiles, as observed in the harvesting of mangroves, or largely confined to top 10–30 cm layers (climate/hydrological change in mangroves and saltmarshes, grazing of saltmarshes, and vegetation cover damage in seagrass meadows). However, deeper soil data (>30 cm) for saltmarshes and seagrasses impacted by grazing and vegetation cover damage, respectively, were underrepresented in the literature, underscoring the need for further research. These divergent depth patterns of soil C_{org} stock loss nevertheless suggest that the current assumption of uniform C_{org} loss across the top meter of soil requires reconsideration.

Relationship between soil C_{org} stock loss and initial C_{org} stock

Soil C_{org} stock losses resulting from the conversion of mangroves to agriculture and aquaculture were strongly dependent on the initial C_{org} stock ($P < 0.001$; Fig. 5 and Supplementary Fig. 3), indicating that mangroves with higher initial C_{org} stocks are more susceptible to greater C_{org} losses following disturbances. This dependence also held for the conversion of saltmarshes to agriculture. While converting saltmarsh to agriculture did not show significant soil C_{org} stock loss in the top meter overall, it did result in significant loss—reaching up to −60%—in saltmarshes with higher initial soil C_{org} stock levels (above ~75 Mg C ha^{-1}, $P < 0.001$). In contrast, the conversion of saltmarshes to agriculture with lower initial C_{org} stocks could enhance, rather than reduce, C_{org} stock levels. Grazing of saltmarshes showed a notable gain of soil C_{org} stock, which positively increased with higher initial C_{org} stocks ($P = 0.028$). In seagrass meadows, both gains and losses of soil C_{org} stock were observed following dredging ($P = 0.143$) or vegetation cover damage ($P = 0.109$), independent of initial C_{org} stock. When

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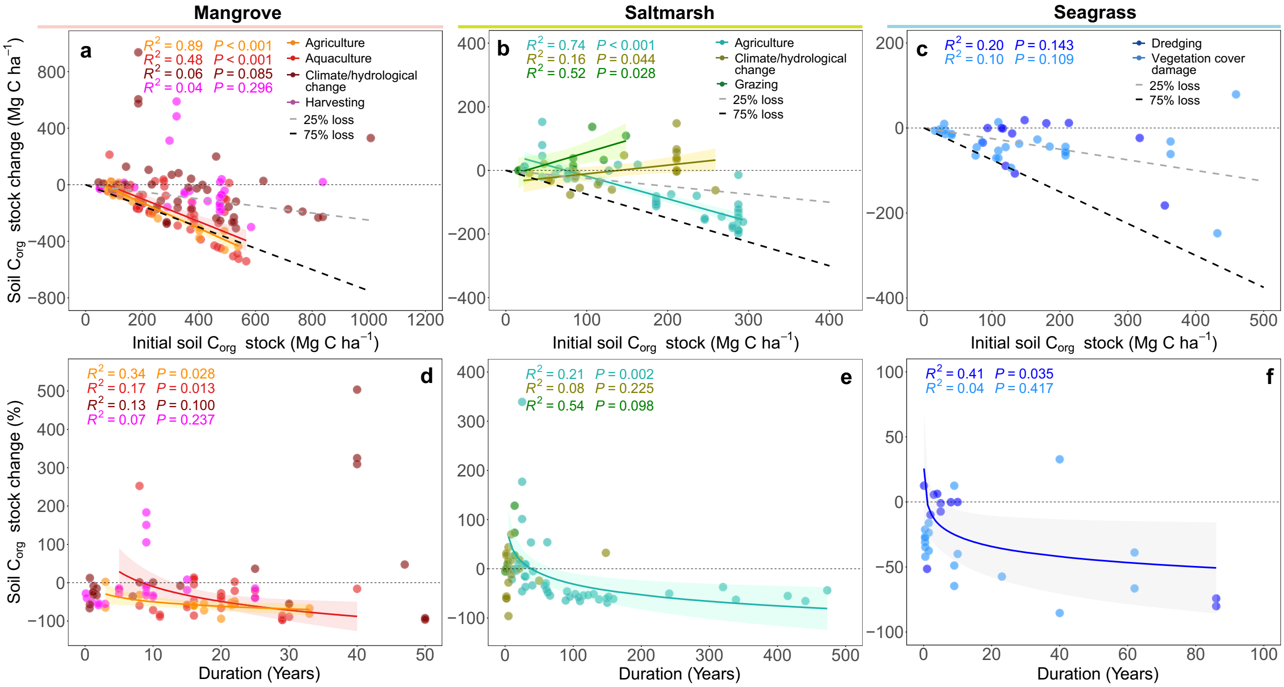

Fig. 5 | Top-meter soil organic carbon (C_{org}) stock change in response to initial C_{org} stock in blue carbon ecosystems (BCEs) and disturbance duration. a–c Top-meter soil C_{org} stock change (Mg C ha^{-1}) in response to initial C_{org} stock, **d–f** Top-meter soil C_{org} stock change (%) in response to disturbance duration. Solid lines represent linear or logarithmic regression, with shading representing the 95% CI of regression fits. The carbon emission factor of 25–75% was taken as a conservative range commonly used in previous studies (Supplementary Table 1). Top-meter soil C_{org} stock change (%) in response to initial C_{org} stock are shown in (Supplementary Fig. 3). Source data are provided as a Source Data file.

compared to the GHG emission factor commonly used in previous studies (taking 25–75% of soil C_{org} stock loss as a conservative range; Supplementary Table 1), we found that as initial C_{org} stocks increased, losses driven by the conversion of mangroves and saltmarshes to agriculture and aquaculture were more aligned with or exceeded this range. However, as initial C_{org} stock increases, soil C_{org} stock losses resulting from seagrass disturbances were not well aligned with the range of the GHG emission factor.

