Coastal Wetland Restoration and Greenhouse Gas Pathways: A Global Meta-Analysis

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Abstract

Coastal wetland restoration is widely promoted as a tool for climate change mitigation, but its effect on the carbon cycle is not well constrained. We conducted a systematic review and meta-analysis of peer-reviewed field studies that directly contrasted restored with altered sites, covering carbon stocks and greenhouse gas fluxes across mangroves, saltmarshes, seagrass meadows, brackish systems, and coastal freshwater wetlands. Literature searches yielded 86 studies and 369 pairwise restored versus altered site comparisons. Multilevel random-effects models with nested study effects showed significant increases after restoration in soil carbon, aboveground biomass, and belowground biomass. Mean greenhouse gas flux changes after restoration were non-significant for CO_2, CH_4, and N_2O. Meta-regressions detected no significant differences among wetland types, though this result is constrained by unbalanced evidence across systems and studied parameters. The available data are geographically biased toward tropical and subtropical Asia, with minimal coverage in Africa and limited data from temperate and cold coastal regions. Among the covered variables dissolved organic carbon is critically underrepresented, constraining whole-system impact estimates. Overall, the data examined in this study show that restoration consistently rebuilds biomass and soil carbon without a detectable systematic "cost" from CH_4 or N_2O, indicating positive outcomes for greenhouse gas fluxes. To translate these findings into policy-ready estimates, monitoring of greenhouse gases and dissolved organic carbon should be expanded, altered versus restored designs should be prioritized, and underrepresented regions and wetland types should be targeted.

Keywords: Coastal wetland restoration, Blue carbon, Carbon stocks, Greenhouse gas fluxes, Methane, Nitrous oxide, Climate change mitigation, Meta-analysis

Introduction

Coastal wetlands, including mangroves, saltmarshes, seagrass meadows, and various brackish-to-freshwater ecosystems, form a diverse group of aquatic systems at the interface between land and sea. Despite covering less than 0.2% of Earth's surface, they provide crucial ecosystem services, including erosion and storm-surge protection, habitat provisioning, recreation, and climate regulation (Barbier et al., 2011; Convention on Wetlands, 2025). Coastal wetlands regulate the climate by releasing or sequestering carbon and storing large amounts of carbon (i.e., blue carbon) and nitrogen in vegetation biomass, in waterlogged soils and sediments, and in the water column as recalcitrant dissolved organic carbon (DOC; Alongi, 2014; Bogard et al., 2020; Mcleod et al., 2011). In intact and waterlogged soils and sediments, in particular, OC can persist for centuries to millennia, creating a vast sink of atmospheric CO_2 (Alongi, 2014; Chmura et al., 2003). Nevertheless, this effect can be offset by the release of CO_2 produced during aerobic remineralization of organic matter (Kristensen et al., 2025) and by CH_4 produced through methanogenesis in anoxic soils or sediments (Arai et al., 2021; Rosentreter et al., 2021). Nitrous oxide (N_2O), which can arise in the nitrogen cycle through nitrification or incomplete denitrification, also has a negative effect on the climate when released into the atmosphere (J. Liu et al., 2021; Moseman-Valtierra et al., 2022).

Over the past centuries, a substantial proportion of the world's coastal wetlands have been lost due to anthropogenic and climatic disturbances, with losses accelerating in recent decades (Fluet-Chouinard et al., 2023; Mao et al., 2018). For example, in Europe, over 65% of coastal wetlands have disappeared since 1900 (Airoldi & Beck, 2007). In addition to land-use pressures, accelerating sea-level rise contributes to the loss and fragmentation of coastal wetlands through increased erosion, saltwater intrusion, and habitat displacement (Charles et al., 2019; Yu et al., 2019). Where infrastructure restricts landward migration, sea-level rise further limits their capacity to adapt (i.e., coastal squeeze; Schuerch et al., 2018; Zhi et al., 2022). Remaining coastal wetlands also affected by multiple stressors, including climate change, invasive alien species, eutrophication, and local anthropogenic pressures (e.g., land-use change), resulting in changes in the carbon cycle and their role as climate regulators (Convention on Wetlands, 2025).

Restoration of coastal wetlands has emerged as a key strategy to recover lost ecosystem functions and enhance climate change adaptation and mitigation, for example by improving flood regulation, increasing carbon sequestration, and reducing greenhouse gas (GHG) emissions. While early restoration actions prioritized biodiversity and habitat conservation, recent international policy initiatives, including the Ramsar Convention (Convention on Wetlands of International Importance Especially as Waterfowl Habitat., 1971), the UN Decade on Ecosystem Restoration (United Nations, 2019), and the EU Nature Restoration Law (European Parliament and the Council of the European Union, 2024), have recognized the role of wetlands in climate change. As coastal ecosystems are highly efficient blue carbon sinks, capable of sequestering carbon at rates exceeding those of most terrestrial forests on a per-area basis (Taillardat et al., 2020), their

restoration might enhance carbon sequestration and avoid emissions by preventing the release of previously sequestered carbon (Lovelock et al., 2023). Yet, the net climate mitigation potential of coastal wetland restoration remains poorly quantified at continental to global scales. Prior syntheses have either focused on intact systems or only a subset of carbon metrics, leaving critical gaps in our understanding of how restoration alters carbon storage and GHG fluxes (Macreadie et al., 2019; Taillardat et al., 2020). To inform policy (such as the 2027 IPCC Methodology Report on Carbon Dioxide Removal Technologies, Carbon Capture, Utilization, and Storage for National Greenhouse Gas Inventories), existing data on the climate change mitigation potential of coastal wetland restoration needs to be compiled and standardized across systems, metrics, and regions.

