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Blue Nitrogen Follows the Fate of Tidal Wetlands

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Key Points:

- Marshes and mangroves, which cover less than 0.1% of the Earth's surface, bury 3.2 Tg N yr^{-1}, representing 13%–15% of total marine N burial
- Mangrove soils have a higher mean C:N ratio than marshes, but the soil N accumulation rates per area do not differ between the two habitat types
- Widespread loss of tidal wetlands, as many predict, could release large amounts of organic nitrogen, exacerbating coastal pollution

Supporting Information:

Supporting Information may be found in the online version of this article.

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Abstract

Tidal wetlands sequester carbon (C) at much higher rates per area than other ecosystems, helping to offset C emissions. The burial of organic C in tidal wetland soils, "blue C", is tightly linked to the cycling of nitrogen (N), which is a key pollutant and limiting nutrient for many ecosystems. The large fluxes of N in and out of tidal wetlands can have strong impacts on surrounding water quality. However, the global burial rate of "blue nitrogen" and its potential response to future sea-level rise (SLR) and wetland extent changes have not been quantified. Here, we assembled a global database of 8012 soil N measurements from 255 tidal wetland sites and found that the relationship between soil C and N concentrations was consistent but differed between the two dominant types of emergent tidal wetlands—marshes and mangroves. Leveraging extensive knowledge of blue C accumulation, we estimated that tidal wetlands, which cover less than 0.1% of the Earth's surface, bury 3.2 Tg N yr^{-1} (2.5 Tg N yr^{-1} in mangroves, and 0.7 Tg N yr^{-1} in marshes), representing 13%–15% of marine N burial. This rate could more than double globally by 2,100 if wetland elevation increases with accelerating SLR, enhancing wetland burial of coastal N prior to release into estuarine bays and oceans. Alternatively, if wetlands submerge and soil erodes, N sinks could reverse and become sources, exacerbating coastal pollution.

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Plain Language Summary Nitrogen pollution can cause major coastal ecological problems, such as harmful algal blooms and fish kills. Tidal wetlands, such as marshes and mangroves, partially offset human-caused pollution by burying large amounts of nitrogen in soils, yet the global rate of this “blue nitrogen” burial has remained unquantified. We found that absorption of nitrogen by marshes and mangroves accounts for 13%–15% of global marine nitrogen burial. This rate could double as rising seas force wetlands to build soil faster, which could help manage future nitrogen pollution. However, if wetlands submerge, they will stop scrubbing surrounding waters and could release stored nitrogen, exacerbating nutrient pollution.

1. Introduction

The accrual of “blue carbon” captured by tidal marshes, mangroves, seagrasses, and tidal freshwater wetlands has made preserving and restoring tidal wetlands a focal natural climate change solution (Adame et al., 2024). Meanwhile, escalating nutrient runoff has fueled recent eutrophication crises, especially in southeast Asia, Europe and eastern North America (Lan et al., 2024), eliciting harmful algal blooms and massive fish kills (Brewton et al., 2022; Howarth, 2008; Wang et al., 2019), even while N loads may be declining in some uplands (Mason et al., 2022). Tidal wetlands absorb N by catching sediment and through uptake by primary producers. Both N pools, exogenous sediment and endogenous organic matter, can be preserved for millennia in soil profiles. Despite the importance of N in ecosystem functioning and water quality (Odum, 1980; Twilley, 1988; Valiela & Teal, 1974), the global magnitude of tidal wetland N soil burial has yet to be explicitly estimated. Inferred estimates of global marine N burial range from 22 to 25 Tg N year^{-1}, but the current contribution of tidal wetlands is unclear (Gruber & Galloway, 2008; Van Engeland et al., 2011; Voss et al., 2013). Moreover, the burial rate is likely to change with ongoing physical changes to tidal wetlands. For instance, accelerating relative sea-level rise (rSLR, the local rate of change in sea level relative to the land surface) causes increased tidal inundation, which can increase sediment deposition, expanding soil volume, increasing N burial rate, and enhancing organic matter preservation (Kirwan & Mudd, 2012; Morris et al., 2002).

The N cycle is intimately linked to the C cycle, so stoichiometric relationships to C can illuminate N budgets. Valuation of the C sink capacity of tidal wetlands as a natural climate solution has led to an increasing demand for tidal wetland conservation and restoration worldwide (Lovelock et al., 2023; Macreadie et al., 2021; Tang et al., 2018). Specifically, blue C syntheses have estimated the magnitudes of C stocks in mangrove and tidal marsh biomass and soil C burial potential (Breithaupt et al., 2020; Rogers et al., 2019; Wang et al., 2021; Weston et al., 2023). In addition to controlling the production of potent greenhouse gases like nitrous oxide, N availability is a key regulator of C fluxes globally (Thornton et al., 2009). For example, N availability is a limiting nutrient for terrestrial and aquatic primary productivity (LeBauer & Treseder, 2008; Paerl, 2018). Thus, N uptake and storage are co-benefits to wetland conservation and restoration that have gained recognition within international sustainability efforts (The Ramsar Convention on Wetlands, 2018). If soil N stoichiometry is predictable from commonly measured parameters, we can leverage extensive knowledge of present and future C burial and storage in wetlands to estimate the current status and future capacity for the critical co-benefit of N burial and storage.