Relationship between soil C_{org} stock loss and disturbance duration

Soil C_{org} stock losses generally increase with disturbance duration for activities that cause extensive soil disruptions, including the conversion of mangroves and saltmarshes to agriculture and aquaculture ($P < 0.05$), and the dredging of seagrass meadows ($P = 0.035$, Fig. 5). Specifically, losses in soil C_{org} stock after converting mangroves to agriculture and aquaculture asymptotically reached −68.2 ± 13.4% and -86.3 ± 14.0%, respectively, 30 years post-conversion. Reclamation of saltmarshes to agriculture resulted in an average soil C_{org} stock reduction of −49.9 ± 9.2% approximately 60 years post-conversion, despite showing both positive (net gain) and negative (net loss) changes at shorter intervals. Available reports indicate that dredging of seagrass meadows leads to a soil C_{org} stock loss of −77.1 ± 8.4% after 80 years since the disturbance. Overall, these patterns are consistent with estimates based on previous modeling, which calculated that cumulative GHG emissions from disturbed BCEs account for 70–80% of the initial top-meter soil C_{org} stocks over 40 years[32]. In contrast, for disturbances with minimal soil impact, the response of soil C_{org} stock loss to disturbance duration lacks a clear pattern.

Discussion

Within the latest NDCs under the Paris Agreement, 41 countries have prioritized ocean or BCEs as key sector for reducing GHG emissions[33]. However, the lack of empirical support for the long-held assumption that disturbance of BCEs leads to uniform loss of C_{org} stock across the top meter of the soil—ranging from 22.7% to 100%—poses a challenge in effectively informing and operationalizing NDCs[34,35]. Our synthesis and quantitative analysis of available evidence demonstrates that the widely used assumption that the top meter of soils in BCEs is uniformly susceptible to loss, regardless of disturbance regimes (e.g., ref. 3), should be rejected. The magnitude and dynamics of soil C_{org} loss in disturbed BCEs are highly dependent on ecosystem type, disturbance regimes, baseline soil C_{org} stock, and the time since disturbance. Furthermore, our estimations provide GHG emission factors for anthropogenic and natural disturbance drivers that were not previously considered by the IPCC (Table 1), such as climate/hydrological change in mangroves and saltmarshes, which increasingly lead to BCE losses[10–14]. This work marks a critical advancement in refining global and national inventories of GHG emissions from BCEs using the IPCC Tier 1 standards, providing robust support for GHG emissions accounting and abatement policy settings for future blue carbon projects and NDCs commitments.

Among the various disturbance drivers examined here, none strictly conformed to the assumption of significant soil C_{org} stock loss to the depth of one meter. For example, the conversion of mangroves to agriculture resulted in soil C_{org} loss up to 200 cm. This disturbance regime involves substantial hydrological manipulation and soil remobilization through drainage and dredging, which reverse hypoxic soil conditions and increase the exposure of C_{org} to aerobic decomposition, thereby extending C_{org} stock losses to deeper layers[5,36–38]. For instance, the drainage of mangrove forests in East Trinity Inlet, Australia, for agricultural production between 1976 and 2000 resulted in a soil elevation loss of 1.3 m—from 0.9 m above sea level to 0.4 meters below—driven by substantial soil C_{org} remineralization[5]. Thus, considering soil C_{org} stock loss only from the top meter may underestimate GHG emissions due to such disturbance. Furthermore, our results indicate that soil C_{org} stock loss from agricultural conversion persists over decades, albeit at a decreasing rate. Therefore, removing

Table 1 | Changes in soil organic carbon (C_{org}) stock of blue carbon ecosystems (BCEs) due to various disturbances and their comparison with Intergovernmental Panel on Climate Change (IPCC) guidelines