Coastal wetland restoration encompasses a range of interventions from passive approaches based on natural regeneration after removal of the pressures, to active measures that directly manipulate environmental conditions within wetlands or their catchment. These interventions affect the carbon cycle through four interconnected drivers: sediment, hydrology, salinity, and vegetation. First, sediment-based restoration (e.g., augmentation, engineered deposition, diversions) restores elevation, reduces erosion, and sustains wetland functioning (Barbier et al., 2011; Elsey-Quirk et al., 2019), boosting plant productivity and thereby carbon storage, while avoiding losses via erosion and sediment oxidation (Eagle et al., 2022; Khalil & Finkl, 2011; Matzke & Elsey-Quirk, 2018). Additionally, sediment inputs influence nutrient and carbon cycling, affecting N_2O emissions through denitrification (Comer-Warner et al., 2022; X. Li et al., 2023) and contributing to DOC cycling by burying organic matter and reducing oxidative DOC release (Maher et al., 2013; Neubauer & Anderson, 2003). Second, hydrological restoration (e.g., levee removal, tidal inlet reopening) reestablishes natural tidal exchange and water level, improving sediment transport and habitat connectivity (P. Williams, 2001). Increased waterlogging increases carbon burial and suppresses aerobic OC decomposition (Eagle et al., 2022), thereby decreasing CO_2 production but potentially stimulating CH_4 production under brackish or freshwater conditions, while tidal flushing and mineral sediment inputs help to constrain these emissions (Kroeger et al., 2017; Rochera et al., 2025; T. Williams et al., 2025). N_2O fluxes typically decline after hydrological restoration through efficient denitrification under saturated conditions, while flushing increases DOC export from coastal wetlands into the ocean carbon pool (Maher et al., 2013; Neubauer & Anderson, 2003). Third, salinity restoration (e.g., reintroducing tidal flows, managing seawater intrusion, controlling freshwater inflows) reshapes vegetation composition and microbial processes in the soil. In saline environments, sulfate reduction outcompetes methanogenesis, reducing the likelihood that CH_4 emissions offsets CO_2 uptake (Camacho et al., 2017; Kroeger et al., 2017). Finally, vegetation restoration (e.g., grazing exclusion, invasive species removal) supports ecosystem recovery by stabilizing sediments, enhancing organic matter inputs, and promoting belowground carbon storage (Gedan et al., 2011; Maxwell et al., 2024; Zedler & Kercher, 2005). Increased photosynthesis and biomass accumulation enhance carbon storage (Eagle et al., 2022), while rapid plant growth in brackish or freshwater environments can stimulate methanogenesis through increased substrate supply and anoxic soil conditions. N_2O emissions often remain relatively low when plant uptake and reducing soil conditions limit the availability

of substrates for N_2O production (Comer-Warner et al., 2022). Vegetation also influences DOC fluxes via litter inputs and root exudation (Tzortziou et al., 2008).

Although coastal wetland restoration is widely implemented, its role in climate change mitigation potential remains poorly quantified, reflecting a global research priority in blue carbon science (Macreadie et al., 2019). Existing literature is dominated by studies on a few types of coastal wetlands (mangroves, saltmarshes), which do not capture the full diversity of coastal wetlands or their system-specific carbon dynamics and restoration trajectories (Lu et al., 2017; Taillardat et al., 2020). Restoration outcomes are commonly assessed by comparing carbon stocks in restored and undisturbed reference wetlands, a method that reveals ecological recovery but not necessarily a net climate benefit. Policy-relevant assessments demand comparisons between restored and degraded sites to quantify restoration-driven gains in carbon storage, while also integrating GHG fluxes and DOC alongside carbon stock (Taillardat et al., 2020). We systematically reviewed and analyzed peer-reviewed research comparing restored and degraded coastal wetlands, focusing on key carbon pathways, including aboveground and belowground biomass, soil carbon, DOC, and GHG fluxes (CO_2, CH_4, N_2O). Our objectives were: (1) to identify biases and gaps in the literature, in terms of studied parameters, geographical distribution, and coastal wetland types; (2) to assess whether restoration enhances carbon storage and modulates GHG fluxes and DOC relative to degraded conditions; and (3) to compare outcomes among different coastal wetland types. We hypothesized that (1) existing research has focused on carbon stocks (soil and biomass) rather than GHG fluxes and DOC, and on a narrow range of ecosystems; (2) restoration generally increases carbon storage (in biomass and soils), while responses of GHG fluxes and DOC may be more variable; and (3) restoration outcomes differ by wetland type. By comparing restored and impacted sites, an approach aligned with policy needs, our analysis provides critical evidence to quantify the climate mitigation potential of coastal wetland restoration, and to inform restoration planning, funding allocation, and site prioritization.

Methods

Literature Selection

The literature selection workflow followed the PRISMA 2020 framework (see Figure S1; Page et al., 2021). Peer-reviewed studies reporting original data on restoration effects on carbon storage and GHG fluxes in coastal wetlands: soil carbon (including sediment carbon), aboveground and belowground biomass, DOC, and GHG fluxes (CO_2, CH_4, N_2O) were systematically searched. Searches were run on the Web of Science Core Collection and Scopus in January 2024 and updated on 7 October 2024 and 12.01.2026, including newly published studies. The following search query was used on the Web of Science to search titles, abstracts, and keywords:

225 “TS=(("carbon stock*" OR "carbon sequestration*" OR "carbon storage" OR "carbon flux*" OR
226 GHG* OR "organic matter*" OR greenhouse* OR CO_2 OR CH_4 OR N_2O OR DOC OR TOC OR
227 DOM OR "recalcitrant carbon" OR "soluble organic matter" OR "soluble organic carbon" OR
228 methane OR "nitrous oxide" OR "carbon dioxide" OR "aboveground biomass" OR "belowground
229 biomass") AND ("coastal wetland*" OR lagoon* OR "salt marsh*" OR saltmarsh* OR "tidal
230 marsh*" OR estuar* OR delta* OR mangrove* OR seagrass OR "sea grass" OR "tidal wetland*")
231 AND (restoration* OR rehabilitation* OR revitali*ation* OR renaturali*ation* OR
232 management*)) AND LA=(English) AND DT=(Article OR Data Paper)”.

233 For SCOPUS the search query was slightly adjusted to fit its specific terminology (see
234 supplementary methods). The searches retrieved 4,921 records from Web of Science and 4,994
235 from Scopus. After duplicates were removed using dplyr (R 4.3.2; R Core Team, 2025; Wickham
236 et al., 2020), 7,786 unique records remained.

237 **Study selection and data extraction**

238 The 7,786 records were screened for eligibility against predefined inclusion and exclusion criteria
239 (Table S1). Studies were considered eligible when they: (i) were peer-reviewed field studies in
240 coastal wetlands, (ii) directly contrasted altered with restored conditions using Before-After,
241 Control-Impact, Before-After-Control-Impact designs, and (iii) reported sufficient statistics to
242 compute effect sizes (mean, number of replicates, standard deviation (SD) or convertible
243 dispersion metrics (standard error (SE), confidence interval (CI), interquartile range, range)) for at
244 least one target variable (carbon pools or GHG fluxes). We excluded laboratory or mesocosm
245 work, modelling-only studies, reviews and meta-analyses, comparisons of restored sites with
246 natural reference sites only, non-coastal ecosystems, and cases where variance or sample sizes
247 could not be derived.

248 Studies were distributed randomly among co-authors for title and abstract screening against the
249 inclusion and exclusion criteria (Table S1). Each record was evaluated independently by two
250 reviewers. If both agreed that the study did not meet inclusion criteria, it was excluded. All others
251 advanced to full-text assessment. Full texts were reviewed by a different reviewer using the same
252 criteria. For studies still suitable, relevant data and descriptive information were extracted. A
253 second contributor verified every extraction and requested revisions where needed. When key
254 statistics were missing for a publication, the corresponding authors were contacted for clarification
255 when the study was published in the past 5 years, and otherwise the papers were removed.