To estimate global blue N stocks and accumulation rates, we (a) assembled a database of soil N concentration in mangroves and tidal marsh soils, (b) identified the best predictors of soil N concentrations, (c) estimated current N stocks and accumulation rates, and (d) projected how those stocks and rates may change in the future. Our database of tidal wetland soil profiles included 8012 samples from 910 cores across 255 tidal wetland sites on 6 continents (Figure S1 in Supporting Information S1), with soils analyzed for both organic C and N concentrations. We developed models to predict soil C:N and used extensive databases of soil C accumulation rates to estimate N accumulation rates in tidal wetlands worldwide (Macreadie et al., 2021; Wang et al., 2021). Finally, because tidal wetlands can adjust to the acceleration of rSLR by increasing rates of surface elevation gain (Morris et al., 2002; Rogers et al., 2019), we extrapolated the potential for global N accumulation in, or release from, tidal wetland soils according to projections for future sea level and change in tidal wetland extent.

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2. Data and Methods

2.1. Data Sources

We assembled a soil nitrogen (N) concentration database in tidal marsh and mangrove soils to feed our subsequent analyses (Figure S6 in Supporting Information S1) using two existing databases, additional individual papers, and unpublished data sets. The Coastal Carbon Library and Atlas (CCLA) provides depth-interval soil organic C concentration data from several thousand cores (Holmquist et al., 2024). We identified individual studies that submitted soil N concentration data to the CCLA and were not included in the primary database, and merged the depth-specific N data with the existing metadata for each core. This included studies from the database published by Maxwell et al. (2024), which has been incorporated into the CCLA, again selecting only studies that included N concentration data. We also included other tidal wetland soil core studies for which we could obtain soil organic C and N concentrations, bulk density, core location, and habitat type, targeting poorly represented geographical regions. We included 31 studies, each comprising from 1 to 32 different sites. Our data included cores from 6 continents, though Africa and Asia, particularly those countries outside of China, were poorly represented (Figure S1 in Supporting Information S1). At each site, measurements were taken from soil cores at various depths, ranging from 0 to 1 m. We restricted data to soil segments with center depth <1 m because relatively few studies included deeper samples, and we intended to extrapolate to only the top meter of soil. We unified all data into a single database. We calculated the center depth of each core segment as the average of the upper and lower depths. In studies that distinguished organic from total or inorganic C, we used organic C. While some inorganic N may contribute to bulk soil N measurements, we estimate that >99% of soil N occurs in organic form (Tobias & Neubauer, 2019). Data were only included where habitats were characterized as tidal marsh or mangrove. We did not restrict by salinity class, but the database was dominated by brackish and saline wetlands.

2.2. Predictive Models for Soil [N], C:N and N Density

Our primary goal was to use statistical models to predict N concentration, N stocks, and N accumulation for our global extrapolations. A secondary goal was to generate predictive formulas that may be useful for scientists or land managers in estimating N stocks and burial. We first aimed to determine the relationship between soil [N] and soil [C]. For this purpose, we sought the best model that considered only habitat type and soil [C], and their interaction while adjusting for the data structure. We filtered a subset of analytical outliers by selecting only soil core segments with a C:N ratio between 5 and 100; samples outside of this range likely result from analytical error in either [C] or [N] (Amorim et al., 2022). These outliers (both high and low C:N) tended to have very low [C], suggesting that measurements were possibly near the detection limit for N measurements, leading to erroneous C:N values. We omitted 330 soil samples, representing 4% of the database. It is possible that some of the low [C] values that had unrealistically high C:N had contamination from carbonates that were not removed prior to analysis (Holmquist et al., 2024). We fit a linear mixed-effects model to predict soil [N], using organic C, habitat (marsh vs. mangrove), and the interaction of organic C with habitat as a fixed effects. Soil cores in sites nested within each study were treated as random effects to account for the hierarchical data structure. This allowed the model to appropriately account for variability at the study, site, and soil core levels, leading to more robust estimates of the fixed effects. To evaluate the prediction accuracy of the model, we conducted leave-one-out cross-validation. For each observation in the data set, we fit the mixed-effects model using the remaining data and predicted soil [N] for the excluded observation. We then calculated the correlation coefficient between the predicted and actual soil [N] values to assess the strength of the linear relationship, where a higher correlation coefficient indicates greater prediction accuracy.

To convert existing soil C stock and flux estimates into N, we built a second linear mixed-effects model to estimate C:N ratios for marsh and mangrove soils, because C:N ratio remains consistent regardless of the measurement units of C and N. This model used C:N as the response variable, habitat (marsh or mangrove) as the fixed effect, and the same random effects as in the first model to account for the hierarchical data structure. We also included the center depth of each soil segment and bulk density as additional fixed effects to determine how other factors may influence soil C:N. We estimated the mean soil N density, the product of N concentration and dry bulk density, for tidal wetlands to allow calculation of net N changes in the projections described below. We built a simple mixed-effects model of N density versus habitat while accounting for the hierarchical data structure using the same random effects as in the first model. For all three response variables, including depth slightly improved the Bayesian Information Criterion (Tables S1–S3 in Supporting Information S1), but we report on the

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simpler models so that predictions can be made for cases in which not all predictors are available. In all models, we excluded predictors that were not independent of the response variable. For instance, we excluded soil [C] as a predictor from the models predicting C:N ratio. In each model, continuous predictors were linearly related to responses and variances. Although model residuals deviated from normality, linear mixed-effects models are robust to such deviations, particularly with large sample sizes (Gelman, 2007; Schielzeth et al., 2020).

