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 66 studies and 257 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 methane or nitrous oxide, 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). Carbon sequestration occurs via atmospheric carbon dioxide (CO_2) fixation by photosynthetic organisms like plants, and long-term storage of organic carbon (OC) in aboveground and belowground biomass and later soils and sediments (Chmura et al., 2003; 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 is offset by the release of CO_2 released 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). Sea-level rise, when combined with infrastructure that restricts landward migration, which further limits their capacity to adapt (i.e., coastal squeeze; Schuerch et al., 2018; Zhi et al., 2022). Furthermore, the remaining coastal wetlands are under pressure from 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), with additional impacts on other ecosystem services (UNESCO, 2019).

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 mitigation and adaptation. Irrespective of the primary goals of restoration, these activities also impact the carbon cycle. 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, 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 that vary in intensity, from passive approaches that rely on natural regeneration after removal of the pressures causing the degradation of the wetland, to active measures that directly manipulate environmental conditions within the wetland itself or its catchment. These interventions affect the carbon cycle through four key drivers: sediment, hydrology, salinity, and vegetation. First, sediment-based approaches (augmentation, engineered deposition and diversions) help restore elevation, reduce erosion, and sustain wetland function (Barbier et al., 2011; Elsey-Quirk et al., 2019), boosting plant productivity and thereby carbon storage in plant biomass, 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 measures such as levee removal or tidal inlet reopening reestablish natural tidal exchange and water level, restoring sediment transport and improving habitat connectivity (P. Williams, 2001). Raising water level leads to waterlogged soils that increase carbon burial and suppress 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 initially (Maher et al., 2013; Neubauer & Anderson, 2003). Third, reintroducing tidal flows, managing seawater intrusion, and controlling freshwater inflows reestablishes natural salinity levels that affect vegetation composition and microbial processes in the soil. In saline systems, for example, sulfate reduction outcompetes methanogenesis, thereby suppressing CH_4 production and reducing the likelihood that methane offsets CO_2 uptake, a common outcome of inland

wetland restoration efforts (Camacho et al., 2017; Kroeger et al., 2017). Finally, active vegetation planting, grazing exclusion, and invasive species removal support natural vegetation recovery, stabilizing sediments, enhancing organic matter inputs, and promoting belowground carbon storage (Gedan et al., 2011; Maxwell et al., 2024; Zedler & Kercher, 2005). Vegetation recovery increases CO_2 uptake, as photosynthesis and biomass accumulation enhance carbon storage (Eagle et al., 2022). In brackish or freshwater environments, rapid vegetation growth can also increase CH_4 emissions by supplying labile organic matter to methanogens and promoting anoxic soil conditions, while N_2O emissions often remain relatively low when plant uptake and reducing soil conditions limit the availability if substrates for N_2O production (Comer-Warner et al., 2022). Vegetation also influences DOC, as plants release distinctive dissolved organic matter through litter and roots (Tzortziou et al., 2008).

Although coastal wetland restoration is widely implemented, its role in climate change mitigation remains one of the least quantified benefits, reflecting a global research priority in blue carbon science (Macreadie et al., 2019). The existing literature is dominated by studies focused on a few types of coastal wetlands (mangroves, saltmarshes), which do not adequately represent the full diversity of coastal wetlands globally (Lu et al., 2017; Taillardat et al., 2020). Hence, the substantial variability among coastal wetland types, with system-specific characteristics influencing carbon dynamics and restoration trajectories, must be addressed to produce robust, policy-relevant estimates of restoration-driven climate change mitigation potential. Furthermore, restoration outcomes are often measured through comparisons of carbon stored in soil and biomass between restored sites and undisturbed reference wetlands, a method that reveals ecological recovery but not necessarily a net climate benefit. However, policy-relevant assessments demand comparisons between restored and degraded/impacted sites to determine the increase in carbon storage attributable to restoration and should integrate GHG fluxes and dissolved organic carbon (DOC) alongside carbon stock inventories to more accurately assess climate outcomes (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, dissolved organic carbon (DOC), and greenhouse gas fluxes (CO_2, CH_4, N_2O). Our objectives are threefold: (1) to identify biases and gaps in the existing literature, including studied parameters, geographical distribution, and coastal wetland types; (2) to determine the extent to which restoration enhances carbon storage and modulates GHG fluxes and DOC concentration relative to degraded conditions; and (3) to compare outcomes among different coastal wetland types. We hypothesized that (1) most studies have emphasized carbon stocks (soil and biomass) over GHG fluxes and DOC, and focused on a narrow range of ecosystems; (2) an overall increase in carbon storage (in biomass and soils) is linked to restoration, while responses of GHG fluxes and DOC may be more variable; and (3) that restoration outcomes would differ by wetland type. By directly comparing restored sites to impacted sites, an approach more aligned with policy needs, our analysis provides a critical synthesis of available information needed to quantify the benefits and potential disadvantages of coastal wetland restoration, and to help practitioners and decision-

makers to understand where and how coastal restoration can best contribute to climate change mitigation and inform future funding and restoration 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 in the Web of Science Core Collection and Scopus in January 2024 and updated on 7 October 2024, including newly published studies. The following search query was used on the Web of Science to search titles, abstracts, and keywords:

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

For SCOPUS the search query was slightly adjusted to fit its specific terminology (see supplementary methods). The searches retrieved 4,071 records from Web of Science and 4,179 from Scopus. After duplicates were removed using dplyr (R 4.3.2; R Core Team, 2025; Wickham et al., 2020), 6,448 unique records remained.

Study selection and data extraction

The 6,448 records were screened for eligibility against predefined inclusion and exclusion criteria (Table S1). Studies were considered eligible when they: (i) were peer-reviewed field studies in coastal wetlands, (ii) directly contrasted altered with restored conditions using Before-After, Control-Impact, Before-After-Control-Impact designs, and (iii) reported sufficient statistics to compute effect sizes (mean, number of replicates, standard deviation (SD) or convertible dispersion metrics (standard error (SE), confidence interval (CI), interquartile range, range)) for at least one target variable (carbon pools or GHG fluxes). We excluded laboratory or mesocosm work, modelling-only studies, reviews and meta-analyses, comparisons of restored sites with natural reference sites only, non-coastal ecosystems, and cases where variance or sample sizes could not be derived.

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

Data from the 66 eligible papers were extracted into a single, structured spreadsheet. For every pairwise comparison (altered versus restored site; one parameter), we captured core metadata (study overview, alteration and restoration context), the parameter of interest (mean, SD, number of replicates, units) and methodological notes (analytic method, used instruments). Various types of data formats were summarized to result in one value per parameter, per restoration and alteration site, following the information available in the publication. Out of the 66 eligible papers, 257 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.

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). Positive Hedges’ g indicates that the restored site has a higher mean than the altered site, while 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 comparisons reported (n = 1).

