Impacts of land management practices on blue carbon stocks and greenhouse gas fluxes in coastal ecosystems – a meta-analysis

(Running head: Analysis of blue carbon management strategies)

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Abstract

Global recognition of climate change and its predicted consequences has created the need for practical management strategies for increasing the ability of natural ecosystems to capture and store atmospheric carbon. Mangrove forests, saltmarshes and seagrass meadows, referred to as blue carbon ecosystems (BCEs), are hotspots of atmospheric CO_2 storage due to their capacity to sequester carbon at a far higher rate than terrestrial forests. Despite increased effort to understand the mechanisms underpinning blue carbon fluxes, there has been little synthesis of how management activities influence carbon stocks and greenhouse gas (GHG) fluxes in BCEs. Here, we present a global meta-analysis of 111 studies that measured how carbon stocks and GHG fluxes in BCEs respond to various coastal management strategies. Research effort has focused mainly on restoration approaches, which resulted in significant increases in blue carbon after 4 years compared to degraded sites, and the potential to reach parity with natural sites after 7 – 17 years. Lesser-studied management alternatives, such as sediment manipulation and altered hydrology, showed only increases in biomass and weaker responses for soil carbon stocks and sequestration. The response of GHG emissions to management was complex, with managed sites emitting less than natural reference sites but emitting more compared to degraded sites. Individual GHGs also differed in their responses to management. To date, blue carbon management studies are under-represented in the southern hemisphere and are usually limited in duration (61% of studies < 3 yrs duration). Our meta-analysis describes the current state of blue carbon management from the available data and highlights recommendations for prioritising conservation management, extending monitoring timeframes of BCE carbon stocks, improving our understanding of GHG fluxes in open coastal systems and redistributing management and research effort into under-studied, high-risk areas.

Introduction

Vegetated coastal ecosystems (mangrove forests, saltmarshes and seagrass meadows) store and accumulate globally significant amounts of organic carbon (McLeod *et al.*, 2011, Nellemann & Corcoran, 2009). Despite occupying less than 1% of the ocean floor, these ecosystems (collectively referred to as blue carbon ecosystem or BCEs) accumulate approximately 50% of all organic carbon buried at sea (Duarte *et al.*, 2013, Serrano *et al.*, 2019). Furthermore, while only occupying 3% of the area of terrestrial forests, BCEs sequester a comparable amount of carbon (McLeod *et al.*, 2011), as carbon burial rates in BCE sediments are approximately 40 times higher compared to forest soils (Breithaupt *et al.*, 2012, Chmura *et al.*, 2003, McLeod *et al.*, 2011, Schlesinger, 1990). This is because the complex vegetated structures in BCEs efficiently trap sediments suspended in tidal flows from internal and external sources, and accumulate carbon via biological inputs from microbial mats and turf algae (Alongi, 2012, McLeod *et al.*, 2011). BCE sediments are also predominantly low in oxygen and high in salinity, which slows down carbon re-mineralisation processes and methanogenesis (Donato *et al.*, 2011, Pendleton *et al.*, 2012, Poffenbarger *et al.*, 2011). This combination of fast carbon burial and slowed carbon re-mineralisation makes carbon sequestration in BCE sediments relevant for mitigating climate change, particularly at the national scale, on which climate mitigation agreements such as the Paris Agreement are based (Taillardat *et al.*, 2018). However, while blue carbon is being increasingly considered by coastal management bodies around the world, there remains limited data on the effectiveness of management on influencing blue carbon stocks.

Current methods for carbon-focused management in BCEs have primarily adopted terrestrial methods and policies (such as Reducing Emissions from Deforestation and forest Degradation, REDD). These often quantify the amount of carbon sequestered as a result of protecting a BCE against ongoing loss from anthropogenic threats such as urbanisation, reclamation, deforestation, eutrophication and pollution (Ahmed & Glaser, 2016, Alongi, 2011, Herr *et al.*, 2017, Lovelock & Duarte, 2019). Adapting such terrestrial forest management strategies to BCEs may enhance carbon sequestration, although the complex and open nature of coastal ecosystems compared to their terrestrial counterparts raise challenges in accurately identifying the underlying mechanisms controlling fluxes of both carbon and greenhouse gases (GHGs) (Belshe *et al.*, 2017, Johannessen & Macdonald, 2016, McLeod *et al.*, 2011). Restoration management for example, which utilises reconstruction or rehabilitation of degraded areas, has long been one of the main approaches used in terrestrial systems to restore ecosystem function to natural areas which have been transformed by deforestation, land-use change and pollution (Camargo *et al.*, 2002, Lamb *et al.*, 2005, Stanturf *et al.*, 2014). Adapting restoration to coastal areas is a more complex process however, which often leads to projects underperforming or varying in success rates (Bayraktarov *et al.*, 2016, Thom, 2000). Innovation to traditional forestry management approaches such as restoration is therefore required in order to increase their efficiency in coastal systems, as is more empirically-driven, evidence-based investigation into the effectiveness of current efforts (Silliman *et al.*, 2015).

Coastal management strategies have been applied all around the world in attempts to enhance blue carbon storage (Howard *et al.*, 2017). These include altering hydrological regimes by increasing or restricting the rate of flow of either saltwater or freshwater, and manipulating sediments to increase or decrease nutrient levels or elevation, or to otherwise enhance conditions for carbon sequestration. For example, restricting freshwater flow by river impoundment or increasing tidal flow under managed realignment alters both the salinity and moisture levels in BCEs, which in turn effects plant growth, soil carbon mineralisation and CO_2 flux (Kathilankal *et al.*, 2008, Setia *et al.*, 2013). While individual studies have assessed carbon-relevant metrics under such management approaches, there is of yet no large-scale synthesis of this data from which to draw conclusions as to the impact of coastal management on blue carbon stocks across different habitats and regions (but see Sasmito *et al.*, 2019).

In order to include these proposed management activities as methods for carbon crediting in emission reduction schemes, we need to clearly demonstrate that a proposed management activity will increase ecosystem carbon stocks over relevant timeframes for climate change mitigation. To address this knowledge gap, we conducted a systematic literature review to determine what empirical evidence is currently available to support the inclusion of BCE management into emission trading schemes for climate change mitigation. Here, we present a quantitative meta-analysis of studies that have reported on changes in BCE carbon stocks or GHG emissions in response to management interventions. The aims of this review were: 1) to assess the current availability of empirical data on blue carbon management, 2) to determine the effect of different management types on carbon sequestration and GHG fluxes in BCEs and 3) to investigate the extent to which factors inherent to different monitoring approaches (i.e. experimental design, timeframe and sampling methodology) influenced measured responses to management.