2.3. Global N Stocks and Accumulation Rates

We applied our C:N ratios for each habitat to recently reported estimates of C accumulation and stocks in marshes and mangroves to estimate N stocks and accumulation rates. To estimate current N stocks in marsh and mangrove soils, we applied our habitat-specific C:N estimates to published estimates of C stocks in these habitats (Macreadie et al., 2021), multiplied by the area of each habitat, and summed to obtain a global total (Table S4 in Supporting Information S1). To estimate the distribution of N accumulation rates per unit area, we applied our model predictions to a database of individual estimates of C accumulation rates that used radio-isotopes to determine soil accumulation rates (Wang et al., 2021). To estimate the global rate of N accumulation, we applied the marsh and mangrove soil C:N estimates to a wetland C database (Macreadie et al., 2021). That study used environmental variables such as mean annual temperature, precipitation, and local rSLR to account for geographically biased sampling in wetland C accumulation measurements in estimating C accumulation for the total wetland area by country (Wang et al., 2021). Because we focus on N, the error in all estimates reflects the model error for the N parameter we used in each case, not the error associated with the C parameters.

2.4. Projections

We considered six N futures as combinations of projections for rSLR and for wetland area gain or loss (Table S4 in Supporting Information S1) to envelop the range of future possibilities of N burial, storage and release. Wetland area change and rSLR were varied independently in generating our scenarios (Figure S5 in Supporting Information S1), despite known linkages, because there is considerable disagreement in the literature about the relationship between rSLR and wetland area (Saintilan et al., 2022; Schuerch et al., 2018). Projections of future wetland area differ widely, owing to uncertainty in factors such as accommodation space, sediment loads, and the maximal rSLR rates that tidal wetlands can tolerate (Saintilan et al., 2022). Moreover, changes in wetland area depend on social and political variables such as coastal development and restoration (Kirwan et al., 2016; Saintilan et al., 2022; Schuerch et al., 2018).

To estimate future wetland elevation gain, we made the simplifying assumption that patterns in local isostatic rSLR will follow global eustatic SLR on the decadal and centennial time scales relevant for management and decision-making. We also assumed that the global mean wetland elevation gain would match the global mean rSLR up to a threshold. Yet, we recognize that the rSLR varies enormously based on local sea-level trends and vertical land adjustment, and that some wetlands, such as tidal freshwater wetlands, are unlikely to experience widespread erosion in this timeframe. Though using one global mean rSLR minimizes this geographic heterogeneity in individual sites, the global average trends in rSLR broadly follow long-term trends in global mean SLR, especially for the temperate zones that have not been glaciated and the passive margins of continents where most coastal wetland area occurs (Oelsmann et al., 2024).

To generate future sea-level scenarios, we chose a low-end and high-end estimate (Jevrejeva et al., 2019). We assume that SLR will accelerate (Morris et al., 2023) as $SLR_y = SLR_{2020} * e^{k(y-2020)}$ where y is the year and k is the growth constant determined as $\log(SLR_{2100}/SLR_{2020})/(2100–2020)$ with the resulting SLR_{2100} determined by each scenario (Table S4 in Supporting Information S1). Low SLR is defined as the mid-range (50th percentile) estimate from the best-case climate scenario (1.5°C warming) by 2100, and high SLR is defined as a mid-range estimate from the IPCC RCP 8.5 (4°C warming, Jevrejeva et al., 2019). To estimate a low-end tidal wetland elevation gain, we assumed that the wetland elevation gain rate matches the pace of rSLR for a best-case scenario in which mean global warming remains at 1.5°C by 2100 (Jevrejeva et al., 2019). To estimate a high-end for global mean elevation gain, we assume that tidal wetlands will gain elevation following rSLR rates up to a rate of 8 mm yr^{-1} by 2100, a mid-range probability of rSLR for the scenario of 4°C warming by 2100 (Jevrejeva et al., 2019). We set the upper threshold of rSLR at 8 mm yr^{-1} as a midpoint among estimates of maximal wetland elevation gain estimated from contemporary measurements (Saintilan et al., 2022), reconstructions (Horton et al., 2018), and modeling for marshes and mangroves (Morris et al., 2023). Though actual SLR could exceed this rate, tidal wetland elevation gain rates are unlikely to exceed 8 mm yr^{-1} (Saintilan et al., 2022).

To generate scenarios that vary in future wetland area, we chose stable, high-loss, and high-gain scenarios. We generated logistic relationships in which rates of change begin close to present-day rates and peak midcentury: $A_y = (A_{2100} - A_{2020})/((1 + e^{(2060-y)})/8)$ where A_y = marsh or mangrove area in year y, A_{2020} = area in the year 2020 with a unique A_{2100} for each scenario according to Table S5 in Supporting Information S1. The inflection point was defined as 2060, the midpoint between 2020 and 2100. Gross wetland area loss is based on 1.5°C and 3°C warming scenarios (Saintilan et al., 2023), and high gain is based on net area gain for RCP 8.5 in the high-density threshold (Schuerch et al., 2018) (Table S1 in Supporting Information S1). Increasing rSLR has been linked to higher rates of soil accumulation and higher proportions of C in accumulating soil (Rogers et al., 2019). Accelerating rSLR could also increase the N concentration of accumulating soil, though our simple model did not assume this.