256 Data from the 86 eligible papers were extracted into a single, structured spreadsheet. For every
257 pairwise comparison (altered versus restored site; one parameter), we captured core metadata
258 (study overview, alteration and restoration context), the parameter of interest (mean, SD, number
259 of replicates, units) and methodological notes (analytic method, used instruments). Various types
260 of data formats were summarized to result in one value per parameter, per restoration and alteration
261 site, following the information available in the publication. Out of the 86 eligible papers, 369

pairwise comparisons were extracted. Each comparison was assigned to a Köppen-Geiger climate zone (Beck et al., 2018). For clearer interpretation, we grouped the climatic zones Cfa (Temperate, no dry season, hot summer) and Cwa (temperate, dry winter, hot summer) into a "subtropical" category. Five wetland types were distinguished: mangrove, saltmarsh, seagrass, freshwater wetland, and brackish wetland. Classification followed the terminology used in the original studies. Where this was absent, systems were assigned based on reported structural, and functional characteristics, particularly vegetation type, salinity regime, and degree of marine influence (Table S2). Definitions for all alteration and restoration categories are provided in Supplementary (Tables S3 and S4).

Statistical Analysis

All statistical analyses were performed in R 4.3.2 (R Core Team, 2025). To show whether the literature was biased in what was studied and where, we tallied comparisons by parameter, time since restoration, wetland type, and climate zone and mapped study locations.

For each altered versus restored comparison, we calculated two effect sizes using the "escalc" function from the *metafor* package (Viechtbauer, 2010): (i) the standardized mean difference (SMD), also known as Hedges' g (Hedges, 1981), and (ii) the log coefficient of variation ratio (lnCVR; Nakagawa et al., 2015; Senior et al., 2020). A positive Hedges' g indicates that the restored site has a higher mean than the altered site, while a positive lnCVR indicates that the restored site has a greater relative variability than the altered site. To permit computation when studies reported exact zeros, we applied two prespecified rules, symmetrically to restored and altered groups: a small constant (1×10^{-6}) was added to means and SDs that were exactly zero, and SDs were floored at 5% of the absolute group mean to prevent undefined coefficient of variation (CV) and extreme standardized differences caused by zeros (Ren et al., 2019; Sweeting et al., 2004). For lnCVR, CV was computed as SD divided by the absolute value of the mean to avoid sign artefacts.

To test whether restoration changes carbon pools and fluxes relative to degraded sites, we fit intercept-only multilevel random-effects models separately for each parameter using the "rma.mv" function from *metafor* with restricted maximum likelihood (REML). The random structure was ~ 1 | study/substudy, to account for within-study dependence. Here, study indexes the publication and substudy indexes distinct experimental units within the publication, e.g., different restoration sites or different restoration actions tested under a common design. This nesting accounts for within-paper dependence and prevents over-weighting studies that contribute many related comparisons. DOC was excluded from the meta-analysis due to an insufficient number of studies reporting results (n = 2).

We tested whether outcomes differed among coastal wetland types with parameter-multilevel meta-regressions ("rma-mv" in *metafor*), including wetland type as a categorical moderator. Models used REML and random intercepts nested by study and substudy (~ 1 | study/substudy).

For each meta-regression, we report the Wald χ^2 tests (reported by *metafor* as QM), which tests the omnibus null hypothesis that the moderator explains no variance in effect sizes across wetland types. For each parameter, we fit intercept-only random-effects models within each wetland type to obtain pooled effects with 95% CIs. We fit subgroup models only when a wetland type had at least two comparisons; when only one comparison was available, we showed the raw point without a pooled estimate. All effect sizes were modelled as SMDs with 95% CIs.

Small-study effects and publication bias were evaluated from unilevel random-effects models ("rma" in *metafor*) fitted per parameter, because Egger's regression is defined for that framework. Funnel plots, which display individual effect sizes against their standard errors and should appear symmetric around the pooled estimate in the absence of bias, were generated from these unilevel models. Egger's regression ("regtest" in *metafor*), a test for funnel plot asymmetry that detects whether small studies tend to report disproportionately large effects, was applied only when $k \geq 10$. We also computed Rosenthal's fail-safe N ("fsn" in *metafor*) as a complementary diagnostic to estimate how many null-results studies would need to exist, unpublished, to reduce the pooled effect to non-significance. All figures were created using the *ggplot2* package (Wickham, 2016), with spatial data handled using *sf* (Pebesma, 2018) and *rnaturalearth* (South, 2017) for mapping study locations.

Results

Dataset overview

We synthesized global data about coastal wetland restoration from scientific literature, resulting in a total of 369 pairwise comparisons (restored versus altered) from 86 studies. Most studies provided data on soil carbon (Figure 1A, 139 pairwise comparisons) and on aboveground (82) and belowground biomass (50). GHG fluxes were studied less frequently, with 35 comparisons for CO_2, 34 for CH_4, and 20 for N_2O. Only two studies covered DOC, so it was excluded from all subsequent analyses. This global dataset covers various study designs where the time between the start of the restoration and sampling ranged from 1 to 114 years (Figure 1B). In 22 comparisons, the restoration took place within 2 years prior to the study, 86 comparisons were made between 2 and 5 years after restoration, 83 comparisons between 5 and 10 years, and 132 comparisons more than 10 years after restoration, while for 46 comparisons, the time of restoration was not explicitly reported. Looking at the types of coastal wetlands studied (Figure 1C), we saw that mangroves (179), saltmarshes (108), and seagrass meadows (44) were most commonly studied, while only 20 comparisons were found for freshwater wetlands and 18 for brackish wetlands. Looking at the climate zone where the research was conducted, we found most comparisons in subtropical (181) and tropical (115) sites, while 54 comparisons were found in temperate, 17 in arid, and only 2 in cold sites (Figure 1D).

Considering the geographical distribution of the studies (Figure 2), we found 221 pairwise comparisons from Asia, followed by North America (67 comparisons), Europe (35), Oceania (28),

South America (13), and Africa (5). The variables measured also differed across continents (Figure 2, Figure S2). In Asia, most of the research reported on soil carbon (38% of the comparisons), aboveground (23%) and belowground (18%) biomass, and only smaller proportions investigated GHG fluxes (CO_2: 9%, CH_4: 9%, N_2O: 3%). In Europe, studies showed a similar dominance of soil carbon (46%), but with fewer comparisons for aboveground (14%) and belowground biomass (14%), and more to CH_4 (11%), CO_2 (9%), and N_2O fluxes (6%). Studies from North America were more evenly distributed: soil carbon (24%), aboveground biomass (34%), belowground biomass (9%), CO_2 (10%), CH_4 (10%), and N_2O fluxes (10%) and DOC (3%). In Oceania, half of the studies addressed soil carbon (49%), while 11% focused on belowground biomass, and 14% each for CH_4, CO_2, and N_2O fluxes; aboveground biomass was not reported. In South America, most studies focused on soil carbon (70%), with some reporting data on aboveground (15%) and belowground biomass (15%), but none on gas fluxes. In Africa, the number of comparisons from two studies only contains soil carbon (40%), aboveground biomass (20%), belowground biomass (20%), and CO_2 fluxes (20%).