Materials & Methods

We performed a broad search of the literature for papers concerning drivers of carbon sequestration and GHG emissions in coastal vegetated ecosystems (i.e. mangroves, saltmarsh and seagrass). Our search, using ISI Web of Science on the 1st of June 2018 used the following search terms: (seagrass* OR "sea grass*" OR saltmarsh* OR "salt marsh*" OR mangrove* OR "tidal marsh*" OR "tidal wetland*") AND TS = ("carbon sequest*" OR methane OR geochem* OR CO2 OR CH4 OR N2O OR "nitrous oxide" OR "carbon dioxide" OR "blue carbon" OR carbon OR biomass* OR root* OR management). We did a follow-up search on June 1st, 2019 to include recent studies and scanned the reference lists of relevant reviews and meta-analyses for additional papers. This returned a total of 11,221 papers. We selected studies according to PRISMA protocols (http://www.prisma-statement.org). Firstly, we filtered search results by title to include only those which likely involved management relevant to carbon levels (i.e. implications for carbon metrics including changes in biomass, soil carbon or sequestration rates) in BCEs or blue carbon specifically. We then filtered by abstract which required specific mention of management policy, field monitoring or experimental data collection in BCEs for the purpose of evaluating land management strategies that influence above- or below-ground biomass, carbon fixation or GHG fluxes. We then assessed full texts for empirical data or estimates based on empirical data related to carbon storage. For papers without suitable data (i.e. literature and policy reviews) we performed a supplementary search of bibliographies for potential source papers overlooked in the initial search. A total of 192 studies met our criteria (see PRISMA diagram, Fig. S1). Finally, we eliminated papers which did not utilise control sites for comparison with the managed site or contained either modelled or qualitative data, resulting in 112 studies.

We allocated studies meeting our criteria into the following five broad management categories for analysis: 1) *Altered hydrology* (including managed realignment, impoundment, diking, altered flow of freshwater, seawater or sewerage 2), *Sediment manipulation* (including chemical treatment, artificial substrate, human transported material, dredging, beach renourishment and sediment supply, 3) *Deforestation* (including cutting, clearing, thinning, logging), 4) *Land-use change* (repurposing/reclaiming BCE habitats for other land usages such as agriculture, aquaculture or urban development), and 5) *Restoration* (including reforestation, transplantation, rehabilitation and creation). Where available, we extracted data on study location, habitat type (mangroves, saltmarsh or seagrass), species, management type, study time frame, sampling methodology (e.g. quadrats, sediment cores, eddy covariance), and experimental design. Experimental designs were classified as assessing treatment effects by either comparing sites with and without management (control / impact or CI), comparing either before management to afterwards (before / after or BA), or by a more rigorous combination of the two (before / after / control / response or BACI (Smith, 2014)). For our response variables, we focused on empirical measures extracted from text, tables or figures (using open source graphical digitiser software; Huwalt, 2001) of carbon stocks (i.e. biomass, soil carbon, sequestration) and fluxes (including CO_2 and CH_4, as well as N_2O, another GHG relevant to climate change (Muñoz *et al.*, 2010)) in order to calculate response ratios (RRs) as follows:

For BA and CI studies:

$$\ln[RR] = \ln[B \text{ or } I] - \ln[A \text{ or } C] \qquad (1)$$

and for BACI studies:

$$\ln[RR] = \ln[I_A / C_A] - \ln[I_B / C_B] \qquad (2)$$

where $\ln[RR]$ is the log response ratio, I is the impacted site mean, C is the control site mean, A is the after mean, and B is the before mean (Hedges *et al.*, 1999). As managed sites may be compared to reference or degraded conditions, effect sizes were calculated independently for each comparator type. The effect of management on a given carbon or GHG metric was considered significant if the 95% confidence interval of the mean effect size did not overlap with zero. Where possible, we extracted multiple individual RRs from a single study, rather than averaging across sites to produce a single overall mean. This allowed us to capture as much information on responses to BCE management as possible and left us with a total of 353 RRs to analyse the response of BCEs to blue carbon management.

For each management type, we constructed a series of generalised linear mixed-effects models. Not all combinations of variables (e.g. management type, habitat, sampling methodology, metric) existed in our dataset, largely precluding the exploration of complex interactions. Instead, we examined management-specific responses of carbon storage (sequestration, soil carbon levels and biomass) and GHG fluxes (CH_4, CO_2, N_2O) (fitted as fixed effects) between managed and control sites after separating data by control site comparator (reference or degraded site). We constructed models with each of the variables included as a single factor, and then with all possible two-way interactions where data allowed (see Supplementary Material for model structures). We also included a site nested within study random effect (see *Weighting and non-independence*). We compared competing models using

Akaike's Information Criterion corrected for small sample sizes (AICc; Burnham & Anderson, 2002), and rescaled these values as the difference between each model and the model with the lowest AIC_c (ΔAIC_c).

We also analysed how carbon storage changes with time since management by creating generalised linear mixed-effects models for the management types with sufficient temporal data: restoration, deforestation and altered hydrology. We included comparator type (degraded or reference site) and years (time since management) fitted as fixed effects. We included years in models as non-transformed, curvilinear and log-transformed. We used the same nested random effect and model comparison approach as described above. To plot responses, we produced unbiased parameter estimates and 95% confidence intervals using restricted maximum-likelihood estimation (REML) and supressed intercepts. We plotted response ratios for each management type (separated by comparator), habitat type, study design, carbon data type and GHG, including grand means for each. Grand means were calculated by excluding the fixed effect for each respective model. When sufficient data were not available to run full models, complexity was reduced (e.g. by removing the nested term) so that model estimates and variances could still be extracted. We used the lmerTest package (Kuznetsova *et al.*, 2015) in R v.3.2.2 (R Development Core Team, 2015) to build models and extract least-squares means and confidence intervals (Stanley & Doucouliagos, 2015).