Future net N change was estimated as the sum of gross N gain in surviving wetlands and gross N released from erosion of lost marsh area. Gross N gain was estimated as the product of mean elevation gain, wetland area at the time of gain, and separate mean N densities of marsh (1.75 mg N cm^{-3}) and mangrove soil (1.98 mg N cm^{-3}). N densities are the product of soil bulk density and soil [N]. Increases in wetland area are included in the soil vertical gain calculations and are assumed to match the vertical gain rate of existing marshes starting from the soil surface at 0 cm depth. To estimate losses, we assumed that wetland area loss results in an erosive release of 50 vertical cm of soil, the approximate difference in elevation between a salt marsh and a mudflat over a retreating margin in an estuary with an average tidal range (Goodwin & Mudd, 2019), but we acknowledge that this estimate is highly uncertain. While ocean bottoms and mudflats may still accumulate N through sediment and organic matter deposition, we assume a net loss during the conversion from wetland. The amount of N represented by this loss was calculated using the N densities above. Net N change is the sum of gross N losses from gross area losses and N gains from elevation and area gains.

The histories of SLR and wetland area change were reconstructed to estimate the change in N budgets relative to the present day. Reconstructions of SLR were taken from a combination of satellite data and gauge records (Table S4 in Supporting Information S1). We applied a linear increase in the rate of SLR from 2.1 mm yr^{-1} in 1970 to 3.5 mm yr^{-1} in 2019. Historical wetland area loss was interpolated from three separate analyses focusing on three historical time intervals (Davidson, 2014; Murray et al., 2022; The Ramsar Convention on Wetlands, 2018). Taken together, the historical record shows a slow loss peaking during the 20th century and slowing more recently.

3. Results and Discussion

3.1. Predictors of Soil N

Habitat type and soil [C] were the most consistently important predictors of soil N and soil C:N. The model using only soil [C], habitat type, and their interaction yielded an accurate prediction of tidal wetland soil [N] (leave-one-out validation, $r = 0.94$, Model 1 in Table S1, Figure S2 in Supporting Information S1). The relationship between soil [C] and soil [N] differed between marshes and mangroves (soil [C] × habitat, $t_{5775} = 28.5$, $p < 0.0001$), but was remarkably consistent within each habitat type (Figure 1, Table S1 in Supporting Information S1). In soils with higher [C], which also have lower bulk density, mangroves had higher C:N ratios than marshes (habitat marsh, $t_{4581} = -4.81$, $p < 0.0001$, Model 2 in Table S2, Figure S3 in Supporting Information S1) likely reflecting the greater influence of lignin-rich woody tissues that resist decomposition (Ola & Lovelock, 2021; Post et al., 1985). Furthermore, soil C:N tended to increase with depth overall when accounting for bulk density and habitat (depth, $t_{7327} = 13.18$, $p < 0.0001$, Model 3 in Table S2, Figure S4A in Supporting Information S1), but the direction and strength of the relationship between C:N and depth varied greatly across studies (Figure S4B in Supporting Information S1). Ultimately, the strong correlation between soil [C] and [N], as well as ample data on wetland soil C stocks and accumulation rates, allows for the prediction of soil N stocks and accumulation rates.

3.2. Tidal Wetland N Stocks and Accumulation Rates

Extrapolating from a database of C accumulation rates across 103 studies representing a broad geographical distribution (Wang et al., 2021), we found that the median N accumulation rate is 7.52 (interquartile range: 4.12–13.61) g N m^{-2} yr^{-1} in mangroves and 8.19 (interquartile range: 5.37–12.44) g N m^{-2} yr^{-1} in marshes

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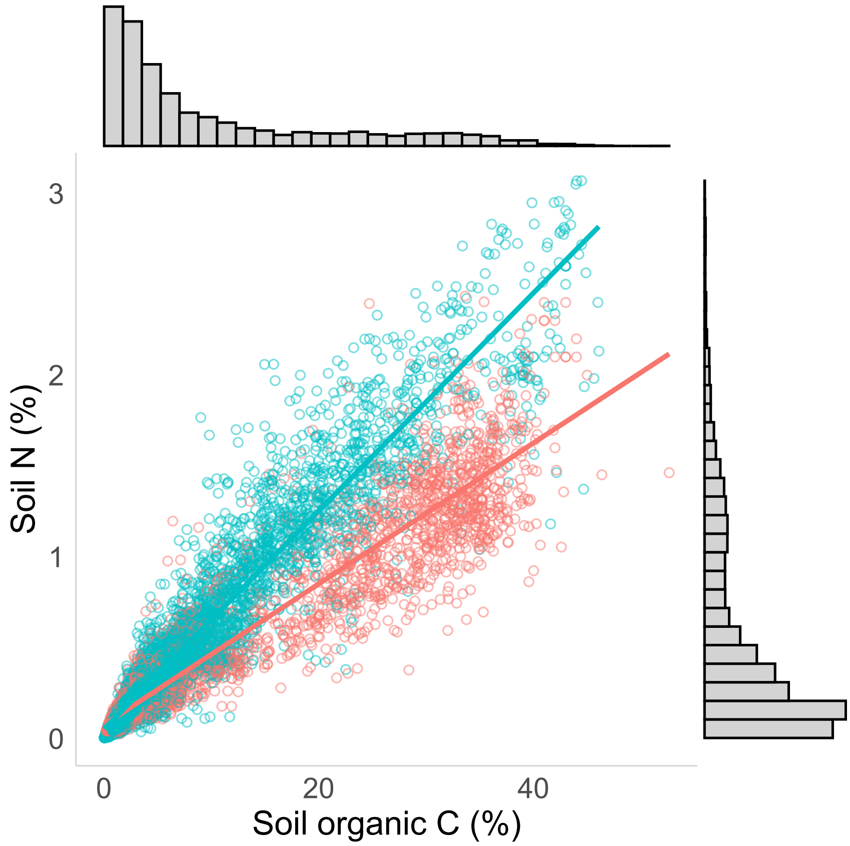