BCE	Disturbance regime	Soil depth with significant C_{org} stock loss	Emission factor for soil layers with significant C_{org} losses (%)	Emission factor for top one meter (%)	Dependence on initial soil C_{org} stock (initial C_{org} stock (Mg C ha^{-1}) = x; emission factor for top one meter (%) = y)	Corresponding management in the IPCC guidelines[18]	IPCC Tier 1 emission factor[18]
Mangrove	Agriculture	Top 200 cm	−71.8 ± 19.8	−68.4 ± 13.4	y = 125.37 − 34.35* ln(x)	Drainage	7.9 C ha^{-1} yr^{-1}
	Aquaculture	Construction: top 66 ± 24 cm	NA	NA	NA	Extraction (Construction)	96%
		Use and discontinued: top 100 cm	−65.4 ± 10.6	−65.4 ± 10.6	y = 163.14 − 38.69* ln(x)	Extraction (Use and discontinued)	Not reported
	Climate/hydrological change	Top 30 cm	−31.3 ± 18.7	−32.1 ± 23.0	Nonsignificant	Not reported	Not reported
	Harvesting	Nonsignificant	NA	+0.8 ± 46.2	Nonsignificant	Forest management	0
Saltmarsh	Agriculture	Nonsignificant	NA	−1.1 ± 36.1	y = 237.57 − 52.09* ln(x)	Drainage	7.9 C ha^{-1} yr^{-1}
	Climate/hydrological change	Top 20 cm	−33.5 ± 31.9	−25.9 ± 30.7	Nonsignificant	Not reported	Not reported
	Grazing	Top 10 cm	−30.4 ± 20.5	+48.6 ± 78.7	Nonsignificant	Not reported	Not reported
Seagrass	Dredging	≥top 50 cm	−50.4 ± 29.1 (Top 50 cm)	−27.4 ± 33.6	Nonsignificant	Extraction	96%
	Vegetation cover damage	Top 30 cm	−32.7 ± 20.8	−34.2 ± 22.4	Nonsignificant	Not reported	Not reported

NA not available.

seawalls or dikes to reintroduce tidal flow—thereby inhibiting aerobic C_{org} decomposition—is key to prevent further C_{org} stock loss and enable cost-effective mangrove recovery[14,39].

In contrast to agricultural conversions, converting mangroves to aquaculture typically involves more intensive soil excavation during pond construction, reaching down to depths of 50–250 cm, which substantially enhances C_{org} destabilization and loss under aerobic conditions[5,38,40,41]. We estimated that aquaculture conversions roughly excavated soil to depths, averaging (± 95% CI) 66 ± 24 cm, with additional soil C_{org} stock loss extending to 100 cm in post-excavated soil, culminating in soil C_{org} stock loss down to approximately 166 cm. This indicates that considering only C_{org} stock loss in the top meter of soils during the initial "construction", without accounting for losses during the "use" and "discontinued" phases[18], may substantially underestimate GHG emissions. Indeed, our results suggest that soil C_{org} stock during these subsequent phases may be completely depleted over 30 years. Overlooking soil C_{org} stock loss in the "use" and "discontinued" phases could result in the allocation of insufficient carbon credits to the project, thereby limiting the scale and ambition of blue carbon projects, as well as diminishing interest in funding these initiatives. Furthermore, our findings indicate that three out of 16 converted aquaculture ponds in our dataset of elevation changes exhibited gains in soil mass (Supplementary Fig. 2). This counterintuitive result may be attributed to the collapse of soil structures due to belowground C_{org} decomposition, combined with bottom leveling during pond construction and soil erosion from berms, or regional variance in soil depth across mangroves and aquaculture ponds[42]. Therefore, clarifying soil depth dynamics is essential to robustly estimate GHG emissions following mangrove conversion to aquaculture[43].

Our findings further demonstrate that soil C_{org} stock losses due to agriculture and aquaculture conversions in mangroves are positively correlated with initial C_{org} stock. This likely reflects the role of hydrogeomorphic processes in shaping C_{org} levels in soils of BCEs, along with the mineral-dominant nature of soils with low C_{org} stock, which attenuates C_{org} remineralization after disturbance through mineral protection[44]. In contrast, soils with higher C_{org} stocks contain greater proportions of particulate organic matter with weaker mineral associations, making them more susceptible to C_{org} remineralization and increased GHG emissions following disturbance[44,45]. As such, conserving mangroves with high soil C_{org} stock against conversion to agriculture and aquaculture yields the greatest reduction in C_{org} loss and, therefore, potential GHG emissions. The variance in initial soil C_{org} stock is influenced by environmental factors, particularly hydrogeomorphic settings[46]. Estuarine interior mangroves generally have higher soil C_{org} stock than open-coast or fringe mangroves[22] but are highly vulnerable to conversion for agriculture and aquaculture due to their fertile soils and flat landscapes, resulting in deep soil C_{org} stock loss[47]. Therefore, incorporating geomorphic context, either alongside or as a proxy for initial soil C_{org} stock can contribute to the development of robust conservation projects, enabling better prioritization of the most vulnerable areas and maximizing GHG emission reductions.

The context-dependent effect of initial C_{org} stock on soil C_{org} stock loss is particularly pronounced in conversion of saltmarshes to agriculture. The assumption of significant soil C_{org} stock loss in the top one meter was only met in saltmarshes with initially higher soil C_{org} stocks. In contrast, reclamation of saltmarshes with lower initial soil C_{org} stock to agriculture leads to the net gain in soil C_{org} stock, likely due to the application of organic or chemical fertilizers that promote the accumulation of plant-derived organic material in soils[48,49]. This tendency is particularly conspicuous in China, where the newly formed saltmarshes, created after long-term sequential reclamation, have low soil C_{org} stock[31,50]. Consequently, the reclamation of these saltmarshes has enhanced their soil C_{org} stock after converting them to paddy or crop lands[31,49]. Similarly, aquaculture production in converted saltmarshes with low initial soil C_{org} stock may introduce substantial organic inputs, such as excessive fodder and feces from cultured fish or shrimp, contributing to soil C_{org} accumulation. Therefore, conservation projects for saltmarshes that do not account for background soil C_{org} stocks may not achieve the anticipated avoidance of GHG emissions. Our results further show that soil C_{org} stock losses following conversion of saltmarshes to agriculture are sustained over centuries. Hence, rapid deployment of restoration efforts where conversion of saltmarshes with high soil C_{org} stock to agriculture already occurred is critical for maximizing GHG emission reduction, while restoring C_{org} sequestration.