Weighting and non-independence

In general, RRs based on larger sample sizes should contribute more weight to the overall estimate than those based on smaller sample sizes, as variance is typically reduced as sample size increases. Here, many of the studies did not report sufficient information to calculate estimates of variance, and others examined responses in managed sites relative to a single control site. These two issues are relatively common in ecological studies on large study systems (e.g. Sievers *et al.*, 2018), and precludes the calculation of standard weightings used in formal meta-analyses (see Lajeunesse, 2011, Lajeunesse, 2015). When variance estimates are not provided within studies, weighting based on sample sizes can be used (Mengersen *et al.*, 2013b). Instead of omitting a high proportion of studies or conducting unweighted analyses, we calculated weights based on the sum of sample sizes (Stanley & Doucouliagos, 2015). Given we had cases where a single control site was used (since we calculated a separate RR for each managed site), this approach down-weighted these estimates relative to a RR based on the average of multiple sites, helping to deal with non-independence.

In addition, we incorporated two unique identifiers as a random effect, where 'site' was nested within 'study'. Site nested within study accounted for any correlation amongst observations at a given site and accounted for common local environmental or contextual effects. The study random effect accounted for any systematic differences due to common regional environmental conditions or study-specific methodologies or biases. Our model structure therefore allowed us to analyse multiple RRs from a given study rather than having to aggregate data to a single mean value. This ultimately accounted for non-independence of multiple entries extracted from the same study, and multiple studies conducted at the same site (Davidson *et al.*, 2017, Krist, 2011). Furthermore, since our models used maximum likelihood methods, studies were implicitly weighted by the uncertainty of the estimates since the regression analyses (and the variation in the regression estimates) were included as part of the model (Mengersen *et al.*, 2013a).

Results

1) Summary of BCE management data

We allocated studies meeting our selection criteria into the five previously defined management categories as follows: restoration (n = 39) was the most studied management approach in BCEs, followed by deforestation (n = 31), altered hydrology (n = 23), sediment manipulation (n = 11) and conservation (n = 7). Geographically, the majority of studies were conducted in North America (31%), followed by Asia (29%), Europe (20%), Oceania (12%), Africa (4%) and South America (4%; Fig. 1). Habitat types were differently represented across locations, with the majority of studies in Europe and North America focused on saltmarshes while mangrove studies dominated other regions (Fig. 1). Most response ratios across habitats were calculated from biomass data (n = 142), followed by soil organic carbon (n = 112), GHG fluxes (n = 54) and sequestration (n = 45) (Table 1). CI study designs were most commonly used (n = 79), followed by BA (n = 21) and BACI (n = 12) (Figure S2).

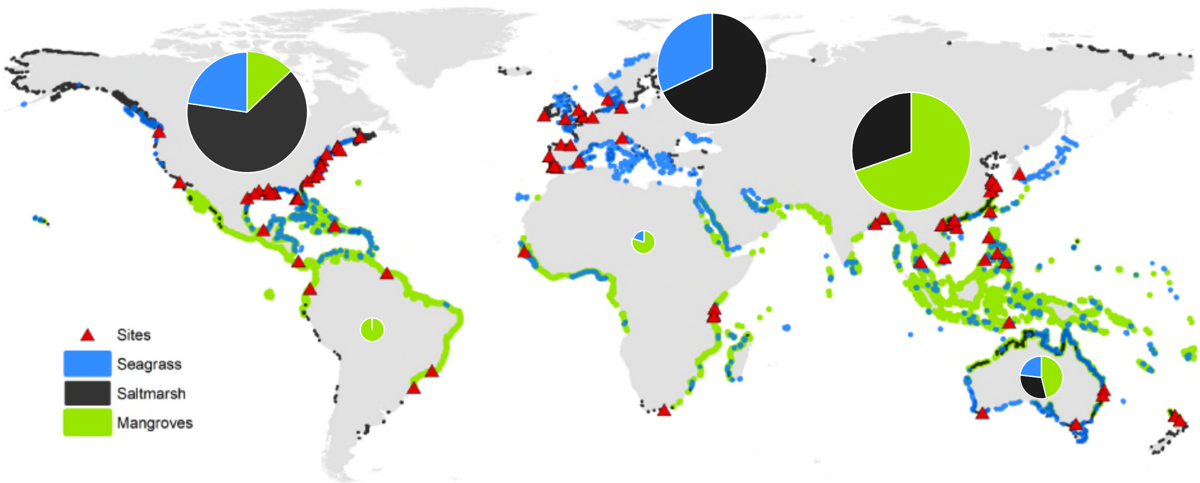

Figure 1. – Regional proportions of studies for each blue carbon ecosystem from North America (n = 33, mangrove = 12%, saltmarsh = 65%, seagrass = 23%), Asia (n = 32, mangrove = 70%, saltmarsh = 30%), Europe (n = 25, saltmarsh = 67%, seagrass = 33%), Oceania (n = 10, mangrove = 43%, saltmarsh = 36%, seagrass = 21%), Africa (n = 4, mangrove = 75%, seagrass = 25%) and South America (n = 4, mangrove = 100%). Three studies pooled data across regions. Individual study site locations indicated and habitat distribution layers are expanded to aid visualisation, adapting existing datasets for mangroves (Giri *et al.*, 2011), saltmarshes (Mcowen *et al.*, 2017) and seagrass habitats (Short, 2016).

Table 1. – Number of response ratios (RRs) taken for each carbon metric from the pool of 112 studies for each management class and habitat. BIO = Biomass, SOC = Soil Organic Carbon, SEQ = Sequestration, FLX = greenhouse gas (GHG) flux (*note: no flux RRs recorded for seagrass, multiple RRs drawn from some studies).