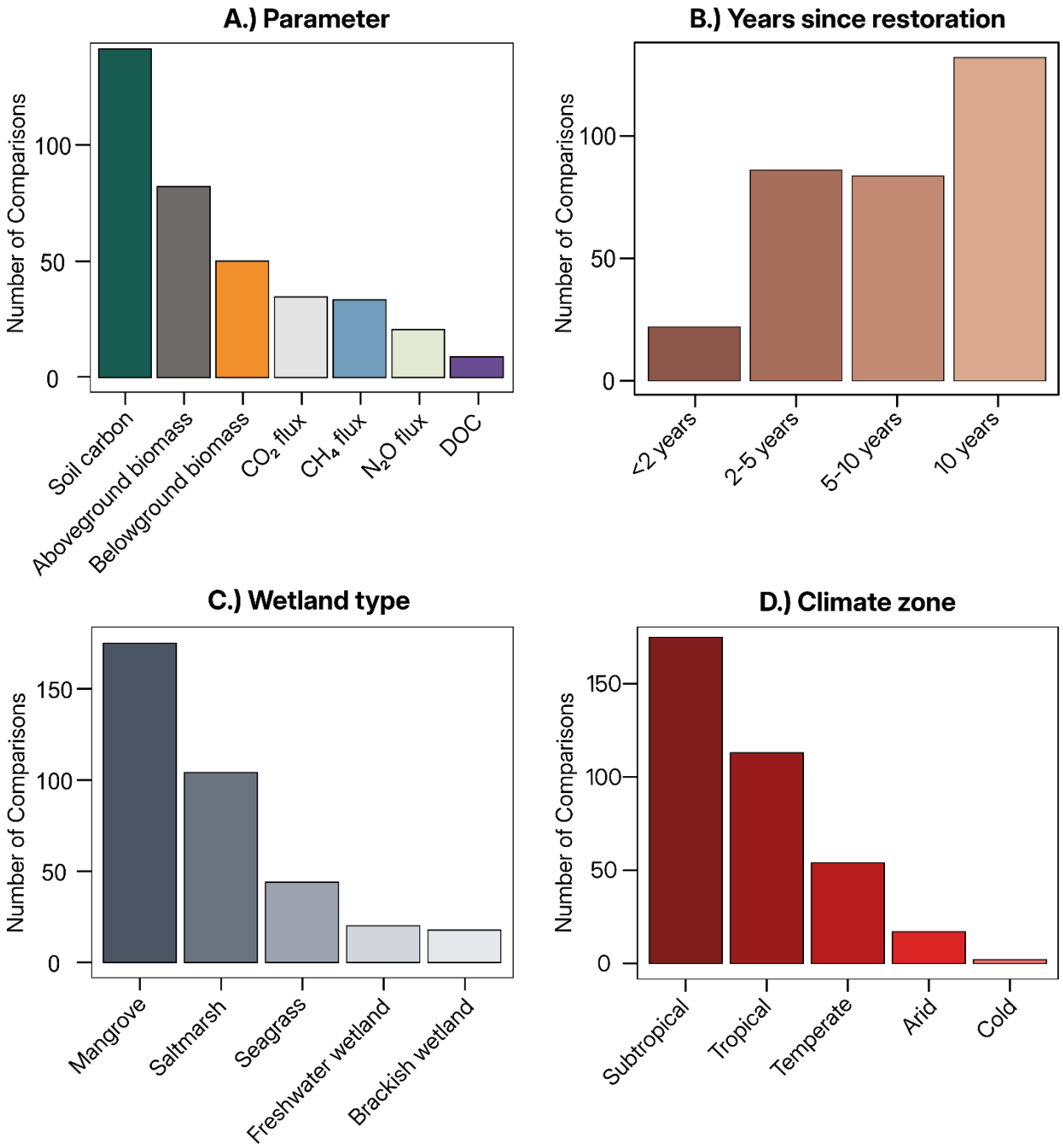

Figure 1: Overview of the available peer-reviewed research on coastal wetland restoration and its impact on carbon storage and GHG pathways. The four plots show the number of pairwise combinations of altered versus restored sites, for (A) different parameters, for (B) different time since restoration (here time between start of restoration and study), for (C) different wetland types and for (D) different climate zones.

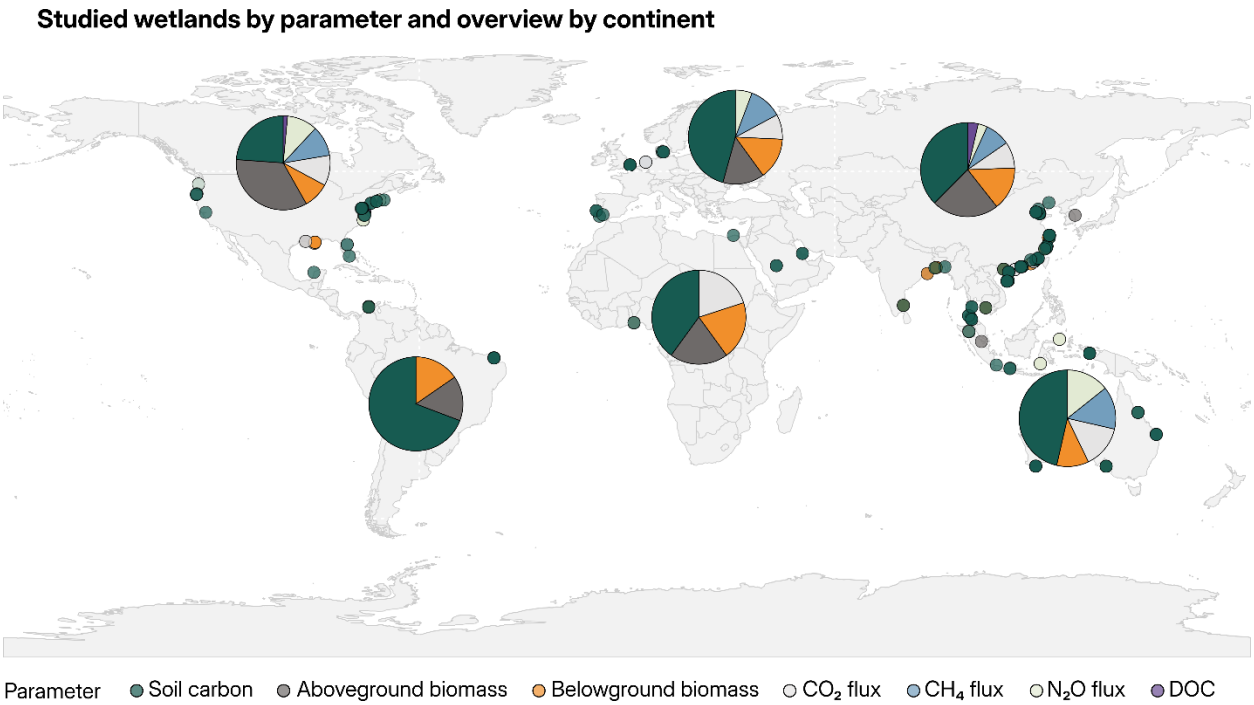

Figure 2: The map shows the location of the studied wetlands; pie charts indicate the proportion of each parameter to the total number of comparisons on each continent. Map lines delineate study areas and do not necessarily depict accepted national boundaries.