Soil C_{org} stock loss due to dredging of seagrass meadows can penetrate to depths of at least 50 cm, aligning with observations that dredging—whether unintentional or intentional—disturbs soil to depths ranging from 10 to 160 cm[26,51,52]. However, significant C_{org} stock loss was not observed when extrapolating to the top meter of soil, owing to the variable response of soil C_{org} stock following dredging. This variability may be attributed to constant seawater inundation of disturbed seagrass soil, which limits oxygen penetration and C_{org} decomposition[1,5,32]. Nutrient release from soil disturbance can also stimulate phytoplankton and bacterial blooms[53,54], which may either lead to the accumulation of labile C_{org} or enhanced C_{org} decomposition through the priming effect[55,56]. Further research is needed to delineate the impact of soil C_{org} stock loss due to dredging, with a particular focus on sampling deeper soil layers in seagrass meadows.

In contrast to disturbance regimes that cause extensive soil disruption, our results show that climate/hydrological changes in mangroves and saltmarshes, harvesting of mangroves, grazing of saltmarshes, and vegetation cover damage in seagrass meadows result in C_{org} stock loss primarily within the top 10–30 cm, or lead to minimal C_{org} stock losses or even net gains due to limited soil disruption. Climate/hydrological changes in mangroves and saltmarshes and vegetation cover damage in seagrass meadows lead to the cessation of new organic material accumulation, while the root decay may trigger a priming effect, enhancing the decomposition of surface soil C_{org} and/or soil erosion[55,57]. Harvesting of mangroves and grazing on saltmarshes substantially reduce aboveground biomass and disturb the surface soil via trampling[28,29]. However, surface soil C_{org} stock losses may be partially offset by increased belowground biomass production[29] and ecosystem recovery post-disturbance, which repromotes C_{org} accumulation[22,28]. Trampling may also compact surface soil, enhancing the development of anaerobic conditions that weaken C_{org} mineralization[58]. Consequently, conservation efforts targeting these disturbances may not achieve the anticipated avoidance of GHG emissions. These findings underscore that overestimated emission factors, based on unverified assumptions perpetuated across studies[3,11,17,20,21], may have led to the issuance of excessive carbon credits to blue carbon projects achieving additionality through avoidance of these impacts. Failure to achieve the committed GHG emission avoidance undermines the credibility of nature-based solutions, including blue carbon projects, despite their important role in contributing to climate change mitigation efforts.

While the avoided GHG emissions from conservation projects focusing on BCEs affected by disturbances causing limited soil disruption are modest, these efforts can still be motivated by the broader benefits of the ecosystem services they provide, such as biodiversity enhancement and coastal protection[14–16]. Management regimes that include sustainably managed harvesting practices, such as logging of mangroves and grazing of saltmarshes, can minimize GHG emissions while supporting provision services that benefit communities[14,29]. Incentivizing such sustainable management practices can achieve modest GHG avoidance while supporting co-benefits, representing a new dimension of blue carbon projects beyond those focused on conservation and/or restoration that deserves consideration.

While our global synthesis provides a compelling analysis of soil C_{org} dynamics across different disturbance regimes, it is restricted by data scarcity for some regions, such as South America and Africa, particularly for saltmarshes and seagrasses. To further reduce uncertainty, there is a need to expand empirical evidence on C_{org} dynamics over long time scales and across a broader range of disturbance regimes. Nevertheless, the disturbance regimes explored in this study can serve as approximations for unstudied disturbance drivers, based on their potential effects on ecosystems and their convergence with studied drivers. For example, disturbances leading to plant die-off and loss due to insects or diseases[59–61] could be approximated using climate/hydrological change category for mangroves and saltmarshes, as well as the vegetation cover damage category for seagrasses, as described in this study. Notably, due to the lack of paired data reporting C_{org} dynamics, we could not explore coastal erosion as a disturbance driver increasingly causing BCE loss, such as reported in Bangladesh, Brazil, USA and Russia[10,11]. Moreover, losses of soil C_{org} are not necessarily equivalent to GHG emissions but provide a useful upper ceiling to resulting GHG emissions. Available evidence suggests that some of the eroded C_{org} from saltmarshes and terrestrial sources delivered to estuaries may be reburied elsewhere in the coastal ocean[62,63]. Therefore, further assessment of the fate of eroded C_{org} will be key to constrain the GHG emission driven by direct or indirect erosion.