		Mangrove				**Salt Marsh**				**Seagrass**			
Management	**Description**	**BIO**	**SOC**	**SEQ**	**FLX**	**BIO**	**SOC**	**SEQ**	**FLX**	**BIO**	**SOC**	**SEQ**	**Total**
Altered hydrology	Managed realignment, impoundment, diking, altered flow		3	3	3	20	16	5	18	4	1		73
Sediment manipulation	Chemical treatment, artificial substrate, dredging, beach renourishment					27	2	1		5	2		37
Land-use change	Repurposing or reclamation of natural habitats for agriculture, aquaculture or urbanisation	2	5	7	6	1	4		1	1		1	28
Restoration	Reforestation, rehabilitation or creation of new areas via plantation or transplantation	13	18	10	7	43	25	3	11	12	19	2	163
Deforestation	Cutting, clearing or thinning of vegetation	16	6	11	8	1	12	1	3				57
	Total	31	32	31	24	91	59	10	33	22	22	3	358

2) Blue carbon and GHG flux response to BCE management

Combining all management categories, managed sites stored more carbon compared to degraded controls (+125.5% more than sites in degraded condition). Each individual management type significantly increased carbon storage (i.e. biomass, soil carbon and sequestration combined) relative to degraded controls, including sediment manipulation (+427.5%), restoration (+67.8%) and altered hydrology (+176.8%) (we found no deforestation or land-use change studies used degraded reference sites as comparators). Overall, managed sites also stored less carbon relative to natural reference controls (-26.1%). This was largely driven by sites that underwent land-use change (-42.8%) and deforestation (34.1%), then restoration (-33%) and sediment manipulation (-28.3%). On the other hand, sites that had their hydrology altered did not differ statistically from their reference comparators (-1.3%). There was no significant difference in carbon storage responses to management between BCEs, although grand mean RRs show seagrass with a notably higher mean response to management compared to mangroves or saltmarshes (Fig. 2).

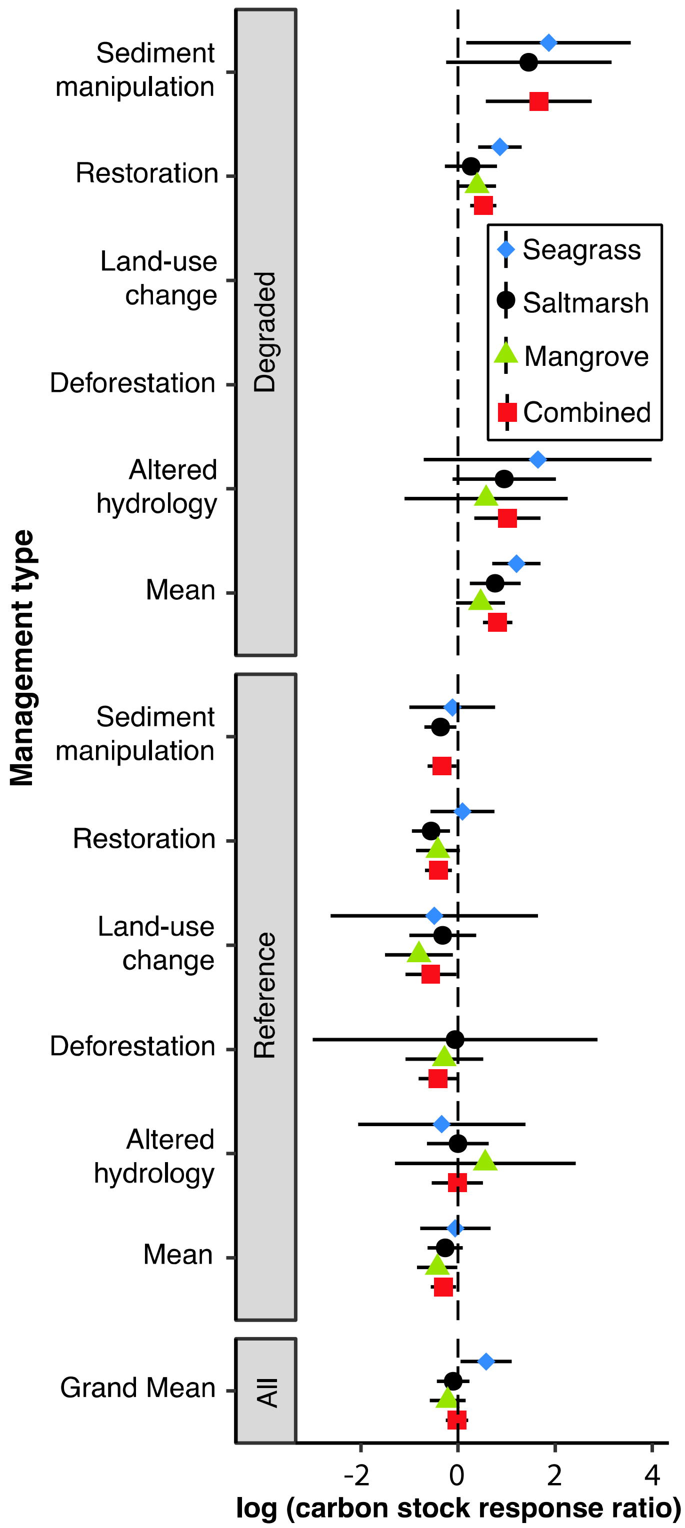

Figure 2. – Forest plots of weighted response ratios (and 95% CI on log scale) for carbon storage across management types in each blue carbon ecosystem, separated by comparator used in the study (i.e. degraded or natural reference control site).

Overall, GHG flux was not significantly different between managed and degraded sites or between managed and reference sites (Fig. 3). However, while most management types did not significantly change GHG emissions, restored sites compared to degraded conditions showed a significant 16.2% emission increase (Fig. 3). For managed sites compared to reference sites, there was considerable variability among management approaches, with sites that underwent restoration (41.6% emission reduction), deforestation (120.1% emission reduction) or had their hydrology altered (18.7% rise in emission) not differing statistically from their reference comparators (Fig. 3).

Mean RRs for each individual GHG across management types were not significantly different with either degraded or reference comparators, except for CH_4. CH_4 emissions increased by 29.3% under restoration and by 89.7% under altered hydrology management compared to degraded sites, and reduced by 464.6% under restoration and by 238.2% under land-use change management compared to reference sites (Fig. 3). Although not significantly different, N_2O showed an emissions increase of 35.7% across every management category, while CO_2 was the only GHG with a positive grand mean response (10.3% emission reduction).