Figure 1. The relationship between soil [N] and soil organic [C] across all depths <1 m for mangroves and tidal marshes. Lines represent the best linear fit. Histograms show the distribution of data along each axis. Number of samples for mangrove = 2,911 and marsh = 5,101.

(Figure 2a). Our estimate falls within previous N burial estimates from regional marsh (Craft, 2007) and mangrove (Cormier et al., 2021) compilations. Scaling to the area of marshes and mangroves that the data set represents (Table S5 in Supporting Information S1), this areal rate yields a global N burial rate of 3.2 ± 0.1 Tg N yr $^{-1}$ (Figure 2b). Our estimate for mangroves, 2.5 Tg N yr^{-1}, exceeds a previous estimate, 1.2 Tg N yr^{-1} (Alongi, 2020), owing partly to a larger mangrove area used herein. The total burial rate is similar in magnitude to global estimates for other external fluxes. Scaling previous area-based estimates for marsh and mangrove fluxes to the globe, denitrification is 3.2 Tg yr^{-1} and N fixation is 1.2 Tg yr^{-1}, but these fluxes have much greater uncertainty (Alongi, 2020; Bouwman et al., 2013; Hopkinson & Giblin, 2008).

We applied our C:N relationships to estimates of C mass in wetlands (Macreadie et al., 2021) to determine that the global mass of N in tidal wetland soils is approximately 278 Tg N to 1 m depth. Our estimate of N accumulation in tidal wetlands represents 13%–15% of the global burial of N estimated for the ocean (Gruber & Galloway, 2008; Voss et al., 2013). As with blue C, the N accumulation rate depends on the soil accumulation rate, which is strongly influenced by the rate of rSLR and availability of sediments, among other factors (Breithaupt et al., 2020; Kirwan & Mudd, 2012; Rogers et al., 2019; Saintilan et al., 2022). How the global N accumulation rate may change in the future depends on the fate of tidal wetland extent and on the amount of vertical accommodation space afforded by increasing rSLR rates (Rogers et al., 2019).

Wetland N burial could represent a strong N sink from surrounding coastal waters, but the impact depends on the sources of buried N. Tidal wetland N demand can be satisfied with inputs of N delivered from tidal flooding, N-fixation, groundwater delivery, and atmospheric deposition (Figure 3) (Alongi, 2020; Tobias & Neubauer, 2019). If the N buried in wetlands derives largely from surrounding waters, then accreting wetlands will draw down estuarine N loads. However, if wetland soil N derives from fixation, wetland soil N gain would constitute less of a sink for estuarine N loads. There is little consensus on the contribution of different N sources

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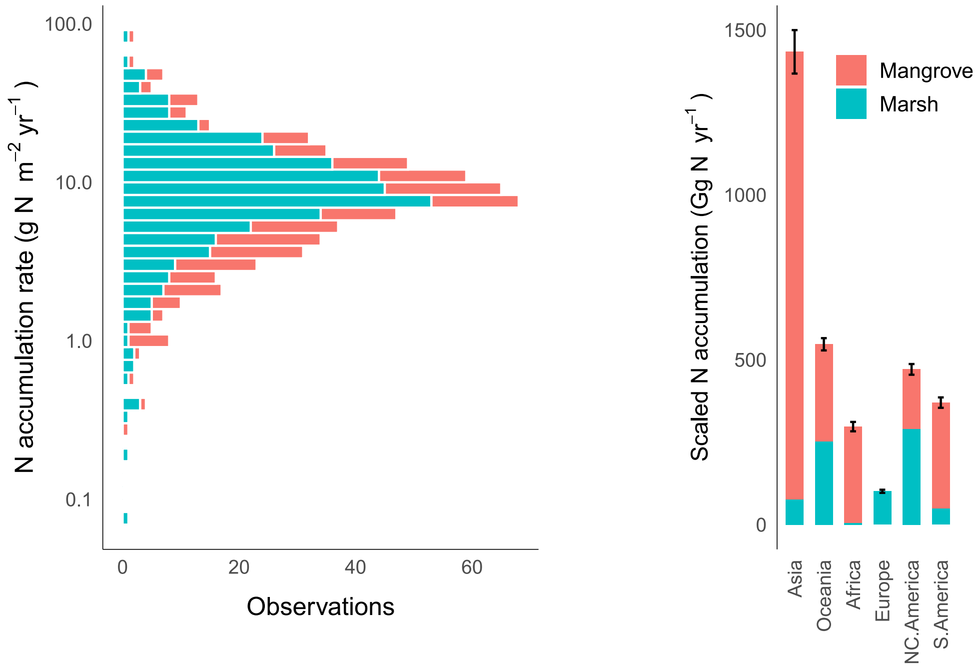

Figure 2. Distribution of area-based and scaled continental N accumulation rates. (a) Histogram of area-based N accumulation rates. Note that the *y*-axis is on log scale. Total observations = 213 for mangrove and 400 for marsh. (b) The global distribution of scaled N accumulation rates, which sum to 3.2 Tg yr^{-1}. Error bars represent the scaled standard error from our modeled C:N estimates.

that support tidal wetland N burial globally (Tobias & Neubauer, 2019), and sources are likely to vary at the scale of catchments (Maúre et al., 2021). However, here we have quantified global rates of tidal wetland N accumulation in soil, which represents a large and relatively certain N flux.