The results presented provide robust, empirically validated estimates across BCEs, allowing for the development of improved emission factors and the assessment of potential GHG emission reductions from actions that protect BCEs at risk from various disturbances. Major soil C_{org} stock losses due to the conversion of mangroves and saltmarshes to agriculture or aquaculture, and the dredging of seagrass meadows, highlight the urgent need for preventing such disturbances, particularly in regions with higher C_{org} stocks, which can be achieved through including habitats highly vulnerable to such disturbances into marine protected areas[14]. Given the enduring nature of C_{org} loss after disturbances, restoration efforts underpinned by major global instruments should prioritize projects in converted areas, improving land tenure and providing economic alternatives for affected communities to prevent further GHG emissions and restore C_{org} sequestration[14,16,64]. Climate/hydrological change results in relatively lower soil C_{org} stock loss due to lesser soil disturbance, imparting optimism that restoring degraded BCEs can recover C_{org} stock while preventing further losses. Furthermore, sustainable, non-destructive use of BCEs, such as selective harvesting of mangroves, can sustain human benefits without impairing soil C_{org} storage. These findings contribute to strengthening blue carbon projects as a robust and effective nature-based solutions, supporting the inclusion of BCEs as an important asset in the NDCs' more ambitious commitments to address the growing magnitude and pace of climate change.

Methods

Literature search and screening

We systematically searched all peer-reviewed literature using the Web of Science (http://apps.webofknowledge.com) with the keywords listed in Supplementary Table 3 published before 20 November 2023, following the PRISMA protocol[65], to systemically investigate the effects of disturbance on soil C_{org} stock loss in BCEs. A follow-up search was conducted on 25 December 2024 using the same keywords to include recent studies published after 20 November 2023. To prevent bias in selection from the 4886 potentially relevant articles obtained from the two systematical searches, we first removed a significant number of articles through title screening, leaving 1943 articles for further inspection (Supplementary Fig. 4). The title, abstract, and full text were screened based on the following criteria: first, they must report the anthropogenic or natural driver of soil C_{org} stock loss in BCEs, such as, agricultural reclamation, aquaculture development, dredging, clearance, and climate/hydrological change; second, they must contain reference sites for the disturbed sites, such as an undisturbed, natural, or controlled plots without apparent human and/or climate change impacts, sharing the same parent material and environmental conditions, including at least one undisturbed site if multiple disturbed sites were present; third, they must report at least the soil C_{org} (or soil organic matter) content for the paired sites; and fourth, they must report the sampled soil depth to determine soil C_{org} content. If an article included multiple sites or different habitat types, each was treated as a separate entry in the database. We excluded studies that did not specify a clear disturbance driver, reported multiple disturbance drivers affecting the same site, or investigated soil C_{org} stock losses based on area changes of BCEs, modeling with scaled-up data (i.e., GIS and remote sensing modelling), or laboratory manipulation experiments. We also excluded studies that investigated soil C_{org} stock changes due to ecological invasion (e.g., the *Spartina alterniflora* invasion in Chinese BCEs[66]), encroachment between BCEs (e.g., mangroves encroached into saltmarshes near the poleward limits[67]), given that the loss of the original habitats is compensated by the gain of another BCE.

All data for the analysis were derived from paired intact and disturbed BCEs, from 140 research articles encompassing 118 mangrove sites, 82 saltmarsh sites, and 39 seagrass sites worldwide (Supplementary Fig. 1 and Supplementary Fig. 4). The major disturbance drivers investigated in these studies were categorized based on two key principles: (1) the purpose of the disturbance to BCEs and the extent of soil disruption, and (2) the availability of sufficient reports to perform statistical analysis. When the number of studies was insufficient, disturbance drivers were combined according to the first principle. For mangroves, the disturbance drivers include: (1) agricultural reclamation for rice fields[36,68–71], pastures[37,72], coconut[40,73,74], rubber[40], or crop plantations[75,76]; (2) aquaculture development for shrimp[34,40,41,72,76–88], or fish[22,88,89], along with salt pond construction[40] and sand mining[90]; (3) harvesting[22,28,91–94], clearance[40,95–102], or grazing of trees[103]; and (4) climate impacts such as typhoons[104–108], saltwater intrusion[109], drought[110–113], or strong El Niño-Southern Oscillation[114], as well as anthropogenic activities like settlement construction[115–129], sediment runoff[130,131], or nutrient effluents from aquaculture ponds[87,132–137]. For saltmarshes, the disturbance drivers included: (1) agricultural reclamation for rice[48,49,138–141] or crop fields[49,139,142–147]; (2) aquaculture development for shrimp[41,88,148,149] or fish ponds[41,49,147]; (3) climate impacts such as typhoons[150], as well as hydrological changes like embankment construction[120,148,151–158] or excessive flooding[159,160]; and (4) the grazing[29,161–167], coppicing[168], shading[169], or damage of plants[170]. We combined (1) and (2) due to only one study reporting soil C_{org} dynamics to one meter soil depth due to fish aquaculture conversion[148]. Furthermore, all reports on converting saltmarshes to aquaculture come exclusively from China[41,49,88,147–149], and no significant difference was observed between (1) and (2) ($P = 0.217$). We have categorized the combined disturbance driver as agriculture as the data is dominating by agricultural conversions. For seagrasses, the disturbance drivers included: (1) unintentional dredging due to mooring[51], boat grounding[26,52], or clam harvesting[171–173], and intentional dredging for sand[174]; and (2) vegetation cover damage due to shading[175–177], removal[178,179], overgrazing[180], loss[23,181–187], or nutrient enrichment[116,188–193].