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). 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 were generated from these uni-level models. Egger's regression ("regtest" in *metafor*) was applied only when $k \geq 10$. We also computed Rosenthal's fail-safe N ("fsn" in *metafor*) when $k \geq 3$ as a complementary diagnostic. Forest plots display pooled estimates with 95% confidence intervals and jittered study effects sized by inverse sampling variance. 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 257 pairwise comparisons (restored versus altered) from 66 studies. Most studies provided data on soil carbon (Figure 1A, 97 pairwise comparisons) and on aboveground (54) and belowground biomass (45). GHG fluxes were studied less frequently, with 22 comparisons for CO_2, 21 for CH_4, and 17 for N_2O. Only one comparison covered DOC, so it was excluded from all subsequent analyses. This global dataset covers various study designs where the time between the end of the restoration and sampling ranged from 1 to 114 years (Figure 1B). In 15 comparisons, the restoration took place within 2 years prior to the study, 50 comparisons were made between 2 and 5 years after restoration, 51 comparisons between 5 and 10 years, and 107 comparisons more than 10 years after restoration, while for 34 comparisons, the time of restoration was not explicitly reported. Looking at the types of coastal wetlands studied (Figure 1C), we saw that mangroves (132), saltmarshes (60), and seagrass meadows (34) were most commonly studied, while only 16 comparisons were found for freshwater wetlands and 15 for brackish wetlands. Looking at the climate zone where the research was conducted, we found most comparisons in subtropical (117) and tropical (80) sites, while 52 comparisons were found in temperate, 5 in arid, and only 1 in cold sites (Figure 1D).

Considering the geographical distribution of the studies (Figure 2), we found 140 pairwise comparisons from Asia, followed by North America (49 comparisons), Europe (35), Oceania (19), South America (13), and Africa (1). The variables measured also differed across continents (Figure 2, Figure S2). In Asia, most of the research reported on soil carbon (35% of the comparisons), aboveground (26%) and belowground (21%) biomass, and only smaller proportions investigated GHG fluxes (CO_2: 7%, CH_4: 6%, N_2O: 5%). 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 (22%), belowground biomass (12%), CO_2 (14%), CH_4 (12%), and N_2O fluxes (12%). In Oceania, over half of the studies addressed soil carbon (53%), while 16% focused on belowground biomass, and 10% 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 sole study reported on soil carbon (100%).

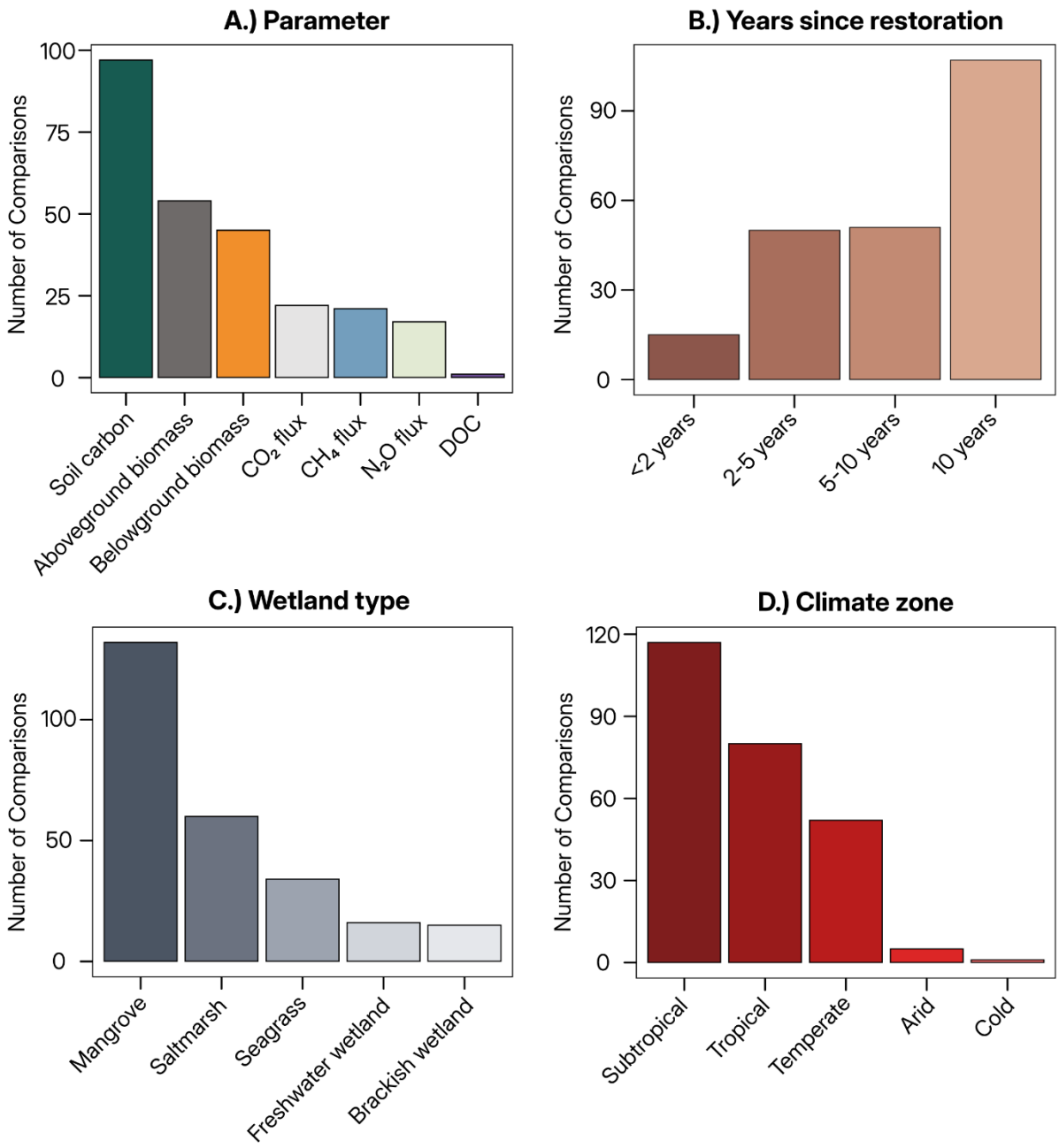

Figure 1: Overview of the available peer-reviewed research on coastal wetland restoration and its impact on carbon storage and GHG pathways. The four plots show the number of pairwise combinations of altered versus restored sites, for (A) different parameters, for (B) different time since restoration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