3.3. Future Potential for Global N Accumulation in Tidal Wetland Soils

We found that global N accumulation in wetlands has likely increased since the beginning of the Industrial Revolution, despite the loss of marsh area, owing to accelerated elevation gain in response to rSLR, similar to

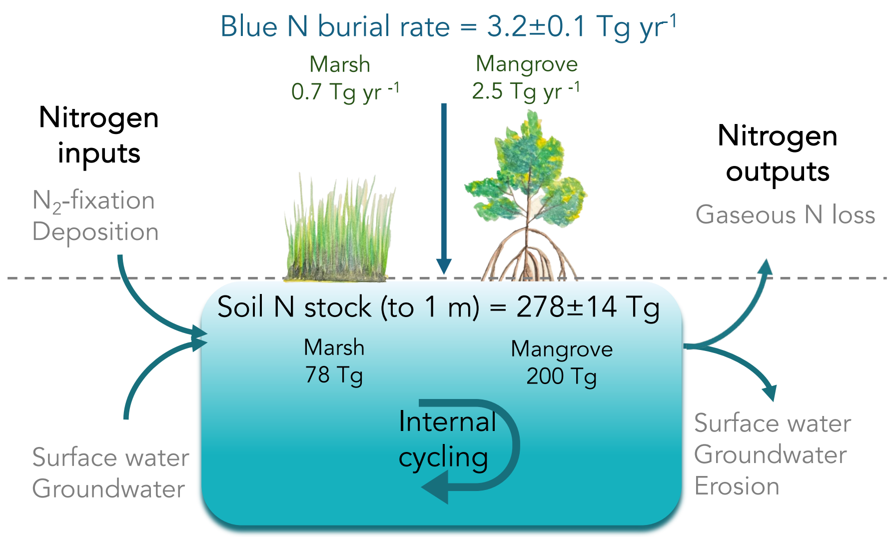

Figure 3. Schematic of global N fluxes in contemporary marshes and mangroves. Soil N stock reflects the upper meter of soil. Error represents the scaled standard error from modeled C:N estimates.

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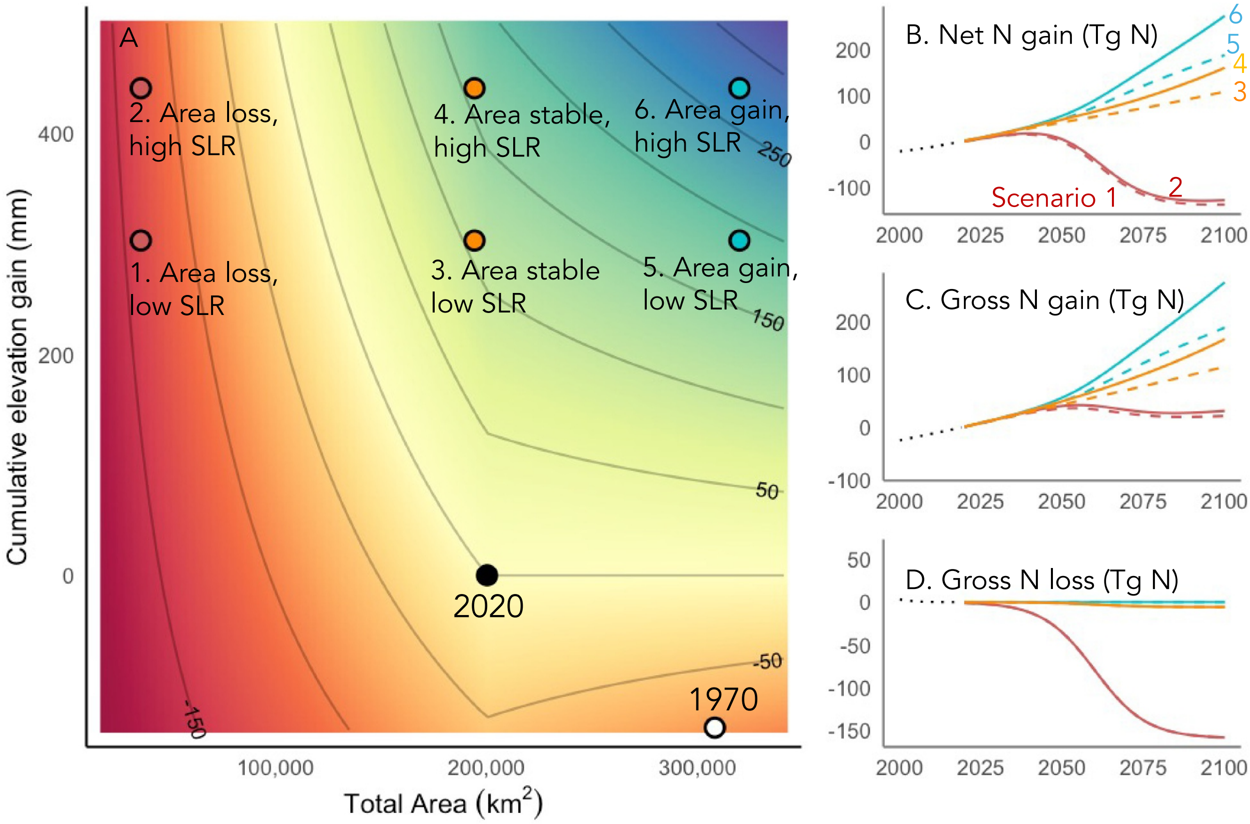