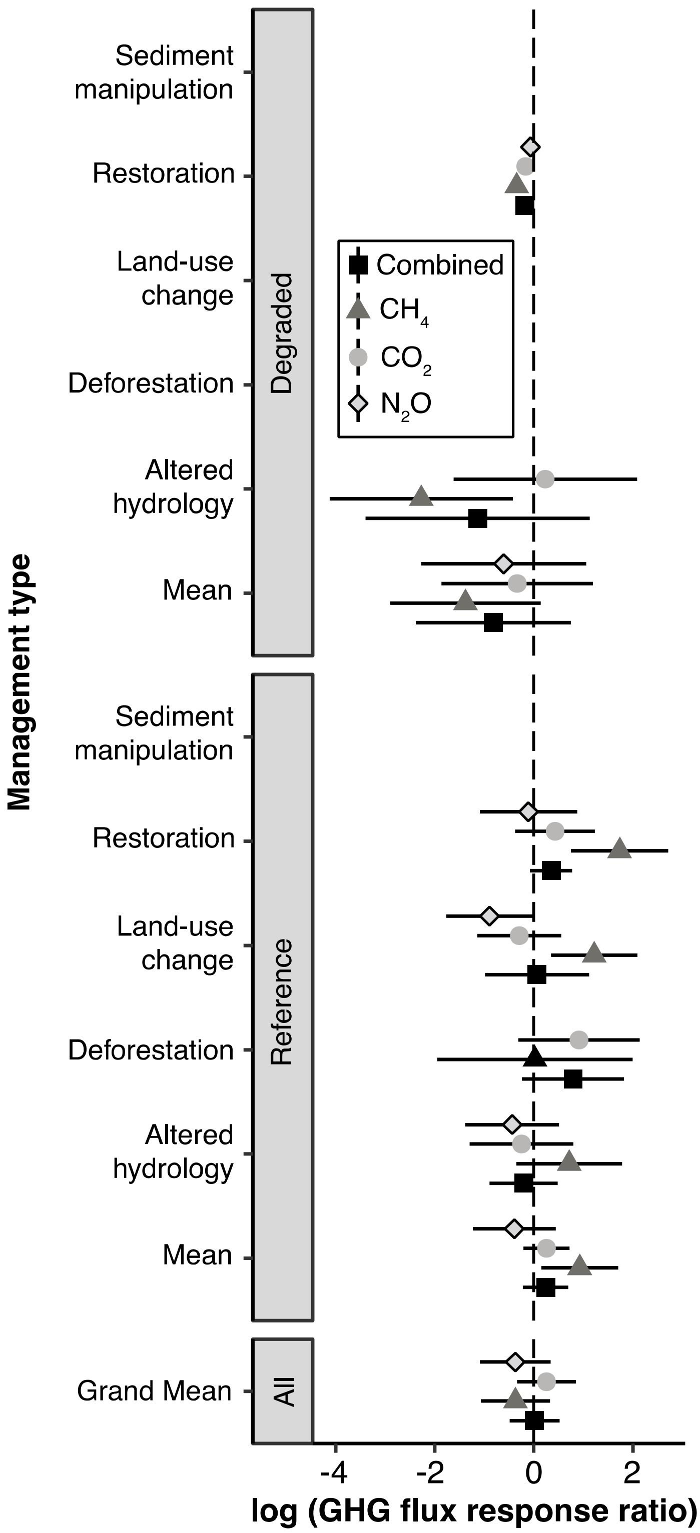

Figure 3. – Forest plots of weighted management response ratios (and 95% CI on log scale) for GHG fluxes (CO_2, CH_4 and N_2O) separated by comparator used in the study (i.e. degraded or natural reference control site). For readability, signs have been reversed such that positive RR = emission reduction and negative RR = emission increase.

3) *Effect of monitoring design on BCE management data*

Overall, estimates from studies that compared managed sites to prior conditions found greater relative carbon storage (BA = 85.7%, BACI = 75.4%) compared to CI studies (-22.6%) when combining comparator types (degraded and reference control sites) (Fig. S2). RRs from CI studies were similar to overall trends, whereby carbon storage was 71.3% greater in managed sites relative to degraded sites, but 33.4% lower relative to natural reference sites. On the other hand, carbon storage RRs from BACI studies increased regardless of comparator type (by 36.1% and 225.8% for reference and degraded comparators respectively), but BA studies followed a similar pattern to CI studies, with carbon storage 215.5% higher relative to degraded sites, but 14.3% lower relative to reference sites. RRs based on the different metrics used to quantify carbon storage – biomass, sequestration or soil organic carbon – were similar in managed BCEs relative to reference controls. When degraded comparators were used, biomass significantly increased by 604.3%, driven by sediment manipulation (1415.3%), restoration (269.8%) and altered hydrology (993.6) (Fig. S3). Sequestration (64.5%) and soil carbon also increased compared to degraded controls (38.6%).

Time since restoration had a significant effect on carbon storage within restored sites. Models with log-linear relationships were most supported (Table S2), with trends significant for both degraded ($p = 0.005$) and reference ($p < 0.001$) comparators. Based on model estimates, restored sites stored significantly more carbon than degraded comparators after 4 years since restoration (RR at year 4: 0.36, CI: 0.03 – 0.68) (Fig. 4). On the other hand, restored sites stored similar amounts of carbon (on average) as reference comparators from 17 years (RR at year 17: 0.01, CI: -0.29 – 0.30), but based on 95% confidence intervals, may reach parity after only 7 years since restoration (RR at year 7: -0.23, CI: -0.47 – 0.02) (Fig. 4). Although there was insufficient data to detect temporal trends for conservation and sediment manipulation, null models for both deforestation and altered hydrology were most supported (Table S2), suggesting no trend through time for these management types.