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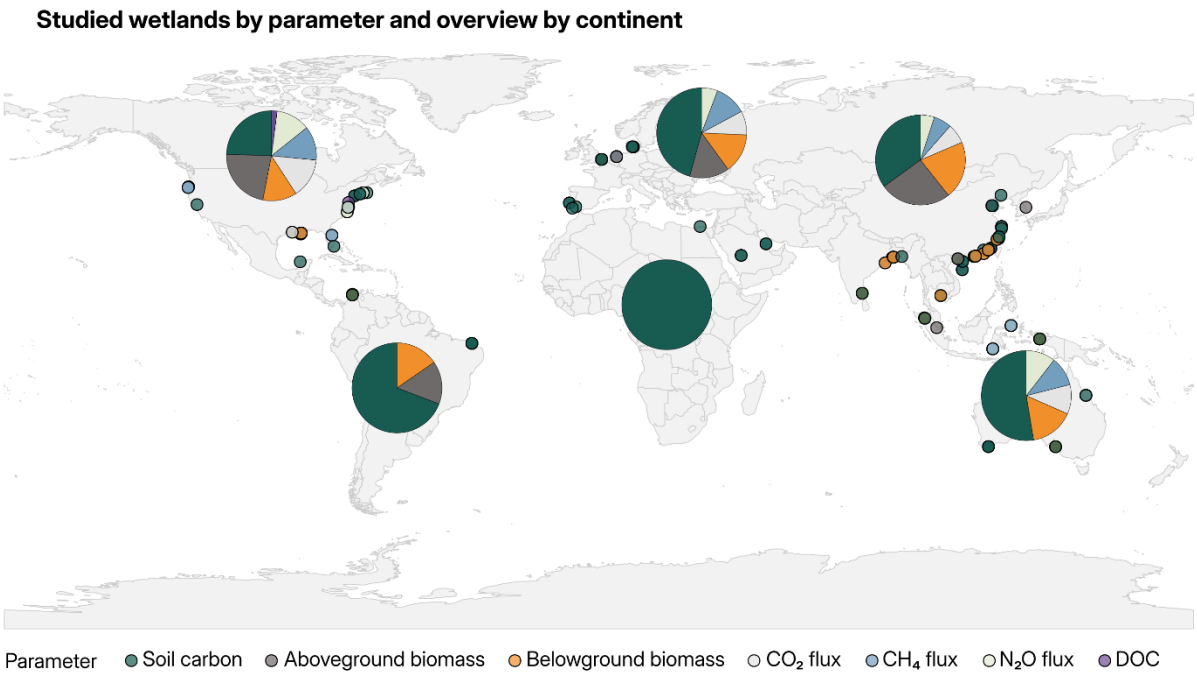

Figure 2: The map shows the location of the studied wetlands; pie charts indicate the proportion of each parameter to the total number of comparisons on each continent. Map lines delineate study areas and do not necessarily depict accepted national boundaries.

Among the restoration types covered by the selected studies, vegetation-related restoration (invasive alien species management, plantation of native vegetation) dominated, accounting for 123 comparisons in total, far more than any other restoration approach (Table 1). This approach was especially prevalent for cases of land use change (41 comparisons), vegetation/habitat loss (20), and loss in water quality/pollution (22). Hydrology restoration was primarily used to address land use change (24) and hydrological alterations (22) but appeared less often for other alteration types. Morphology restoration was mainly implemented in response to hydromorphological alteration (8) and land use change (16), while passive restoration was also most frequently associated with land use change (16) and vegetation/habitat loss (13). Approaches focusing on water quality or soil were infrequent, rarely exceeding 6 comparisons for any alteration type. Notably, alterations due to exotic species, morphology, and natural disasters were almost exclusively addressed through vegetation-related restoration, with 19, 3, 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	Hydrology	Morphology	Passive restoration	Water quality	Related to soil
Alteration type	Exotic species	19	0	0	0	0	0
	Hydrology	4	22	4	1	2	0
	Hydromorphology	0	4	8	0	0	0
	Land use change	41	24	16	16	0	0
	Morphology	3	0	0	0	0	0
	Natural disasters	14	0	1	0	0	0
	Vegetation/habitat loss	20	0	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 = 0.78, p-value = 0.006, Table S2), aboveground biomass (3.09, < 0.001), and belowground biomass *(*1.18, 0.001). All three GHGs showed small but nonsignificant negative effects (i.e., reduction). Looking at the lnCVR, we found significantly less variability in aboveground (lnCVR = -0.57, p-value = 0.002, Table S2) and belowground (-0.26, 0.02) biomass, while the other parameters did not show a significant change.

Egger's regression test showed significant funnel plot asymmetry (Table S3, Figure S3) for soil carbon (Egger Z = 2.34, p-value = 0.021) and aboveground biomass (5.39, <0.001), suggesting potential small-study or publication bias. Rosenthal's fail-safe number indicated that overturning significance would require about 289 additional null studies (i.e., studies reporting no restoration effect) for soil carbon, 354 for aboveground biomass, and 69 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.