These disturbance drivers cover all common disturbances occurring in BCEs[10,12,14,194] and were categorized based on their potential impact on soils, following the framework proposed by Lovelock et al.[5]. Consequently, the drivers for mangrove disturbance were divided into four categories: agriculture, aquaculture (extensive soil disturbance), climate/hydrological change, and harvesting (limited soil disturbance). For saltmarsh disturbance, the drivers were divided into three categories: agriculture (extensive soil disturbance), climate/hydrological change, and grazing (limited soil disturbance). For seagrass disturbance, the drivers were divided into two categories: dredging (extensive soil disturbance), and vegetation cover damage (limited soil disturbance). All the studies were conducted in paired sites using the "space for time" approach[27]. Given that soil C_{org} may take several years or decades to reach a new equilibrium[32,195], there were limited studies with time series data extending back to pre-disturbance conditions. Consequently, for each paired site, it was assumed that soil conditions were similar before the disturbance. For chronosequences, data from all time points were utilized in this study.

Data extraction and standardization

Data on latitude and longitude, mean annual temperature (MAT), mean annual precipitation (MAP), sampling depth (and intervals), sampling number, soil C_{org} (or soil organic matter) content, dry bulk density (DBD), and duration of disturbance were extracted from the eligible literature. For those data not provided numerically but graphed, we determined values from figures with Web Plot Digitizer (https://automeris.io/). For studies reported soil organic matter, we used the conversion factor reported for mangroves (soil $C_{org} = 0.21 \times$ soil organic matter ^ 1.12, ref. 196), saltmarshes (soil $C_{org} = 0.52 \times$ soil organic matter −0.21, ref. 196), and seagrasses (soil $C_{org} = 0.40 \times$ soil organic matter −1.17, ref. 197) to recalculate soil C_{org} content to facilitate the calculation of soil C_{org} stock. For studies providing only age intervals (e.g., 10–25 years or >66 years), we averaged the range (e.g., 17.5 years) or use the lower range if the upper range was not reported (e.g., 66 years). If a publication presented data for multiple control sites, we combined those control sites and recalculated their mean, standard deviation and sample size.

Despite its importance for determination of soil C_{org} stock, DBD was reported in only 55% of the included studies (34/77 for mangroves, 19/39 for saltmarshes, and 10/28 for seagrasses). We therefore used a random forest algorithm to estimate the missing DBD for each soil depth which reported soil C_{org} data, using all available predictor variables, including soil C_{org} content, BCE species (only for intact sites), middle depth of each sampling depth interval, MAT, MAP, disturbance driver (only for disturbed sites) and disturbance duration (only for disturbed sites), in a manner analogous to a pedotransfer function[198] (Supplementary Fig. 5). For observations where MAT and MAP were not reported, we filled in the missing data using the WorldClim 2.0 dataset (spatial resolution: 30 s, https://www.worldclim.org/data/worldclim21.html) by averaging values within a 1 km buffer of each site's longitude and latitude using ArcGIS 10.8 (ESRI, Redlands, CA). Random forest model combines a large number of regression trees, trained using bootstrap aggregation, to build a robust predictive model resistant to noise in the data[199]. The DBDs predicted using the random forest algorithm show significant linear agreement with the observed DBDs for both intact and disturbed BCEs, with an overall R^2 of 0.89 (range: 0.73–0.99; $P < 0.001$) and a root mean square error of 0.11 ± 0.02 g cm^{-3} (mean ± standard error; range: 0.05–0.16 g cm^{-3}; Supplementary Fig. 6), supporting the validity of this imputation method.

Standardization of soil depth was necessary due to the depth pattern of soil C_{org} stock loss being a core focus of this study, but there was considerable variation in soil depth and intervals reported, with depth classes ranging from 5 to 300 cm across eligible studies. To ensure comparability, we identified the most common sample depths for each ecosystem: 0–15, 15–30, 30–50, 50–100, 100–200, and 200–300 cm for mangroves; 0–10, 10–20, 20–30, 30–40, 40–60, 60–80, and 80–100 cm for saltmarshes; and 0–5, 5–10, 10–15, 15–20, 20–25, 25–30, 30–40, 40–50, 50–60, 60–70, 70–80, 80–90, 90–100, 100–120, 120–140, 140–160, and 160–180 cm for seagrasses. Data from the included research articles that reported sample depths not aligning with these distinct ranges were standardized using two methods[198,200], depending on the maximum depth intervals reported. First, if at least five depth intervals were reported in the study, we fitted the soil C_{org} content and DBD to soil depth for each profile using various regression models, including linear regression, second- and third-order polynomial regressions, exponential function, loess regression, and spline regression[200]. The midpoint of each soil layer (e.g., 7.5 cm for the 0–15 cm soil layer of mangroves) was used as the representative depth. The best-fit model, determined by the smallest residual standard error, was then used to predict soil C_{org} content and DBD within the sampled profile[200]. The average residual standard error of the best-fit model for DBD in intact and disturbed BCEs are 0.05 ± 0.004 (range: 0.002–0.16) g cm^{-3} and 0.05 ± 0.004 (range:

0.002–0.22) g cm^{-3}, respectively. Meanwhile, the average residual standard error of the best-fit model for soil C_{org} in intact and disturbed BCEs are 0.58 ± 0.08 (range: 0.01–3.02) % and 0.48 ± 0.07 (range: 0.002–4.41) %, respectively (Supplementary Fig. 7).