Among the restoration types covered by the selected studies, vegetation-related restoration (invasive alien species management, plantation of native vegetation) dominated, accounting for 196 comparisons in total, far more than any other restoration approach (Table 1). This approach was especially prevalent for cases of vegetation/habitat loss (56 comparisons), land use change (55), and exotic species (41). Hydrology restoration was primarily used to address land use change (33) and hydrological alterations (22) but appeared less often for other alteration types. Morphological restoration (i.e. activities focused on recreating or enhancing wetland physical features and habitats, such as channels, islands, or shoreline structures) was mainly implemented in response to hydrological alteration (16) and land use change (16), while passive restoration was also most frequently associated with land use change (20) and vegetation/habitat loss (13). Approaches focusing on water quality or soil were infrequent. Notably, alterations due to exotic species, morphology, and natural disasters were almost exclusively addressed through vegetation-related restoration, with 41, 4, and 14 comparisons, respectively.

Table 1: Combinations of alteration and restoration actions; entries give the number of comparisons per combination. For comparisons with several alterations or restoration actions, we selected the action presented as primary in the source paper.

		Restoration type					
		Vegetation related restoration	Hydrological restoration	Morphological restoration	Passive restoration	Water quality restoration	Soil related restoration
Alteration type	Exotic species	41	4	0	4	0	0
	Hydrological alteration	4	22	16	1	2	0
	Hydromorphological alteration	0	6	8	0	0	0
	Land use change	55	33	16	20	0	0
	Morphological alteration	4	0	0	0	0	0
	Natural disasters	14	0	1	0	0	0
	Vegetation/habitat loss	56	4	0	13	0	12
	Water quality/pollution	22	0	5	0	0	6

Effects of coastal wetland restoration

We found a significant mean effect (SMD) of coastal wetland restoration on three of the six tested parameters and on the variability (lnCVR) of two parameters (Figure 3). Looking at the SMD, we observed a significant positive effect (i.e., increase) of restoration on soil carbon (SMD = 0.86, p-value = 0.003, Table S5), aboveground biomass (3.74, < 0.001), and belowground biomass (1.27, 0.002). All three GHGs showed small but nonsignificant negative effects (i.e., reduction). Looking at the lnCVR, we found significantly less variability in aboveground (lnCVR = -0.67, p-value = 0.001, Table S5) and belowground (-0.33, 0.020) biomass, while the other parameters did not show a significant change.

Egger's regression test, which assesses funnel plot asymmetry as an indicator for small-study effects or publication bias, showed significant asymmetry (Table S6, Figure S3) for soil carbon (Egger Z = 2.70, p-value = 0.008), aboveground biomass (6.43, <0.001) and CH_4 fluxes (-3.07, 0.004). For the three significant parameters Rosenthal's fail-safe number (i.e. the number of unreported null studies required to overturn a significant result, assuming publication bias) indicated that overturning significance would require about 528 additional null studies (i.e., studies reporting no restoration effect) for soil carbon, 681 for aboveground biomass, and 77 for belowground biomass. These numbers are higher than the existing number of comparisons per parameter, indicating that a substantial number of null studies would be needed to overturn these results.