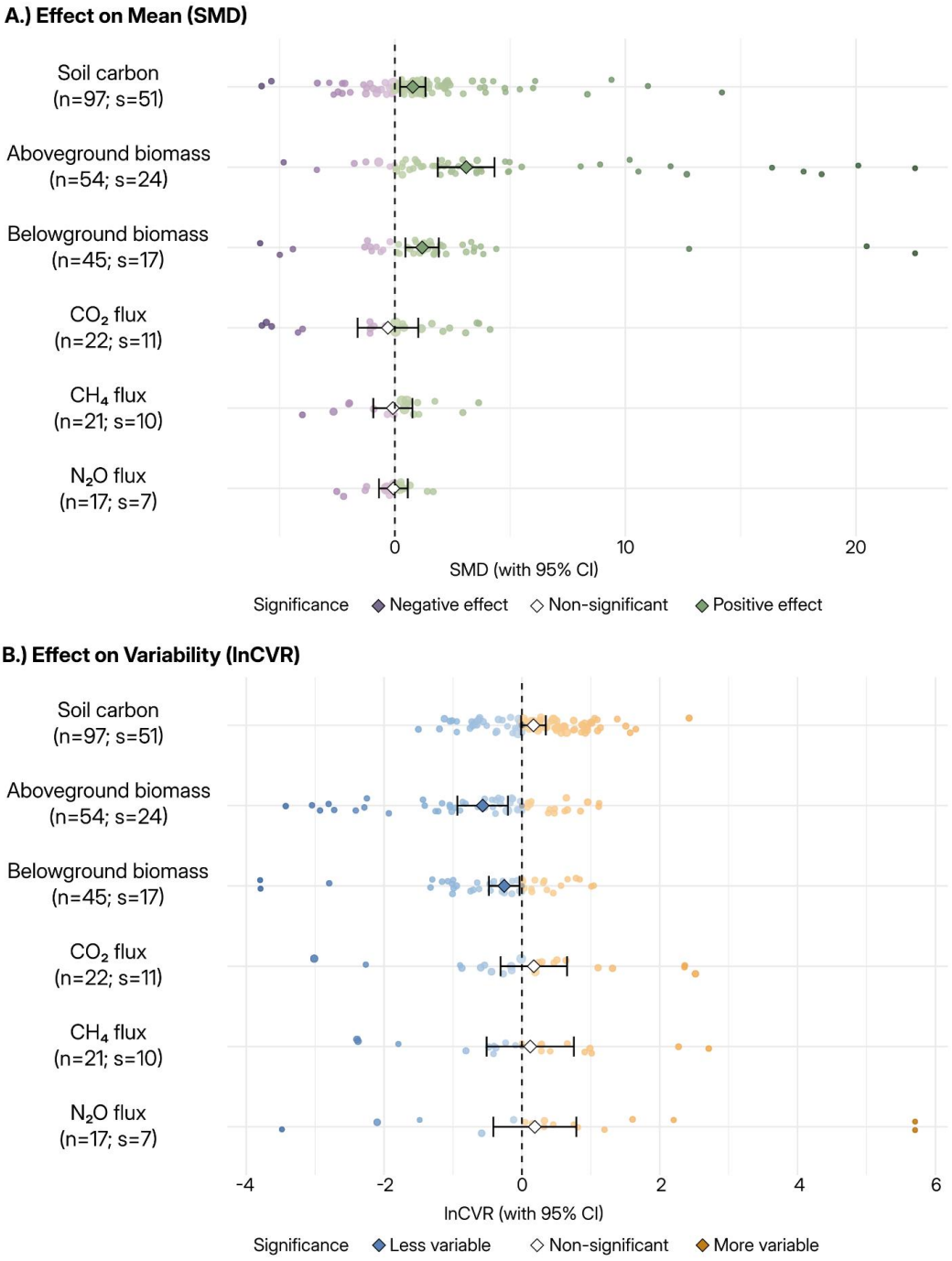

Figure 3: Overall effects of restoration across parameters on means and variability. Top panel: pooled SMDs with 95% CIs from multilevel random-effects models (REML; random intercepts for study and substudy). Bottom panel: pooled log coefficient of variation ratios (lnCVR) with 95% CIs from the same 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 neither 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

To assess whether the effects of restoration vary by wetland type, we conducted meta-regression analyses with coastal wetland type as a categorical moderator for each studied parameter (Table 2). We found no significant between-type differences for any parameter. 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 be decreasing 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.

Table 2: P-values from the Wald χ^2 test (QM) for the wetland-type moderator in parameter-specific multilevel random-effects models (REML, rma.mv, random intercepts for study and substudy).

Parameter	Wetland Type p-value
Aboveground biomass	0.800
Belowground biomass	0.277
Soil carbon	0.194
CH_4 fluxes	0.600
N_2O fluxes	0.967
CO_2 fluxes	0.996

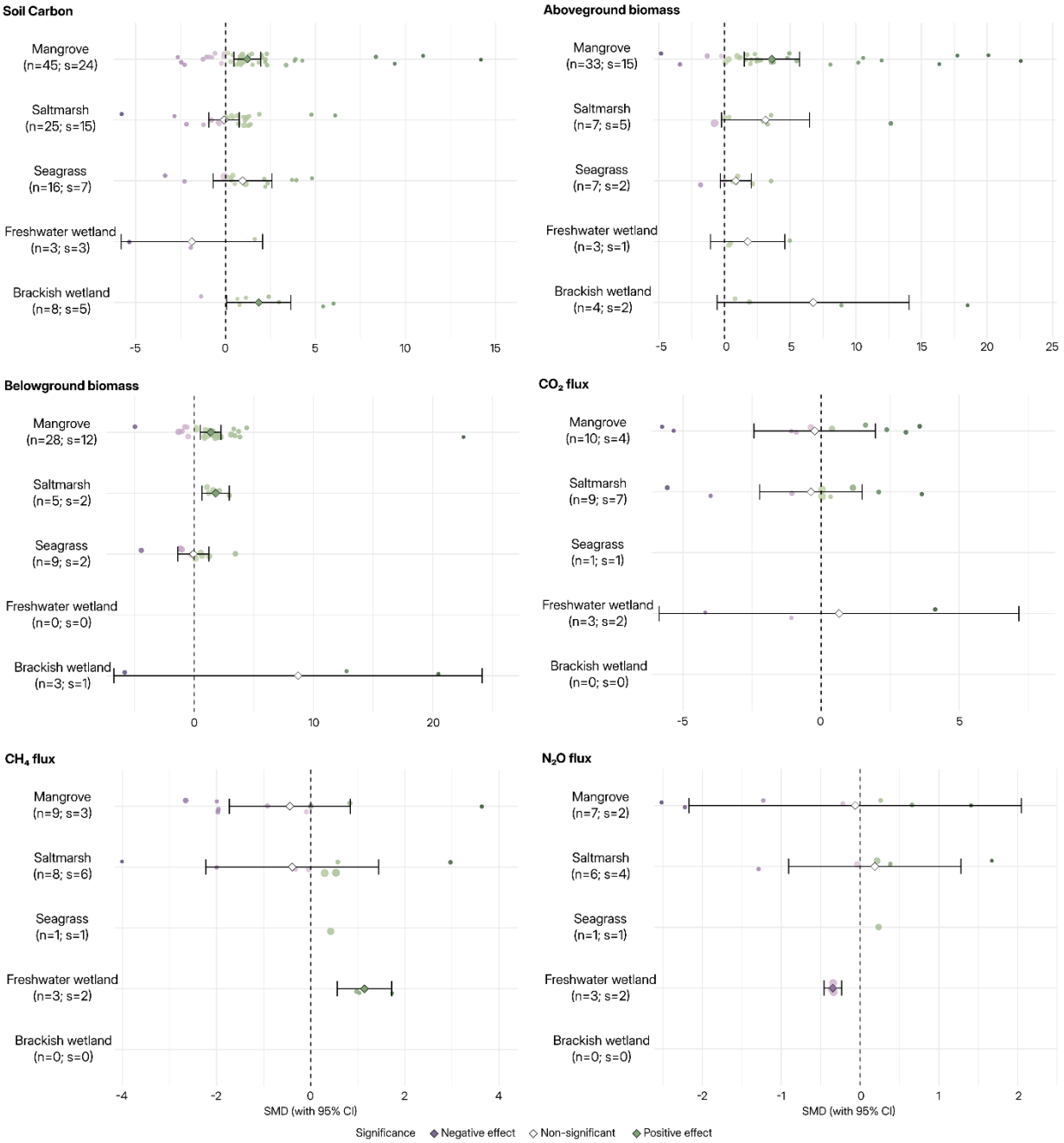

Figure 4: Parameter-specific effects by coastal wetland type. Diamonds show pooled standardized mean differences (SMD) within each wetland type with 95% CIs from multilevel random-effects models (REML; random intercepts for study and substudy). Points are individual comparisons, colored by effect direction 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 in mangrove ecosystems), whereas Africa is severely underrepresented, with only one comparison. Tropical and subtropical climates dominate the dataset, leaving coastal wetlands located in temperate and cold climates underrepresented. Similar spatial biases have previously been observed for other global blue carbon research unrelated to restoration (Mcleod et al., 2011). This geographic and climatic imbalance may limit the transferability of the 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.