It is worth noting that for mangrove soils, studies often report soil C_{org} content and DBD for the entire 100–300 cm depth range based on a published protocol[201]. However, the actual sampling depth is typically just 5 cm in the middle of this range, specifically from 197.5 cm to 202.5 cm. To avoid introducing uncertainty in assessing C_{org} stock changes for the 200–300 cm depth range, we regarded this data as representing the 100–200 cm depth. Therefore, only data specifically sampled from the 200–300 cm range were included for this depth category.

Second, if fewer than five depth intervals were reported, we assigned soil C_{org} content and DBD to specific depth categories by first finding the median of the reported depth increment and then determining the appropriate depth category as best as possible[198]. For studies reporting multiple soil C_{org} content and DBD measurements within one depth category, a single value was calculated using a weighted average[198]. The weighted average for soil C_{org} ($C_{org,weighted}$, g C kg^{-1}) was calculated by using the normalized depth intervals as the weighting factor for C_{org} of each layer ($C_{org,i}$ and $C_{org,i+1}$). The depth intervals were normalized by dividing each depth interval (T_i and T_{i+1}) by the sum of the depth intervals, ensuring that the sum of the weighting factors equaled one (formula 1). The same approach was applied to DBD (g cm^{-3}).

$$C_{org,weighted} = C_{org,i} \times \frac{T_i}{T_i + T_{i+1}} + C_{org,i+1} \times \frac{T_{i+1}}{T_i + T_{i+1}} \tag{1}$$

Soil C_{org} stock calculation and extrapolation

Converting mangroves to aquaculture ponds typically involves soil excavation, but the actual depth of excavation is seldomly reported in literature[43]. To estimate the potential soil excavation depth, we filtered soil depth data from our database for paired intact mangroves and aquaculture ponds reported to have been sampled to the bedrock or parent material. Studies that did not sample to these depths were excluded from this analysis due to inability to accurately capture soil depth changes. We first calculated the soil mass to the bedrock or parent material of intact mangroves (M_{intact}, Mg ha^{-1}) and aquaculture ponds ($M_{aquaculture}$, Mg ha^{-1}) by multiplying DBD, the thickness of the corresponding soil layer (T_i, m), and a unit conversion factor (10^4 m^2 ha^{-1}) (formula 2). Then, we calculated changes in soil mass and estimated the potential excavation depth ($T_{excavation}$, m) by dividing the soil mass loss by the average DBD of the intact mangroves (formula 3).

$$M_{intact}\,(\text{or } M_{aquaculture}) = \sum_{i=1}^{n} DBD_i \times T_i \times 10^4 \tag{2}$$

$$T_{Excavation} = \frac{M_{intact} - M_{aquaculture}}{DBD_{intact}} \tag{3}$$

We caution against directly comparing the soil C_{org} stock at the original depth for intact mangroves with that at excavated soil depths in aquaculture ponds, as this may introduce uncertainty in estimating soil C_{org} stock loss. However, our investigation into the depth pattern of soil C_{org} density in intact mangroves using a linear mixed model revealed that soil C_{org} density did not vary significantly across different soil depths (Supplementary Fig. 8; $P = 0.104$). This supports the direct comparison of soil C_{org} stock between intact mangroves and aquaculture across soil profiles without correction.

We recalculated soil C_{org} stocks in disturbed BCEs using a soil mass equivalent approach rather than fixed depth and general soil C_{org} stock differences[22,27,39]. This soil mass equivalent approach compares soil C_{org} stock between intact and disturbed sites on an equivalent soil mass basis for each soil layer, reducing uncertainties associated with DBD variations. This method can address C_{org} density changes caused by processes such as soil compaction, subsidence, drainage, or flooding, which are often significant during or following disturbances[22,27,37,202]. Briefly, we first calculated the dry soil mass (M_i, Mg ha^{-1}) of each layer by multiplying DBD, the thickness of the corresponding soil layer (T_i, m), and a unit conversion factor (10^4 m^2 ha^{-1}) (formula 4). Then, we calculated the areal C_{org} stock ($C_{i,fix}$, kg C ha^{-1}) of each layer to a fixed depth by multiplying C_{org} content ($C_{org,i}$, g C kg^{-1}) and M_i (formula 5). Next, we calculated the soil mass difference between the intact and disturbance sites for each soil layer by subtracting M_i from the selected minimum soil mass ($M_{i,equiv}$, Mg ha^{-1}) (formula 6). Finally, the equivalent C_{org} mass (kg C ha^{-1}) in a soil layer was calculated by subtracting the added C_{org} mass from top soil layer and adding C_{org} mass from the bottom soil layer (formula 7).

$$M_i = DBD_i \times T_i \times 10^4 \tag{4}$$

$$C_{i,fixed} = C_{org,i} \times M_i \tag{5}$$

$$M_{i,add} = M_{i,equiv} - M_i \tag{6}$$

$$C_{i,equiv} = C_{i,fixed} - C_{org,top} \times M_{i-1,add} + C_{org,bottom} \times \left(M_{i,add} - M_{i-1,add}\right) \tag{7}$$