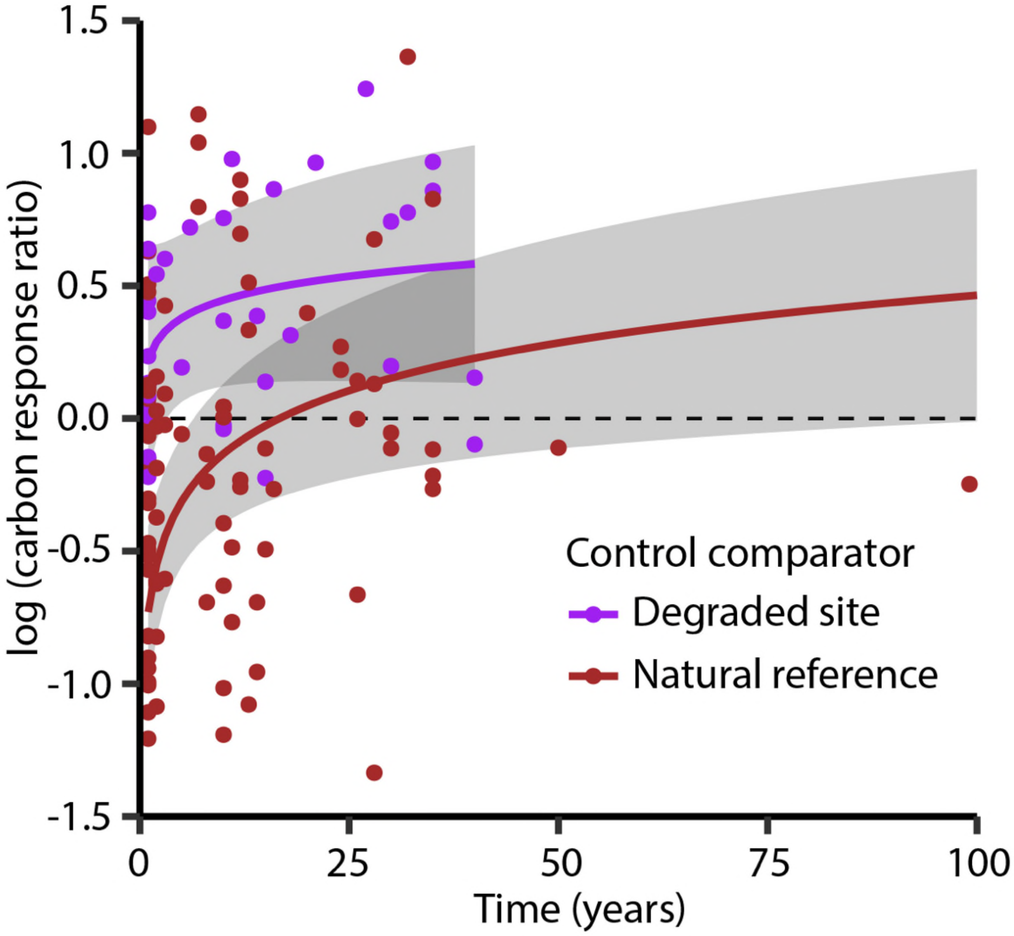

Figure 4. – Log-linear trend lines (with 95% CI) of carbon storage response to restoration in BCEs relative to degraded (i.e. low or pre-managed, n = 40) and reference (natural or undisturbed, n = 100) comparator sites over time.

Discussion

Current limitations and biases of BCE management data and methodologies

Our global meta-analysis of 112 studies from 36 countries provides a comprehensive overview of current research into management approaches intended to influence carbon storage and GHG flux in BCEs. Management-focused studies came predominantly from North America, Asia and Europe, with only 20% of studies conducted in the southern hemisphere, which has a lower proportion of sustainably managed areas and higher threat levels due a combination of industrialisation, land-use change and climate change impacts (Chowdhury *et al.*, 2017, Cresswell & Semeniuk, 2018, Gedan *et al.*, 2009). As an example to illustrate this, Europe and North America combined provided 80% of seagrass management studies in our analysis, while constituting only an estimated 6% of the global seagrass cover (Short, 2016). This contrasts to a country such as Indonesia, which provided only 11.6% of mangrove management studies while accounting for 22.6% of the global cover (Giri *et al.*, 2011). Despite the scarcity of empirical evidence, blue carbon ecosystems in Indonesia are reported to be declining in cover due to land-use change and deforestation (Unsworth *et al.*, 2018), resulting in an annual estimated CO_2 emission to the atmosphere-ocean pool of 29,040 Gg (Alongi, 2016). This geographical bias in management monitoring is unlikely to be linked solely to factors relating to the human development index (HDI), as countries such as Japan (HDI = 0.909) with higher HDIs than Indonesia (HDI = 0.694) returned no studies with empirical management data in our analysis (United Nations Development Programme, 2018), but rather also cultural and political attitudes towards prioritising sustainable BCE management.

Biomass and soil carbon were the most commonly used metrics assessing carbon stocks in BCEs, as common methodologies for obtaining these data require less effort and technical equipment compared to sequestration rates and GHG fluxes. However, it is important to note that there is still debate over the reliability of methodologies currently used to obtain estimates of these carbon stock metrics, more specifically uncertainties around sampling design and processing and under-sampling of spatial variation leading to over- and under-estimations by over 30% (Fest *et al.*, submitted, Jeffrey *et al.*, 2019, Young *et al.*, 2018).

Another clear pattern in our data, also common to ecological meta-analyses, was the dominant use of CI designs over BA or BACI, where managed and control sites were compared without temporal context. This is perhaps expected as despite being less robust to confounding by inherent spatial differences, CI studies are often simpler in terms of experimental design, required funding and monitoring effort (Sievers *et al.*, 2018). However, studies in our analysis over longer timescales showed significant fluctuations and variability in carbon metrics over time (e.g. Craft *et al.*, 2002, Hahn *et al.*, 2015, Lee *et al.*, 2011). In addition, our analyses and estimates of carbon storage over time from other studies indicate temporal thresholds exist, whereby carbon storage gradually increases towards natural levels over a number of years (e.g. Andrews *et al.*, 2006, Salmo *et al.*, 2013). As half of all of studies included here (49%) represented assessments carried out within a timeframe of < 1 year. This is an important factor to account for when assessing past data and during planning of future research. In any case, standard CI studies still showed a clear benefit of BCE management for blue carbon stocks when compared to sites experiencing some form of degradation. When natural reference sites are used for comparison in management assessment, care must be taken in interpreting the results. The carbon response to management reported here was mainly negative when compared to natural sites, however these comparators represent the optimal goal of management, and so interpretation should focus on where management effort closes the gap most efficiently. In this way, it is clear that management altering hydrology or manipulating sediment can have as positive an effect as restoration on blue carbon, and deforestation results in a more substantial decrease from natural levels.

Management effects on blue carbon and GHG flux

Our meta-analysis suggests improvement in carbon stocks of managed areas for each management activity except, as expected, for deforestation and land-use change. Restoration-based management (including reforestation, afforestation, rehabilitation and transplantation) was the most commonly studied. The response of carbon storage to restoration in BCEs was highly dependent on the comparator type used for controls and the time since restoration. Relative to degraded conditions, restored sites stored significantly greater amounts of carbon, suggesting that restoration is a useful management approach for offsetting GHG emissions. However, most studies (81%) compared restored sites to natural reference sites (i.e. those of high condition, or the condition that restoration seeks to attain). Under these circumstances, the mean response was negative, indicating that carbon storage had not reached parity with these reference sites. This is likely due to the short time frame following restoration when samples were taken, with more than 35% of studies reporting carbon storage within a year and 74% of studies reporting carbon storage within ten years. Confirming this, we found a significant positive trend through time, providing evidence that, on average, carbon storage in restored sites is comparable to reference sites after 17 years. Although this indicates restoration is a viable management strategy to enhance carbon storage in BCEs, it also reinforces the need for temporal context to be taken into account when assessing responses to management activities. Short-term monitoring is likely to underestimate achievable carbon sequestration benefits our models suggest maximum increases accumulate over number of years. Individual restoration studies performed over longer time periods reflect similarly increasing patterns of carbon storage, however short-term monitoring-based trajectories could also overstate long-term carbon stock benefits given the asymptotic trend in our models (Burden *et al.*, 2019, Greiner *et al.*, 2013). In any case, the variance in restoration management success highlights the importance of conservation management of pristine BCEs.