We observed a clear imbalance in the reported carbon and GHG parameters: most studies focus on carbon stocks (soil carbon and biomass) while far fewer quantify 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 is another important research gap. Mangroves and other coastal wetlands can export DOC into the ocean or trap DOC by reducing flow velocity, which can significantly affect the net carbon balance of a system (Dittmar et al., 2006; Maher et al., 2013). Ignoring DOC dynamics alongside carbon stocks and GHG fluxes neglects a critical component of comprehensive whole-system carbon accounting (Bauer et al., 2013).

With regard to our selected carbon and GHG parameters, vegetation-based restoration actions (e.g., planting mangroves or marsh grasses) dominate, particularly in cases where restoration aims to reestablish habitats lost through land-use changes. For example, mangrove replantation is a common technique after mangrove forest loss (Gerona-Daga & Salmo, 2022; Lovelock et al., 2022), which might explain the focus on biomass and soil carbon in many restored mangrove sites reported in the literature. In contrast, 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). While these interventions are critical for recovering ecosystem functions, they often involve transforming a non-wetland state (e.g., a drained agricultural field) back into a wetland. In such cases, researchers may skip sampling the pre-restoration site since a heavily altered site like a farm field is not considered a “wetland” per se. This means those projects do not yield a paired comparison for analysis and thus fail to meet

our selection criteria. Although understandable from a local management perspective, it results in lost opportunities to quantify the carbon benefits of those types of restoration. The imbalance in available data across restoration strategies limits our ability to compare the effectiveness of different approaches under varying conditions, information that would be very useful for prioritizing restoration projects if carbon sequestration is a desired outcome of future policies.

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 not published internationally. Indeed, many restorations are primarily motivated by goals like enhancing biodiversity or habitat quality and thus may lack carbon cycle and GHG flux monitoring entirely (Bertolini & da Mosto, 2021). Further data may appear in grey literature (e.g., technical or local reports) or in non-English publications, which are excluded from our global review and could potentially bias our results (Hannah et al., 2024; Konno et al., 2020). Additionally, many studies adopt designs that do not incorporate a restored *versus* altered site comparison - for instance, monitoring a restored site over time or comparing a restored site to a reference natural wetland - especially when the pre-restoration condition is something as dissimilar as a cornfield or shrimp pond. Although such approaches yield valuable insights, they do not provide the direct comparison that best isolates restoration impacts on the carbon cycle (Mahlum et al., 2018; Smokorowski & Randall, 2017). In summary, the evidence base contains substantial gaps, and our findings should be interpreted in light of these biases in the literature.

Restoration Effects on Carbon Pools, GHG Fluxes, and DOC

Our meta-analysis indicates that coastal wetland restoration generally has positive effects (i.e., accumulation) on carbon pools. On average, restored sites had higher aboveground biomass, belowground biomass, and soil carbon than their altered (degraded) counterparts. These results underline the importance of healthy wetlands as carbon sinks and align with well-understood mechanisms of carbon sequestration in wetlands (Guo et al., 2025; L. Li et al., 2024; Taillardat et al., 2020) . Recovering the vegetative cover enables plants to fix CO_2 from the atmosphere into organic matter, increasing carbon stored in living biomass, while root growth and litter fall introduce new organic inputs into the soil (Dontis et al., 2020). Even if submerged plants are more labile and contribute less to recalcitrant carbon burial, helophytes and other higher plants, including wetland associated trees, provide a significant amount of refractory carbon that could be stored in the soil. 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). Likewise, restoring natural hydrology, for example, by reintroducing tidal inundation to a previously cut-off area, creates waterlogged soil conditions that slow down aerobic decomposition, helping to preserve and build up soil organic matter (Lorenz & Lal, 2018), though if redox potential is high, for example in organic matter rich sediments of polluted wetlands, methane fluxes may increase

in turn. We also noted that variability in above- and belowground biomass across sites declined after restoration (restored sites were more uniformly carbon-rich in plant biomass), whereas variability in soil carbon remained high, suggesting that soil carbon response is more dependent on site-specific factors and time for accumulation (Brown & Norris, 2018; Shao et al., 2022). The large differences we observe among studies likely reflect factors such as sediment availability, inundation regimes, rates of elevation change, and the initial condition of the site prior to restoration. For instance, converting a drained, carbon-poor tract of land (e.g., an agricultural field or aquaculture pond) back to a tidal salt marsh can lead to rapid initial carbon accumulation once tidal exchange and plant growth resume (Eagle et al., 2022; Gulliver et al., 2020). In contrast, restoring a degraded wetland that still retains some ecological function (e.g., a sparsely vegetated saltmarsh) is expected to yield more modest additional carbon gains, because plant biomass and soil carbon stocks are already present and the system is closer to a natural state (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). These pulses are hard to capture without permanent monitoring but considering the comparably short time between restoration and study within our dataset, those pulses might be better reflected than in long-term studies. 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. It is important to note that the GHG flux dataset was relatively small (especially for CH_4 and N_2O), so the lack of statistically significant effects should be interpreted with caution. Transient spikes of GHGs immediately following restoration actions (e.g., upon initial re-flooding of dried soils) might not have been captured in most studies, as only a few conduct high-frequency monitoring during the early post-restoration period. Future research that includes continuous or long-term gas measurements would help to identify short-term emissions pulses and confirm the longer-term trajectories of GHG fluxes in restored coastal wetlands.

DOC was the most underrepresented component of the carbon cycle in our meta-analysis. We obtained only one single comparison involving DOC, which is insufficient to evaluate any general effect. This is a critical knowledge gap because DOC can play a crucial role in carbon storage and fluxes. Depending on its biological lability it can be an important sink of carbon, particularly for refractory dissolved aromatic substances. Further, DOC fluxes can be a significant pathway by which wetlands lose, gain or redistribute carbon to adjacent aquatic systems (Alongi, 2014). For example, mangrove forests and seagrass meadows are known to release substantial amounts of DOC into coastal waters (Barrón & Duarte, 2015; Ray et al., 2023; Sippo et al., 2017). A portion of this DOC is refractory, meaning it can persist and be transported offshore, effectively transferring carbon from the wetland to oceanic storage (Dittmar et al., 2006; Maher et al., 2013). Unfortunately, current data is insufficient to quantify this effect. We therefore emphasize the importance of including DOC monitoring in future assessments of wetland restoration. A full accounting of carbon gains and losses in restored ecosystems should integrate carbon stocks, GHG fluxes, and DOC concentration and its biological lability to determine actual net carbon sequestration.