Figure 4. Historical reconstructions and hypothetical future N burial and release. (a) Past (the year 1970), present (2020) and hypothetical future (year 2100 for N futures 1–6) global net N burial in tidal wetlands. Contour colors indicate the global rate of N burial as it varies with the elevation gain rate of surviving wetlands and gains or losses in wetland area. Six N futures (numbered 1–6) represent combinations of high and low projections for tidal wetland area change and wetland elevation gain (Table S5 in Supporting Information S1, Methods). Mean wetland elevation in 2020 is defined as zero. (b) Global net N burial is the sum of (c) gross N burial by changes in elevation gain and (d) gross N loss by wetland area loss for futures 1–6. Colors in (b–d) represent the six N futures.

observations for C uptake (Weston et al., 2023) and burial (Rogers et al., 2019). If the current global mean rate of rSLR remains constant and the wetland area does not change, tidal wetlands would bury an additional 109 Tg N by 2100 (Figure 4, Table S4 in Supporting Information S1). However, future rSLR and wetland area projections vary substantially based on policy, social, and ecological uncertainties, as well as global variations in isostatic adjustment in land elevation that influence rSLR which exerts significant control over tidal wetland organic matter accumulation (Rogers et al., 2019).

The six N futures we generated resulted in vastly different outcomes for global N accumulation (Table S4 in Supporting Information S1). Considering future scenarios of wetland area change and rSLR, total net N loss or accumulation ranges from −138 to +274 Tg N by 2100. High-rSLR scenarios (2, 4, and 6 in Figure 4) resulted in greater N burial than the corresponding low-rSLR scenarios (1, 3, 5 in Figure 4; Table S4 in Supporting Information S1) owing to more rapid N uptake in surviving wetlands. To isolate the influences of area change and elevation gain, rSLR and area change varied independently (Table S4 in Supporting Information S1); however, higher rates of rSLR (scenarios 2, 4, and 6) are likely to lead to greater wetland losses (Saintilan et al., 2023). If high rSLR occurs with gains in wetland area (Boto & Wellington, 1983), then global wetland soil N stocks could nearly double by 2100, an effect that would tend to offset some N loading. At the other extreme, in scenario 1, high wetland losses and low rates of elevation gain could result in a net global release of half of existing tidal wetland soil N stocks.

Summing gross N fluxes to global net rates masks important variation across the local scale where N fluxes hold ecological relevance. Importantly, unlike CO_2 and its effects on climate, the impacts of N eutrophication are heterogeneous and relatively local, such that N trends could have opposing influences in different places. For instance, N may be released from a collapsing salt marsh in one estuary, while topography may allow lateral

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transgression of a mangrove and thus high N uptake elsewhere. To capture the possibility of these effects, we estimated gross N uptake as the rate of soil N accumulation in surviving wetlands (Figure 4c) and gross N loss rates assuming the erosional loss of 50 cm of soil when wetlands are lost (Figure 4d). Even if accelerating wetland elevation gain yields an increase in global *net* N burial, erosion of collapsing wetlands could result in a *gross* release of 150 Tg of N, which could have strong effects on local N fluxes (Figure 4d).

3.4. Will Wetland Elevation Gain Be N Limited?

The large values for global N burial estimated for some scenarios above raise the question whether N scarcity could limit elevation gain at least at some localities in the future. While it is known that N availability commonly limits productivity in most tidal marshes (Langley & Megonigal, 2010; Valiela & Teal, 1974) and some mangroves (Boto & Wellington, 1983; Feller et al., 2003), it is not yet known to what extent plant N limitation translates into N limitation of elevation gain (Mack et al., 2024). The N burial rate of a tidal wetland will relate to the rate of elevation gain given that correlations between rates of rSLR and soil C burial are robust, and variation in soil C:N ratios is constrained (Figure 1). If the elevation gain rate accelerates in response to rSLR, N accumulation in tidal wetlands will also increase.

The extent to which N availability limits elevation gain depends on biophysical context (Worthington et al., 2020). Where exogenous sediment supply is large and soil gain derives primarily from exogenous sediment, N availability is unlikely to exert a biogeochemical limitation on the rate of soil gain, though N limitation may still influence the sediment trapping capacity of plants (Morris et al., 2002). In peat-forming Caribbean mangroves, elevation gain did not appear to be limited by N availability, but these mangroves may be more strongly limited by P than N (McKee et al., 2007). We hypothesize, based on conservation of mass, that organogenic wetlands (those in which soil derives mostly from endogenous organic matter) in ecosystems where plant growth is N limited may have lower potential elevation gain due to N scarcity, particularly where rates of rSLR are high and increasing.