Unsurprisingly, carbon storage responses to deforestation and land-use change in BCEs, starkly contrasted with the other management types we analysed. Although deforestation is not a management approach for increasing carbon storage in BCEs, many of the world's BCEs are subject to deforestation through reclamation for urban development and conversion for use in agriculture or aquaculture (Ahmed & Glaser, 2016, Richards & Friess, 2016, Yang *et al.*, 2019). Management involving deforestation clearly showed the most negative effect on carbon storage of any management type we investigated. This result would have been even stronger if not for one study showing a large increase in biomass following mangrove clearing, driven by elevated algal biomass compared to the reference sites (Granek & Ruttenberg, 2008). However, the response of soil carbon storage to deforestation showed an even greater decrease than biomass, likely caused by increased remineralisation of the carbon stored in the sediment as a result of exposure to oxic conditions (Brodersen *et al.*, 2019), but also leaching, erosion and reduced soft sediment accretion levels (Castillo *et al.*, 2017, Duncan *et al.*, 2016). This confirms that the loss of carbon sequestration potential in BCE sediments due to deforestation is far more substantial than the loss of carbon from biomass, which is predicted by the relatively higher sequestration potential of sediment compared to biomass (McLeod *et al.*, 2011). In order to better understand BCE carbon fluxes, more research is needed to quantify the amount of belowground carbon subsequently lost via CO_2 and CH_4 emissions back to the atmosphere.

Altering hydrological regimes in BCEs increased both carbon storage and GHG emissions. The GHG-specific analysis indicated that CH_4 was the primary driver of this trend rather than CO_2, though these fluxes were not significantly different. Studies in this management category primarily aimed to increase or reinstate saltwater flows to BCEs, most commonly via tidal restoration. Studies which involved restricting salt water flow often noted reductions in biomass and soil carbon (Boyd & Sommerfield, 2016, Yang *et al.*, 2017). Increased moisture and salinity are conducive to effective carbon storage, as oxidation, methanogenesis and denitrification are reduced (Livesley, 2012, Marton *et al.*, 2012). Tidal restoration is also considered an effective management tool for reducing GHG emissions (Kroeger *et al.*, 2017). We found, however, that sites managed this way were also potential sources of N_2O and CH_4 (Adams *et al.*, 2012, Hahn *et al.*, 2015). Differing GHG flux responses to altered hydrology management are often dependent on elevation and rainfall, as managed sites with higher elevation and more stable patterns of precipitation have comparatively lower GHG emissions (Burden *et al.*, 2013, Mazik *et al.*, 2010, Negandhi *et al.*, 2019). Although the vast majority of studies in our meta-analysis investigating the effects of altered hydrology in BCEs were from saltmarsh sites, these processes are likely to be similar for mangroves (Kroeger *et al.*, 2017). For naturally inundated seagrass habitats, increases in salinity to enhance carbon storage potential were achieved through reducing freshwater in-flow. This approach increased biomass and may also reduce turbidity and hydraulic disturbance (Adams & Talbot, 1992, Leston *et al.*, 2008).

Sediment manipulation has the potential to enhance carbon budgets in BCEs via establishing optimal elevations, increasing soil fertility and providing a buffer from variable environmental effects to increase biomass. Carbon storage in BCEs that underwent sediment manipulation was similar to other management types included here (i.e. higher compared to degraded sites and lower compared to reference sites), though temporal scale was limited, with only 5% of studies quantifying carbon stock beyond 1 year since manipulation. More than any other management type however, threshold effects must be considered as excessive sediment manipulation quickly leads to detrimental impacts on carbon storage. In saltmarshes, sediment addition has an optimal level for restoration of vegetation which varies relative to local hydrology, above and below which growth and soil development are impaired (Mendelssohn & Kuhn, 2003, Tong *et al.*, 2013). The effect of sediment supply is also dependent on soil characteristics, with sediment-poor sites much more sensitive to carbon storage enhancement via this method than sediment-rich sites (Mudd *et al.*, 2009). Seagrass habitats are particularly sensitive to sediment manipulation, and loss of biomass due to increased turbidity and smothering as a result of sediment movement must be considered in the management of these and adjacent areas (Gonzalez-Correa *et al.*, 2009). Fertilisation and nutrient addition to sediments can be beneficial for managed seagrass habitats, but again, threshold effects, which are often determined in seagrasses by local light availability, sediment redox conditions and trophic interactions (Peralta *et al.*, 2003, Tol *et al.*, 2016), will need to be considered when managing for carbon storage.

GHG flux responses to BCE management, similarly to carbon storage, were dependant on the comparator type used in the study. When natural reference sites were used as the control comparator, GHG emissions were lower for most management types, though altered hydrology did not follow this trend. For studies where a degraded control was the comparator, we observed higher overall GHG emissions for all management types. The GHG trend for management types in studies where a degraded comparator was the control were mostly driven by changes in CH_4 flux, while CO_2 and N_2O fluxes were very similar. A higher CH_4 emission as a consequence of restoration and altered hydrology is most likely related to an increase in methanogenic processes in the sediment (Livesley & Andrusiak, 2012, Poffenbarger *et al.*, 2011). Given that most studies only monitored GHG flux over limited timeframes (< 1 year) it is highly likely that increased soil moisture and inundation lead to an initial increase in methanogenic activity in the sediment and therefore higher emissions of CH_4. Furthermore, methanotrophic bacterial communities in the soil that consume CH_4 under oxic conditions (Fest *et al.*, 2017) will likely start to adapt to the lower oxygen conditions by relying on porewater exchange in shallow sediment structures such as crab burrows (Conrad & Rothfuss, 1991, Nauer *et al.*, 2018).