Variation among Coastal Wetland Types and Contexts

Our meta-regressions did not detect statistically significant differences in restoration outcomes among wetland types in contrast to our hypothesis. However, this result does not imply that all wetland types respond identically to restoration, but rather that the current dataset is too limited and unbalanced to conclusively distinguish type-specific effects. The coverage is heavily skewed towards mangroves that account for a large fraction of the data, whereas other ecosystems have very few studies, reducing the power to compare groups. Nonetheless, some qualitative patterns emerged when considering each ecosystem type individually.

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).

In restored saltmarshes, plant biomass generally increases substantially. Nearly all studies have observed a recovery of vegetation (aboveground biomass) following restoration (Bartolucci & Fulweiler, 2024; Flynn et al., 1999). Our findings indicate that saltmarsh vegetation (grasses and succulents) can recolonize and thrive once conditions like tidal flow are restored or grazing pressure is removed. However, changes in soil carbon in saltmarsh restorations were variable. Some restored marshes showed marked increases in soil carbon compared to the altered condition, while others showed little to no change within the time frame of the study. This variability likely reflects differences in 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). It may also be that soil carbon accumulation in saltmarshes is a slower process and short-term studies might not yet capture significant gains. Over long time scales, however, restored saltmarshes are expected to accumulate carbon at rates comparable to natural systems (Kirwan & Mudd, 2012).

Seagrass restoration is comparatively under-studied in terms of carbon outcomes, though the limited data still suggest a positive trend (S. Liu et al., 2023; Tanner et al., 2021). Successful seagrass meadow restorations can lead to the gradual buildup of organic-rich sediments as the seagrass traps particles and add their own detritus to the soil (Johannessen, 2022). This would increase soil carbon over time. Indeed, our meta-analysis hinted at soil carbon gains in restored seagrass meadows, although with only a few data points, the trend was not statistically significant. Above- and belowground biomass data for seagrass restorations are scarce, but presumably restoration enables these plants to resume their role in carbon storage both in plant tissue and sediments. Overall, while promising, the evidence for seagrass carbon sequestration after restoration remains limited.

Brackish coastal wetlands showed patterns of carbon recovery broadly similar to saltmarshes (Doroski et al., 2019). In our dataset, restored brackish sites exhibited significant increases in soil carbon, in fact, brackish marshes had one of the clearest soil carbon gains beside mangroves (Shiau et al., 2016). Aboveground biomass also increased with restoration (typically reestablishment of salt-tolerant reeds or grasses), although relatively few studies focused on brackish systems specifically. Brackish wetlands often receive some tidal influence but also freshwater input, and the restoration outcomes may combine aspects of both salt and freshwater wetlands. For example, they could accumulate carbon without increased CH_4 emissions at moderate salinity (Chen et al., 2022; Holm et al., 2016), but detailed data on this are lacking.

Freshwater coastal wetlands were the least represented wetland type in our analysis. This is partly because many coastal freshwater restorations involve converting a non-wetland land use (like agricultural fields) back into wetland, a scenario that was often excluded from our literature selection if the pre-restoration site was not sampled. From the few cases available, it appears that restored freshwater wetlands do see increases in plant biomass similar to other types (vegetation comes back when hydrology is restored), and likely gains in soil carbon as well, especially if previously drained peat or muck soils begin to re-accumulate organic matter. On the other hand, freshwater restorations are also the cases where we observed the potential for higher CH_4 emissions post-restoration, since re-flooding organic-rich soils creates ideal conditions for methanogenesis (Poffenbarger et al., 2011). We also saw one example of a large decline in N_2O flux after tidal reconnection, suggesting that altered hydrology and anoxic, rewetted soils may shift denitrification towards N_2 and reduce N_2O emissions (Cadier et al., 2023). Given the extremely small sample size, these findings should be considered suggestive rather than representative of all freshwater restorations.

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). This evidence can be used by policymakers and carbon finance programs (e.g., blue carbon credit initiatives) to justify the inclusion of coastal wetland restoration in climate mitigation portfolios (Sapkota & White, 2020). While the overall message is positive, we also caution that the evidentiary foundation is uneven across regions and ecosystem types. Many parts of the world (for example, Africa or temperate Europe) have little or

no data on the carbon impacts of coastal restoration, yet these are regions where significant restoration could be applied in the future. This lack of representation means that current global estimates of blue carbon benefits carry considerable uncertainty. Before the full climate mitigation potential of coastal wetland restoration can be realized and confidently quantified for all regions, there is a pressing need to fill these data gaps. Policymakers may thus consider investing in capacity-building, long-term monitoring, and region-specific research to ensure that carbon sequestration estimates are grounded in local evidence. 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. Overall, the restored wetlands in our synthesis acted as net carbon sinks, absorbing and storing carbon while not becoming significant additional sources of climate-relevant gases. 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.

These findings carry important implications for climate policy, conservation strategy, and future research. They provide quantitative support for integrating coastal wetland restoration into climate change mitigation frameworks such as national climate plans and carbon offset projects. By demonstrably enhancing carbon sequestration, restored wetlands can contribute to achieving emissions targets, all while delivering co-benefits like biodiversity conservation and coastal protection. 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. Filling these knowledge gaps, especially by studying underrepresented wetland types, climatic regions, and including full carbon accounting (stocks, fluxes), will refine our understanding of how much climate mitigation coastal restoration can realistically provide. In summary, coastal wetland restoration has proven carbon sequestration benefits and minimal GHG drawbacks in the contexts analyzed, making it a strong candidate for nature-based climate solutions. With expanded evidence and careful implementation, it can be scaled up as a reliable strategy to help meet climate mitigation goals while we continue to improve estimates of its full potential through further study.

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Supplementary Material:

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

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Methods:

Scopus search query:

TITLE-ABS-KEY(("carbon stock*" OR "carbon sequestration*" OR "carbon storage" OR "carbon flux*" OR GHG* OR "organic matter*" OR greenhouse* OR CO2 OR CH4 OR N2O OR DOC OR TOC OR DOM OR "recalcitrant carbon" OR "soluble organic matter" OR "soluble organic carbon" OR methane OR "nitrous oxide" OR "carbon dioxide" OR "aboveground biomass" OR "belowground biomass") AND ("coastal wetland*" OR lagoon* OR "salt marsh*" OR saltmarsh* OR "tidal marsh*" OR estuar* OR delta* OR mangrove* OR seagrass OR "sea grass" OR "tidal wetland*") AND (restoration* OR rehabilitation* OR revitali*ation* OR renaturali*ation* OR management*)) AND (DOCTYPE(ar) OR DOCTYPE(dp)) AND LANGUAGE(English)

Figures:

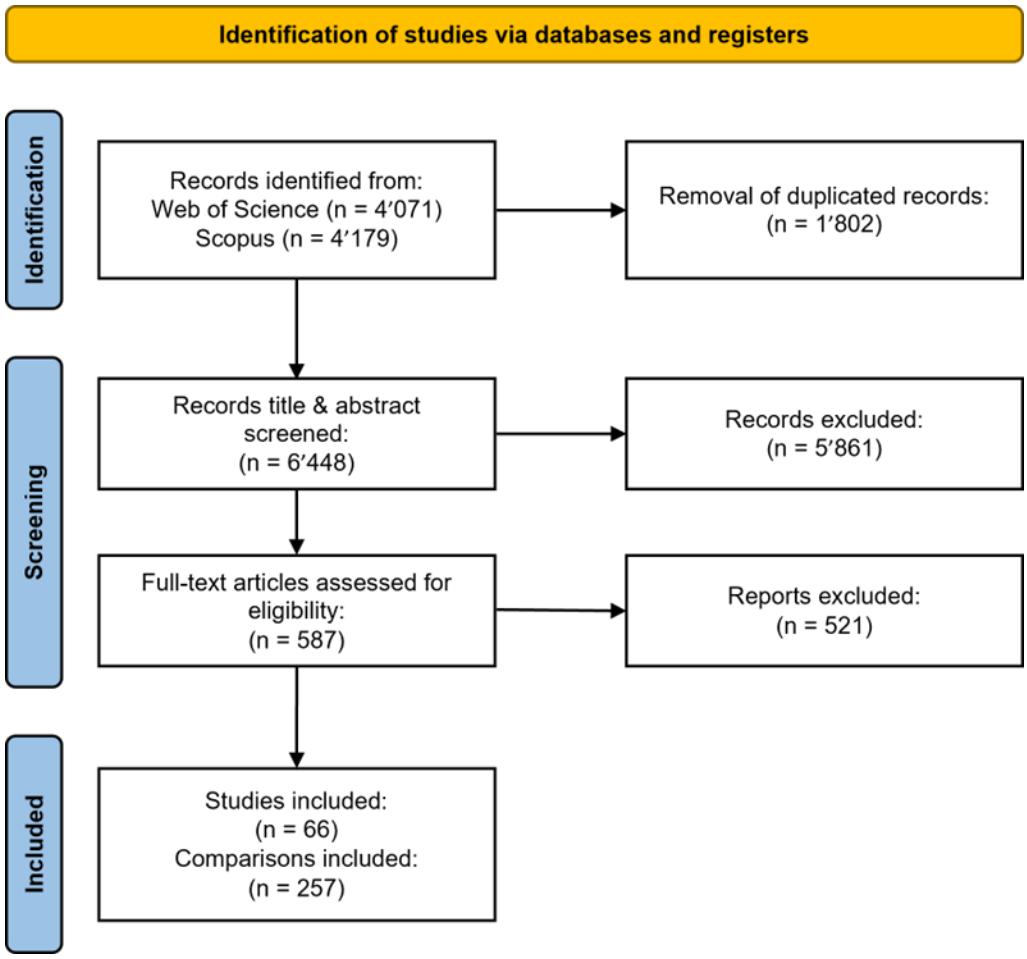

Figure S1: PRISMA 2020 (Page et al., 2021) Flow diagram showing the workflow from identified records to the final selected studies and comparisons (each comparison represents one measured parameter in a pair of degraded versus restored sites).

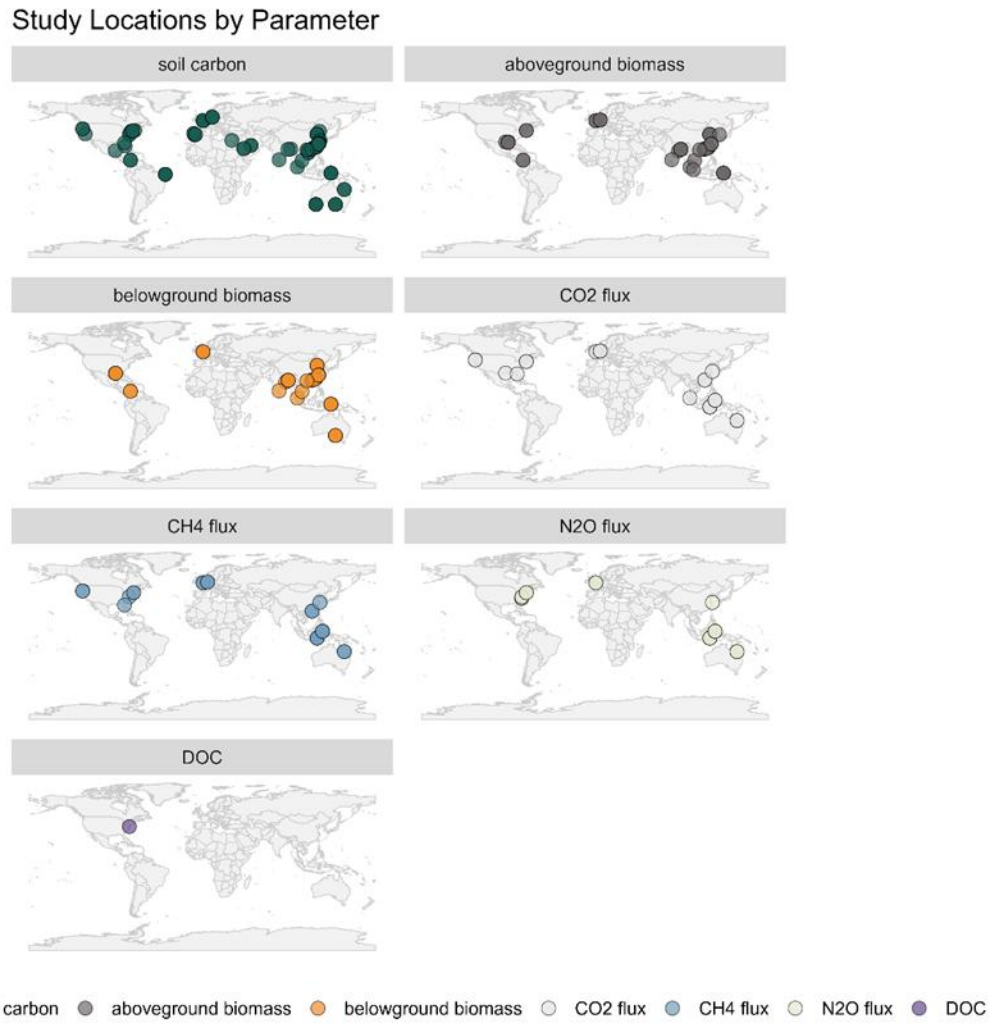

Figure S2: Global distribution of studies by parameter, where each dot corresponds to one comparison; darker shading denotes overlapping data points.