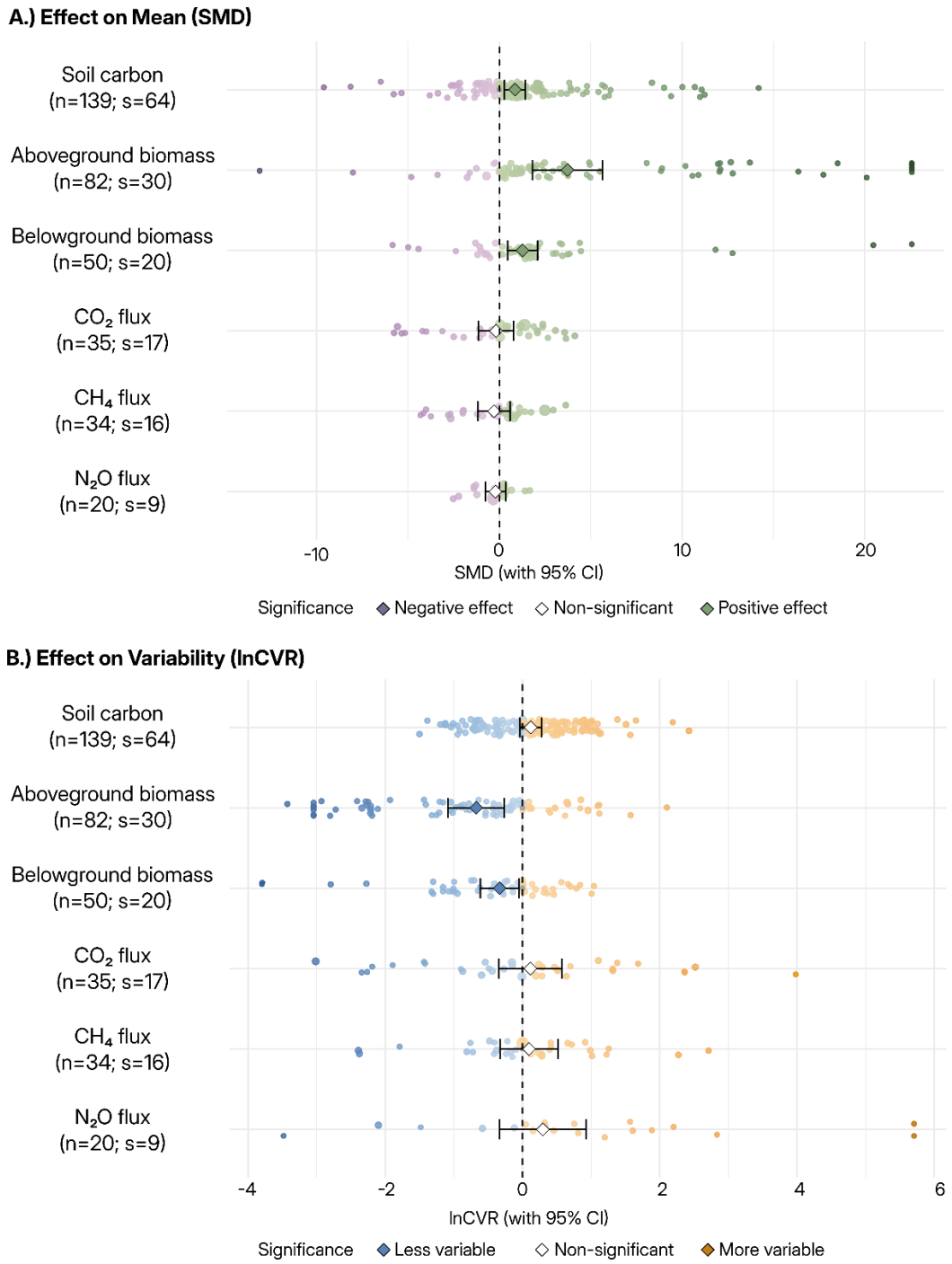

Figure 3: Overall effects of restoration across parameters on means and variability. Top panel: pooled SMDs with 95% CIs from multilevel random-effects models (REML; random intercepts for study and substudy). Bottom panel: pooled log coefficient of variation ratios (lnCVR) with 95% CIs from the same model structure. Points show each individual comparison, colored by effect direction and magnitude. Labels include the number of comparisons (n) and studies (s) per parameter. The vertical dashed lines at 0 marks no change in either direction. Positive SMD indicates a higher mean in the focal group relative to its comparator; positive lnCVR indicates greater variability.

Differences by coastal wetland type

The meta-regression analyses to test restoration effects by coastal wetland type showed no significant between-type differences for any parameter (Table S7). Nevertheless, given that coverage was uneven across wetland types, and often based on low sample sizes, we still show wetland-specific pooled SMDs with 95% CIs to visualize patterns (Figure 4). Aboveground biomass increased across all wetland types; however, the increase was significant only for mangroves. Belowground biomass was significantly positively affected in mangroves and saltmarshes, while brackish wetlands showed a huge variation in effects and seagrasses showed no clear trend. For freshwater wetlands, no data was available. For soil carbon, a significantly positive impact was detected in brackish wetlands and mangroves, while in the other types of wetlands no clear effects were found. For CO_2, we found a high variation of the results around zero in freshwater wetlands, mangroves and saltmarshes, while data for brackish wetlands and seagrasses were not available. CH_4 fluxes showed a positive effect only in freshwater wetlands, while mangroves and saltmarshes showed no clear trend; seagrasses were represented by one comparison and for brackish wetlands, data were lacking. N_2O fluxes seemed to decrease with restoration in freshwater wetlands, however these results represented only three comparisons from one study. N_2O fluxes from mangroves and saltmarshes showed a high variation and no trend, while for seagrasses (one comparison) and brackish wetlands (no comparison), we lacked representative data.

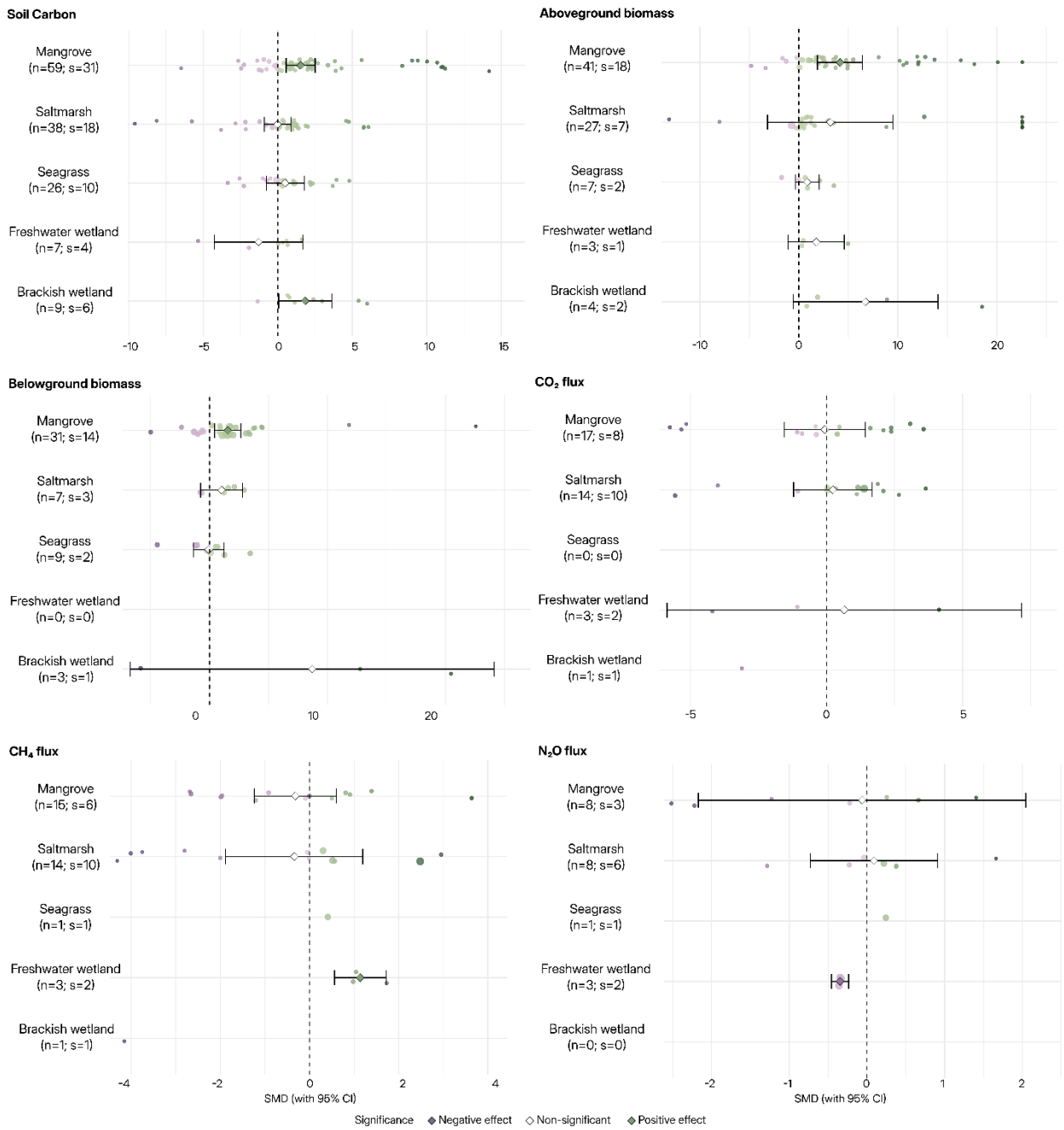

Figure 4: Parameter-specific effects by coastal wetland type. Diamonds show pooled standardized mean differences (SMD) within each wetland type with 95% CIs from multilevel random-effects models (REML; random intercepts for study and substudy). Points are individual comparisons, colored by effect direction and magnitude. Labels include the number of comparisons and unique studies per wetland type. The vertical dashed line marks no effect. Pooled estimates are shown only when a wetland type has at least two comparisons.