We extrapolated soil C_{org} stock to the top one meter to allow the comparison with the current paradigm and literature values. For studies that reported soil C_{org} stock to depth less than 100 cm depth, we assumed that the C_{org} density of the unmeasured layer was the same as that of the deepest measured layer[39]. For example, if the deepest measured soil layer was 30–50 cm, the C_{org} density in the 50–100 cm layer was assumed to be the same as that in 30–50 cm. This approach, however, might overestimate soil C_{org} density in the deeper layers given that top soil section contains typically more C_{org} than the older bottom section[31,197]. Nonetheless, our compiled soil C_{org} content and DBD data showed limited depth trend from 30 cm to 100 cm (Supplementary Fig. 9), supporting the plausibility of this approach. Moreover, to further reduce uncertainty, we did not include studies that only measured surface soil C_{org} stock (<30 cm) when calculating top-meter soil C_{org} stock. In addition, if studies reported total soil C_{org} stock to a depth less than 100 cm, we projected C_{org} stock in the top meter by multiplying the C_{org} stock by the reciprocal of the ratio to 100, to extrapolate the C_{org} stocks to 100 cm depth[54]. To estimate emission factors for soil layers associated with significant C_{org} stock losses, we used only the measured or depth-standardized values without extrapolation to minimize uncertainty.

Data synthesis of disturbance on soil C_{org} stock

The natural log-transformed response ratio (*lnRR*) was employed to quantify the effect sizes of disturbance on soil C_{org} stock across different soil depths (formula 8; ref. 203):

$$lnRR = Ln\left(\frac{C_{disturbed}}{C_{intact}}\right) \tag{8}$$

where $C_{disturbed}$ and C_{intact} are the soil C_{org} stock in the disturbed and intact BCEs, respectively.

However, in our dataset, standard deviation or the standard error of soil C_{org} stock was not reported in 96 of the 140 articles, preventing us from estimating the variance of the effect size based on the standard deviations, sample numbers and mean values of the disturbed and intact groups. Therefore, we employed the number of replications for weighting (formula 9; refs. 204–206):

$$W_r = \frac{N_{disturbed} \times N_{intact}}{N_{disturbed} + N_{intact}} \quad (9)$$

where W_r is the weight for each observation, $N_{\text{disturbed}}$ and N_{intact} are the numbers of replicates in the disturbed and corresponding intact BCEs, respectively.

To examine the influence of ecosystem or disturbance regimes on top-meter soil C_{org} stock, a linear mixed model was employed with ecosystem or disturbance denoted as fixed factors, while study as a random factor to account for the autocorrelation among observations within each study (formula 10). These analyses were conducted using restricted maximum likelihood estimation (REML) with the lme4 (version 1.1.29) and emmeans (version 1.7.5) packages with W_r as the weight for each corresponding observation in R (version 4.0.4, http://www.r-project.org/).

$$lnRR = \beta_1 \times (Disturbance - 1) + \pi_{study} + \varepsilon \quad (10)$$

where β_1 is the coefficient to be estimated; π_{study} is the random effect factor of study; ε is sampling error.

Similarly, to examine the influence of disturbance regimes on soil C_{org} stock across the depth profile, we employed depth interval denoted as fixed factors for each disturbance driver, while study was treated as a random factor as before (formula 11).

$$lnRR = \beta_1 \times (Depth - 1) + \pi_{study} + \varepsilon \quad (11)$$

For ease of interpretation, *LnRR* and its corresponding 95% CIs were further transformed into percent soil C_{org} stock change (ΔC_{org},%; formula 12). If the 95% CI of ΔC_{org} does not cover zero, the effect of individual disturbances was significant (positive or negative, at $P = 0.05$) in affecting soil C_{org} stock.

$$\Delta C_{org} = (e^{lnRR} - 1) \times 100 \quad (12)$$

To explore the response of soil C_{org} stock loss to initial C_{org} stock and disturbance duration, we first statistically compared the linear and logarithmic functions with the factor of interest as the fixed effect, and 'site' as the random effect, using the Akaike information criterion (AIC). We found that the logarithmic functions resulted in lower or similar AIC values for the response of soil C_{org} stock change (%) to initial soil C_{org} stock and disturbance duration (Supplementary Table 4), while linear functions resulted in lower or similar AIC values for the response of soil C_{org} stock change (Mg C ha^{-1}) to initial soil C_{org} stock (Supplementary Table 5). Therefore, we used logarithmic or linear functions to construct the response of soil C_{org} stock change to initial soil C_{org} stock and disturbance duration and visualized with ggplot2 (version 3.3.6) packages in R.

Data availability

The source data supporting Figs. 2–5 are available in the Source Data file. The complete database is deposited in Figshare database (https://doi.org/10.6084/m9.figshare.27074590). Source data are provided with this paper.

Code availability

Codes for creating each figure are available at Figshare database (https://doi.org/10.6084/m9.figshare.27074590).

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Author contributions

C.M.D. conceived this study. C.F., S.G.K. and J.B. performed the paper searching and screening. C.F. collected data and conducted the analyses with the support from S.G.K., K.K.L. and A.S. C.F. and C.M.D. wrote the original manuscript. All authors contributed to reviewing and editing of the manuscript.

Competing interests

The authors declare no competing interests.

Additional information

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