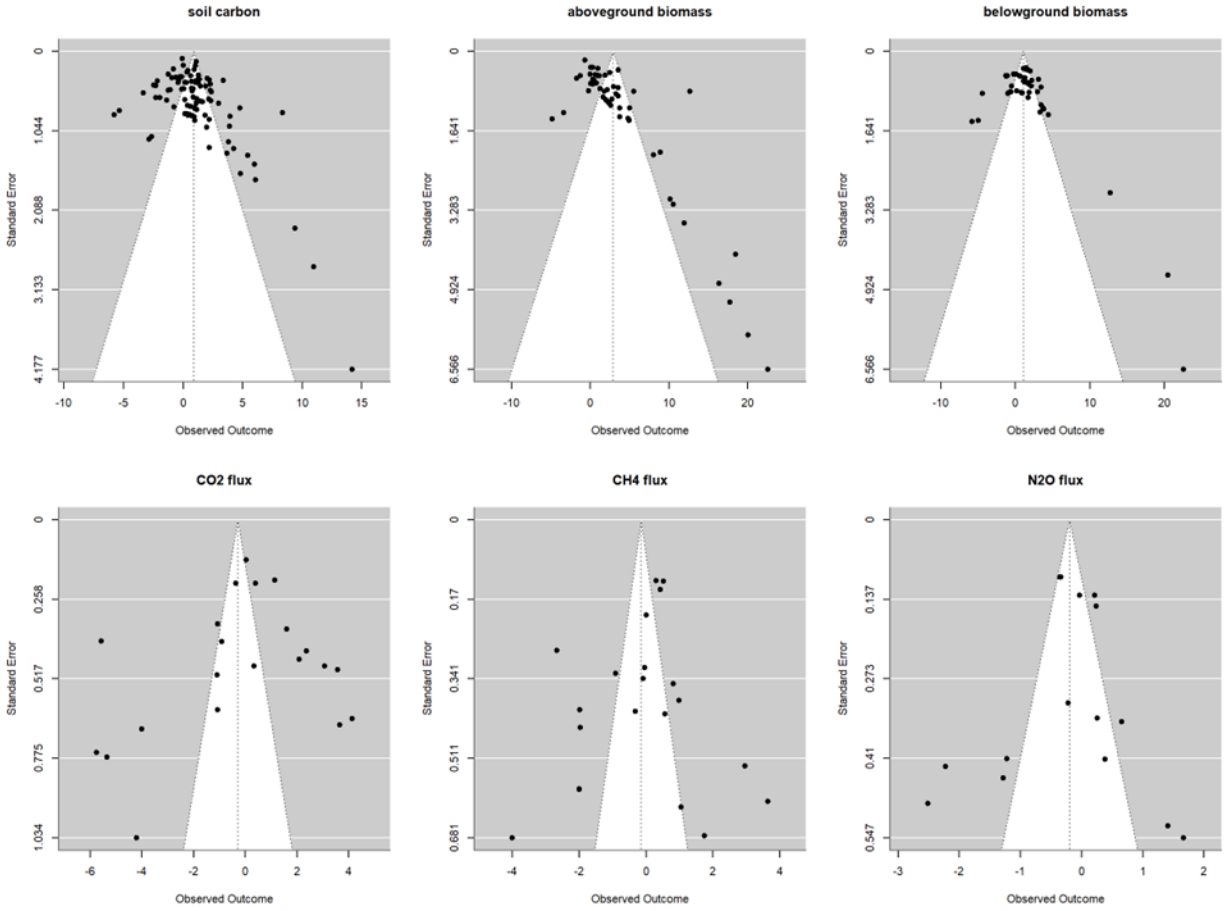

Figure S3: Effect size (SMD; x) versus standard error (y) for each studied parameter. The vertical line marks the pooled estimate from the univariate random-effects model. Diagonal lines indicate 95% confidence interval.

Tables:

Table S1: Inclusion and exclusion criteria used to screen records for the meta-analysis.

Criteria	**Include**	**Exclude**
Source	Web of Science Core Collection and Scopus	Other data sources
Publication type	Research articles or data papers reporting original **field** measurements	Reviews, meta-analyses, modeling-only studies, extrapolations without field data, laboratory or mesocosm studies
Language	English	All other languages
Ecosystem	Sites historically classed as coastal wetlands	Non-coastal wetlands and non-wetland habitats
Geography	Studies conducted across the globe	-
Restoration action	Any active or passive restoration or management action that aims to recover coastal wetland structure or function	Monitoring of only natural or only impacted sites with no restored control sites
Variables	Quantified carbon pools (soil or sediment, aboveground or belowground biomass), DOC or GHG fluxes (CO_2, CH_4, N_2O)	Studies that do not report any of these variables
Design	Direct comparisons of altered (degraded, impacted) versus restored sites; BA (Before-After), CI (Control-Impact), or BACI (Before-After-Control-Impact) comparisons	Studies reporting only restored versus natural reference comparisons, or restored sites only
Data provision	Means, SDs, and sample sizes for each group, or convertible statistics allowing SD derivation (SE, CI, IQR, range)	Statistics insufficient to derive SD or unclear sample sizes even after author contact (for studies published in the past 5 years)

Table S2: Overall pooled estimates with 95% CIs and p-values. Top block reports standardized mean differences (SMD) for mean responses; bottom block reports log coefficient of variation ratios (lnCVR) for variability.

SMD

Parameter	Estimate	CI lower bound	CI upper bound	p-value
soil carbon	0.78	0.23	1.33	0.0057
aboveground biomass	3.09	1.86	4.32	0
belowground biomass	1.18	0.46	1.91	0.0014
CO2 flux	-0.3	-1.61	1.02	0.6564
CH4 flux	-0.09	-0.93	0.76	0.8423
N2O flux	-0.06	-0.69	0.56	0.8409

lnCVR

Parameter	Estimate	CI lower bound	CI upper bound	p-value
soil carbon	0.17	-0.01	0.35	0.0653
aboveground biomass	-0.57	-0.94	-0.2	0.0024
belowground biomass	-0.26	-0.48	-0.03	0.0237
CO2 flux	0.17	-0.31	0.65	0.4825
CH4 flux	0.12	-0.51	0.75	0.707
N2O flux	0.19	-0.41	0.79	0.5412

Table S3: For each parameter: number of comparisons (n), Egger's regression (Z, p), and Rosenthal's fail-safe N from univariate random-effects fits.

Parameter	Number of comparisons	Egger's Z statistic	Egger's p-value	Fail-safe number
Soil carbon	97	2.34	0.021	289
Aboveground biomass	54	5.39	0.000	354
Belowground biomass	45	1.03	0.309	69
CO2 fluxes	22	-0.5	0.621	0
CH4 fluxes	21	-1.02	0.322	0
N2O fluxes	17	0.1	0.925	0