To illustrate with a case study, a brackish marsh on the Chesapeake Bay, USA, has historically gained elevation at roughly 2 mm yr^{-1}, near the millennial rate of rSLR, requiring an N accumulation of 4 g N m^{-2} yr^{-1}. However, from 2000 to 2020, the rSLR rate rose to 7 mm yr^{-1} (Zhu et al., 2022), which would require 14 g N m^{-2} yr^{-1} if the marsh were to gain elevation at this rate, acknowledging this demand could be lower with adjustments in C:N through changes in N and biomass allocation or plant species turnover. Given that depositional inputs are negligible (Langley et al., 2024) and the soil C:N ratio is consistent down the profile at this site (Pastore et al., 2017), a large increase in N inputs through fixation and tidal input or a large decrease in N losses from denitrification or tidal export will be required to meet the increased demand (Figure 3). This adjustment is unlikely considering recent evidence that N limitation of plant productivity has strengthened in this marsh (Langley et al., 2024). We predict that N-limited organogenic tidal wetlands with low terrigenous inputs are the tidal wetland settings most likely to experience increasing N limitation of both plant productivity and elevation gain. Soil accumulation approaching future rates of rSLR will likely either be limited by N availability or necessitate a substantial increase in ecosystem net N uptake rates in organogenic tidal wetlands.

3.5. Fates of Released N

Regardless of its source, once N enters the soil organic N pool, especially below the rooting zone, it may remain nearly inert for millennia depending on the biochemical complexity of organic matter, redox status, soil type, and mineral associations (Steinmuller et al., 2019), though some suggest that buried organic N may be released slowly (Costa et al., 2022). If the wetland is disturbed or the soil erodes, soil N can be abruptly released into surrounding waters (Ardón et al., 2013) and subjected to mineralization, desorption, or reburial; however, the ultimate fate of eroded soil N remains unknown. Exposing previously anoxic soil to oxygenated waters could accelerate the release of mineral N (Steinmuller et al., 2019). One incubation study suggested that only a small portion of eroded organic N will be mineralized, but noted the difficulty of tracking the fate of actual soil in a realistic system (Cornwell et al., 2022). Perhaps eroded soil remains inert as it is deposited elsewhere. If some of the tidal wetland soil N pool is mineralized, it could contribute significantly to local coastal N pollution. Revisiting the example above, Chesapeake Bay has 160,000 ha of coastal wetlands that hold roughly 2.4 Tg of soil N to 1 m depth and are currently burying 6.4 Gg N yr^{-1} given the soil N densities reported here. As rSLR accelerates, surviving wetlands are likely to increase N burial up to 22 Gg N yr^{-1}, possibly partially offsetting historically high riverine N input. However, wetland area loss is expected to accelerate as well. If wetland area is lost at a rate of 1% per year, the N

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loss from erosion could negate N buried by surviving wetlands. Given the large and potentially dynamic pool of N in tidal wetland soils, we highlight an urgent need to better understand the fates of N released from wetlands that are eroding or transforming to other ecosystem states.

4. Conclusions

Historically, wetland soil accumulation has functioned as a robust N sink, reducing N load delivery to estuarine and marine waters. The future influence of tidal wetlands on the river-estuary-ocean continuum hinges on wetland response to rSLR and changes in wetland extent, as well as other drivers such as warming, elevated CO_2, and nutrient pollution. If wetlands survive rSLR or undergo large-scale restoration, high rates of N uptake driven by accelerating wetland elevation gain could improve water quality for sensitive seagrass beds and coral reefs. In less polluted regions, accelerating N burial could strengthen N limitation and oligotrophication (Mason et al., 2022). However, wetland erosion could diminish N uptake rates and release currently stored N, exacerbating eutrophication that leads to harmful algal blooms, dead zones, and fish kills (Brewton et al., 2022). While wetlands are hotspots for other key N-cycle components like denitrification, anammox, and fixation (Bowen et al., 2023), the global rates of many of these fluxes remain highly uncertain (Tobias & Neubauer, 2019). The relatively predictable flux of wetland N-accumulation offers a high-leverage tool for land managers and policymakers, deserving the same strategic attention as "blue C."

We estimate that 3.2 Tg N yr^{-1} is sequestered by mangroves and marshes, representing 13%–15% of global N burial (Gruber & Galloway, 2008; Van Engleland et al., 2011; Voss et al., 2013). Including other types of blue N ecosystems, such as seagrass beds and mud flats, will increase this estimate. The global monetary value of N burial is difficult to estimate owing to small-scale variation in sources and additivity. Yet, multiplying estimates of the monetary value of blue N burial (\$22–106 kg^{-1}, Jenkins et al., 2010) by our global N burial rate (3.2×10^9 kg yr^{-1}) suggests that the global value of blue N burial (\$70–339 billion USD) could exceed that of blue C (\$191 billion, Bertram et al., 2021). Future research examining the sources of N, as well as the alternative fates of N when wetlands are lost, will refine the value of blue N burial and its contribution to coastal and estuarine water quality.

Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

Availability Statement

All data are available (Langley, 2026). Data from the CCLA (Holmquist et al., 2024) and from a marsh soil carbon database (Maxwell et al., 2024) were used herein. *Code Availability*: All code is available (Langley, 2026). The code used *R* version 4.3.3.

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