Discussion

Evidence Gaps and Research Biases in Coastal Wetland Restoration Research

Our literature analysis reveals substantial biases and research gaps in the global coastal wetland restoration literature in relation to carbon storage and GHG pathways. Over half of the restoration comparisons in our meta-analysis originate from Asia, predominantly from mangrove ecosystems. In line with this geographic concentration, vegetation-based restoration actions, such as mangrove or marsh planting following habitat loss due to land-use change (Gerona-Daga & Salmo, 2022; Lovelock et al., 2022), are far more frequently represented than hydrological or geomorphological interventions. This reflects a broader bias towards mangrove systems in the available evidence base. At the same time, important gaps in geographic coverage and restoration strategy remain. Africa is severely underrepresented, with only two studies reporting data and also coastal wetlands located in temperate and cold climates are underrepresented. Similar spatial biases have previously been observed for other global blue carbon research unrelated to restoration (Mcleod et al., 2011). Furthermore, hydrological and geomorphological restoration measures (such as reconnecting tidal flow or restoring natural elevation and substrate) are much less represented in the published studies (Simenstad et al., 2006), partly because they often involve transforming non-wetland states (e.g., drained agricultural land) back into functioning wetlands. In such cases, pre-restoration sites may not be sampled or considered suitable as reference wetlands, meaning that these projects do not yield paired comparisons that meet our selection criteria. While these interventions are critical for recovering ecosystem functions, their underrepresentation reduces opportunities to quantify their carbon benefits. Together, these geographic, climatic, and methodological imbalances may limit the transferability of existing knowledge to regions like Europe, where restoration policies such as the EU Nature Restoration Regulation (European Parliament and the Council of the European Union, 2024), increasingly demand evidence-based planning tailored to local conditions.

Finally, we observed a clear imbalance in the reported carbon and GHG parameters: most studies focused on carbon stocks (soil carbon and biomass) while far fewer quantified GHG fluxes or DOC concentrations. Carbon stock measurements (e.g., soil carbon content or tree biomass) are relatively inexpensive and straightforward to perform compared to measuring GHG fluxes. These GHG assessments require specialized expertise and costly instruments, along with complex, spatially intensive sampling designs (Zaki & Abdul-Aziz, 2022), probably contributing to a lack of studies on GHG emissions in restoration contexts. The lack of representative studies on DOC dynamics represents an important research gap for comprehensive whole-system carbon accounting, as quantifying DOC at the scale of restoration projects remains methodologically challenging (Bauer et al., 2013).

Our synthesis is based solely on results reported in peer-reviewed journals (in English) that included comparisons between altered (degraded) and restored sites. This criterion excludes many restoration efforts that are not monitored for carbon outcomes or are not published internationally.

Indeed, restoration is often motivated by goals like enhancing biodiversity or habitat quality and thus may lack carbon cycle and GHG flux monitoring (Bertolini & da Mosto, 2021). Further data may appear in grey literature (e.g., technical or local reports) or in non-English publications, which were excluded from our global review and may contribute to bias (Hannah et al., 2024; Konno et al., 2020). Additionally, many studies adopt designs that do not directly compare restored and altered sites, such as temporal monitoring of areas or comparisons with reference natural wetlands, especially when pre-restoration conditions differ strongly (e.g., cornfields or shrimp ponds). While these approaches yield valuable insights, they do not isolate restoration impacts on carbon dynamics as effectively (Mahlum et al., 2018; Smokorowski & Randall, 2017). In summary, substantial evidence gaps remain, and our findings should be interpreted in light of these limitations.

Restoration Effects on Carbon Pools and GHG Fluxes

Our meta-analysis indicates that coastal wetland restoration generally enhances carbon pools with restored sites showing higher aboveground biomass, belowground biomass, and soil carbon than degraded counterparts. These results underline the importance of restored wetlands as carbon sinks and are consistent with established mechanisms of wetland carbon sequestration (Guo et al., 2025; L. Li et al., 2024; Taillardat et al., 2020). Vegetation recovery increases CO_2 fixation with a positive effect on living biomass, while root growth and litter fall introduce new organic inputs into the soil (Dontis et al., 2020). In mangroves especially, restored vegetation contribute substantial woody biomass and dense belowground root networks that stabilize sediments and promote the accumulation and retention of soil carbon (Rovai et al., 2021; Twilley et al., 2017). Hydrological restoration, including tidal reconnection, creates waterlogged soil conditions that suppress aerobic decomposition and favour soil organic matter preservation (Lorenz & Lal, 2018), although CH_4 fluxes may increase under certain redox conditions.

We also observed reduced variability in above- and belowground biomass following restoration, whereas variability in soil carbon remained high, likely reflecting differences in sediment supply, inundation regimes, rates of elevation change, and the initial site condition (Brown & Norris, 2018; Shao et al., 2022). Consequently, restoration of heavily altered, carbon-poor sites (e.g., an agricultural field or aquaculture pond) may lead to rapid carbon accumulation once tidal exchange and plant growth resume (Eagle et al., 2022; Gulliver et al., 2020), whereas gains are typically more gradual in degraded wetlands that still retain some ecological function (Burden et al., 2019; Suir et al., 2019).

In contrast to the clear increases in carbon stocks, we did not find consistent, statistically significant changes in GHG fluxes linked to restoration. The overall effect sizes for CO_2, CH_4, and N_2O fluxes were small and not statistically distinguishable from zero. However, this does not necessarily mean that restoration has no impact on gas emissions, but rather, that the effects appear to be context-dependent and our ability to detect them was limited by the relatively small number

of studies providing only snapshots in time and space. We found no uniform trend in net CO_2 exchange with restoration. Many studies in our analysis measured only soil or sediment CO_2 flux or exchange at the water surface and did not account for the uptake of CO_2 by the recovering plant community (Fitch et al., 2022; Ouyang et al., 2024). As a result, any increase in CO_2 sequestration via plant photosynthesis is often missing from those flux measurements. This likely led to an underestimation of restoration-driven CO_2 uptake. In short, while vegetation data show increased carbon storage, the sparse CO_2 flux data (excluding plant uptake) remained near neutral on average, so a positive CO_2 balance from restoration is plausible but not well-captured by available studies.

We also did not detect a consistent effect of restoration on CH_4 or N_2O emissions. Some sites reported increases, others decreases, and overall, the average effect for both GHGs was not significant. This variability likely reflects differences in site conditions, restoration methods, and monitoring periods. For CH_4, the responses can be highly transient, with short-term emission pulses possible immediately after re-wetting (Boonman et al., 2023; Wilson et al., 2009). There is evidence in the literature that restoring natural wetland conditions (e.g., tidal inundation or hydrologic connectivity) can reduce N_2O emissions by promoting more complete denitrification and less nitrification (Cadier et al., 2023; Yang et al., 2020). It is possible that N_2O responses are highly site-specific, depending on factors like excess nutrient availability and soil redox conditions during restoration. Given the small number of paired comparisons for all three GHGs, the current evidence is insufficient to draw general conclusions, underscoring the need for more systematic and long-term monitoring of CO_2, CH_4 and N_2O fluxes in restored coastal wetlands.