Interestingly, across management types, sites that were managed had lower CH_4 fluxes compared to reference conditions. This is again likely related to changes in the ratio of methanogenic and methanotrophic processes in the sediment as a result of increased sediment moisture levels. In addition, an initial increase in carbon remineralisation via methanogenesis is likely to take place unless sediment and water salinity levels reach the critical threshold for methanogenesis (Poffenbarger *et al.*, 2011, Sela-Adler *et al.*, 2017). This observation highlights that management aiming to increase carbon stocks in BCEs could unintentionally lead to increased methane emissions. Methane thus needs to be monitored so that it can be accounted for in carbon offset programs. Only then can we achieve a more realistic picture of carbon sequestration in BCEs which considers the ratio between carbon sequestration and the GHG emissions potentially offsetting or reinforcing blue carbon benefits (Neubauer & Megonigal, 2015).

Emissions of N_2O were similarly elevated at managed sites relative to reference sites. A variety of different metabolic pathways can lead to changes in N_2O fluxes (Butterbach-Bahl *et al.*, 2013, Davidson *et al.*, 2000). Generally, higher N_2O emissions due to increased sediment moisture levels and inundation is most likely related to increased denitrification in the sediment when water-filled pore spaces increase and oxygen levels decline (Bollmann & Conrad, 1998, Linn & Doran, 1984). Management altering hydrology to increase flow to BCEs therefore may be conducive to elevated N_2O emissions. In BCE soils and sediments, however, N_2O emissions can originate from multiple processes (nitrifier nitrification, nitrifier denitrification, denitrification and co-denitrification) which can occur in parallel across the aerobic-anaerobic sediment continuum (Butterbach-Bahl *et al.*, 2013). Given the overall trend of higher N_2O emissions in response to management activities in BCEs, more research on quantifying these emissions is needed given the high global warming potential of N_2O compared to CO_2.

Across all management types, CO_2 flux was similar to both reference and degraded controls. The slightly elevated CO_2 emissions for restoration management in studies with degraded site comparators, which enhanced carbon stocks, may be related to an increase in sediment microbial processes or a shift in sediment microbial communities in response to a change in the quality of the organic material that reaches the sediment (Chen *et al.*, 2012, Schlesinger & Andrews, 2000). An increase in belowground biomass and litter as a result of ecosystem restoration will likely lead to increased organic matter input into the top sediment layers and anaerobic and aerobic diagenetic processes can lead to higher soil CO_2 emissions (Lloyd & Taylor, 1994, Schlesinger & Andrews, 2000). In addition, root respiration can contribute to increases in sediment CO_2 emissions (Elberling *et al.*, 2011, Lloyd & Taylor, 1994, Raich & Schlesinger, 1992, Schlesinger & Andrews, 2000). Again, it is important to consider these GHG fluxes when monitoring carbon stocks in order to properly understand carbon budgets in BCEs.

Implications for managing BCEs for climate mitigation

We synthesised, for the first time, data from empirical studies focusing on the management of carbon and GHG fluxes in BCEs globally and showed that restoration, altered hydrology and sediment manipulation methodologies have demonstrated potential to positively influence sequestration of blue carbon by improving various carbon metrics. Additionally, our analysis provides estimates of the relative consequences to various blue carbon stock metrics of disturbing BCEs for land-use change or other purposes impacting biomass, sediment or hydrology. Perhaps more importantly, our meta-analysis demonstrates the low number of management studies using empirical data, particularly with a robust (i.e. BACI) design structure. Despite this, there are BCE restoration projects being implemented which state the management approaches we investigated here as “applicable” and “appropriate” methodologies (e.g. Verified Carbon Standard, 2015). Our data suggest that BCE restoration may not return carbon stocks to natural levels at the decadal scale. This, combined with the negative carbon stock response to land-use change, emphasises the importance of prioritising BCE conservation management options. In other words, blue carbon management preventing degradation provides greater dividends than rehabilitating degraded areas.

Overall, GHG fluxes in BCEs need more attention given that they are not often assessed and are difficult to quantify, as lateral exchange in these open systems can remove large amounts of GHGs in dissolved form during high tide and via porewater exchange (Fuentes & Barr, 2015, Maher *et al.*, 2013, Santos *et al.*, 2019, Sippo *et al.*, 2017). Studies concentrating on surface-to-atmosphere GHG exchange will therefore not be able to accurately capture sediment carbon cycling, a factor which is as of yet largely unaccounted for in BCE monitoring (but see Maher *et al.*, 2018). This may in part account for the clear differences detected between carbon stocks in control and managed sites but not for GHG fluxes in our analysis. A greater understanding of the factors driving GHG fluxes in BCEs, how to monitor them and how they influence blue carbon budget estimates is desperately needed, particularly as management plans will need to consider GHG fluxes in order to ensure more realistic assessment of their impact on climate change mitigation efforts in BCEs.

Where possible, assessment of managed BCE sites should incorporate multi-year monitoring designs in order to account for temporal variability in environmental conditions that affect sediment carbon fluxes. In addition, significant long-term increases in carbon stocks may only be verifiable after decades of management. The initial condition of the managed site is also a major determinant of the effectiveness of carbon stock enhancement, and thus should be thoroughly assessed during the planning phase. This would allow BACI study designs to be implemented, increasing our ability to more accurately assess the effects of the management action. This review of the effect of different management types on blue carbon in mangrove, saltmarsh and seagrass habitats highlights the scarcity of studies currently available to guide decision-making, and outlines factors to be accounted for in the monitoring, evaluation and reporting of blue carbon management plans, order to maximise the potential for BCEs to contribute to offsetting of global CO_2 emissions. As BCEs are increasingly considered as instrumental for carbon storage and helping offset anthropogenic CO_2 emissions, it is important that we develop a comprehensive understanding of how different management approaches influence their ability and capacity to store carbon. We hope that this quantitative analysis provides the basis for this understanding and will help guide future research into this topic. In light of the available data showing the variability in carbon sequestration benefit from different restorative management approaches (particularly taking into account their relative costs) compared to the detrimental effects of deforestation and land-use change, and the estimated time frames to reach natural parity determined in our analysis, we recommend that 1) conservation management be prioritised in these systems, 2) monitoring of blue carbon management projects be extended to a minimum of 7 years to account for temporal factors and 3) GHG fluxes in BCEs require more investigation and incorporation into management design to improve carbon budget estimates.

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