Variation among Coastal Wetland Types and Contexts

Our meta-regressions did not detect statistically significant differences in restoration outcomes among coastal wetland types in contrast to our hypothesis. However, this likely reflects the limited and unbalanced evidence base rather than true ecological similarity, as the dataset is strongly dominated by mangrove studies, while other wetland types are represented by only a few comparisons. Consequently, the following discussion focuses primarily on mangrove restoration, where the available data allow for more robust interpretation of carbon responses. Evidence from other coastal wetland types suggests broadly comparable recovery trajectories but remains too sparse to support detailed cross-system comparisons.

Mangrove restoration tends to show the largest and most consistent gains in carbon stocks. Recovering mangroves rapidly rebuilds aboveground biomass (trunks, branches, and foliage) and extensive belowground root networks, which together contribute to substantial carbon sequestration (Alongi, 2014). Mangrove soils also accumulate carbon as tidal flows resume and trap sediments. Notably, some of the longest-running restoration sites in our dataset are mangroves (with up to 114 years since restoration), demonstrating that given time, restored mangrove ecosystems can approach or even exceed the carbon stock of natural mangroves (Rogers et al.,

2019). The high carbon recovery in mangroves can likely be attributed to their tree stature and high productivity, which create a large carbon input, and their ability to engineer the environment (e.g., by stabilizing sediment and building peat; Doughty et al., 2016).

Across coastal wetland types other than mangroves, restoration generally promoted vegetation recovery and carbon accumulation, although responses were more variable and evidence was limited. In saltmarshes, restoration consistently increased plant biomass (Bartolucci & Fulweiler, 2024; Flynn et al., 1999), whereas soil carbon responses were variable and appeared to depend on the pre-restoration state (e.g., a fully drained former marsh versus a partially degraded marsh), the specifics of restoration methods (e.g., whether sediments were added, or natural recolonization was allowed versus actively planted), type of saltmarsh (mineral, or organic saltmarsh) and sediment availability which leads to carbon import (Fettrow et al., 2024).

Seagrass restoration showed promising but still sparse evidence for soil carbon gains (S. Liu et al., 2023; Tanner et al., 2021), likely driven by sediment trapping and organic matter inputs from recovering meadows (Johannessen, 2022).

Brackish coastal wetlands showed patterns of carbon recovery broadly similar to saltmarshes (Doroski et al., 2019). with the clearest gains in soil carbon, beside mangroves (Shiau et al., 2016), and inaboveground biomass with restoration (typically reestablishment of salt-tolerant reeds or grasses.

Freshwater coastal wetlands were least represented; available studies indicate increases in plant biomass and soil carbon following rewetting, but also highlight the potential for higher CH_4 emissions post-restoration, since re-flooding organic-rich soils creates ideal conditions for methanogenesis (Poffenbarger et al., 2011).

Overall, the strong bias in the scientific literature toward a few ecosystem types (notably mangroves) and parameters means that while we can draw robust general conclusions about the impacts of coastal wetland restoration, our ability to attribute these benefits to specific wetland types or environmental contexts is limited. In reality, differences in climate, geomorphology, and land-use history are likely to modulate restoration outcomes. For instance, a subtropical mangrove restoration may not be directly comparable to a temperate saltmarsh restoration, colder climates typically have slower organic matter decomposition rates and lower baseline CH_4 emissions, which could influence the net carbon balance (Rogers et al., 2019). Similarly, the type of degradation (e.g., eutrophication, drainage, vegetation removal) and the specific restoration methods used (e.g., natural regeneration versus active planting; partial versus full tidal reconnection) will result in different carbon and GHG trajectories. To improve predictive power, more data are needed across a range of climatic zones, nutrient conditions, and restoration approaches. Such data would enable us to disentangle how factors like climate zone, trophic status, and initial site condition influence GHG fluxes and carbon sequestration in restored wetlands, allowing for more tailored and accurate expectations for restoration projects in various contexts.

Implications for Policy and Practice

Our findings strengthen the case that coastal wetland restoration is a valuable nature-based solution for climate change mitigation. Restoration consistently increased carbon stocks in biomass and soils without a corresponding increase in GHG emissions, meaning restored wetlands tend to function as net carbon sinks (Temmink et al., 2022). However, the evidentiary foundation remains uneven across regions and ecosystem types. Policymakers may thus consider investing in capacity-building, long-term monitoring, and region-specific research to reduce uncertainty. For on-the-ground restoration practitioners, our meta-analysis underlines that projects focusing on reinstating native vegetation and natural hydrological processes, particularly tidal fluxes are likely to yield sustained carbon benefits and managers can confidently include carbon sequestration as an expected co-benefit for their restoration projects. Actions like mangrove planting, saltmarsh re-vegetation, seagrass meadow restoration, or tidal flow reconnection should be pursued not only for their biodiversity and coastal protection merits but also for their climate mitigation value (Doughty et al., 2016). Indeed, some large-scale coastal restoration efforts (particularly mangrove restorations) have already been enrolled in carbon offset markets and national climate strategies, reflecting the growing recognition of these dual benefits (Taillardat et al., 2018; Zeng et al., 2021). Although we did not find a net increase in CH_4 or N_2O emissions across studies in coastal wetlands, it remains important for restoration projects to monitor these GHGs to validate climate benefits on a case-by-case basis. By incorporating GHG management into restoration planning (through careful hydrological control and vegetation choices), practitioners can ensure that climate benefits are maximized, and any unintended emissions are kept in check.

Conclusions

Coastal wetland restoration emerges from this study as a promising climate mitigation strategy that rebuilds ecosystem carbon stocks. Our meta-analysis confirms that restoring degraded mangroves, saltmarshes, seagrass meadows, and other coastal wetlands leads to significant increases in plant biomass and soil carbon storage relative to the unrestored (altered) condition. Crucially, these carbon gains were achieved without a concurrent increase in GHG emissions. We found no overall surge in CH_4 or N_2O after restoration. This balance of outcomes underscores the climate value of coastal ecosystem restoration and solidifies its status as an effective nature-based solution for mitigating climate change.

However, our analysis also highlights that the current evidence base is not yet comprehensive. It is heavily weighted towards certain regions (e.g., subtropical Asia) and ecosystem types (mangroves in particular), while data remain scarce for other regions and ecosystem types. It also lacks sufficient information on GHG fluxes, concentrations and related carbon export via DOC, and long-term monitoring needed to constrain variability. Therefore, we urge caution in generalizing the climate mitigation abatement potential globally and emphasize the need for more extensive monitoring and research.

Data and Code Availability

All data and code supporting this study are openly available on Zenodo at https://doi.org/10.5281/zenodo.20110894.

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