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RESEARCH ARTICLE

Land use change drives decadal-scale persistence of sediment organic carbon storage of restored mangrove

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Abstract

The restoration of blue carbon ecosystems is becoming increasingly recognised as a potentially important tool towards sustainable development because of their capacity to store carbon. However, the spatial and temporal variability of organic carbon storage is high, and the drivers regulating that variability remain poorly defined. This makes it difficult to plan habitat restoration with blue carbon objectives. Here, we quantified spatial and temporal variability of carbon storage in mangrove forests in northern Vietnam, comparing between old-growth forests, forests that have spontaneously naturally regenerated, and areas of intentional restoration. We found that sedimentation rates have increased over the past 25–50 years, but this did not translate to a general increase in carbon accumulation, suggesting a decrease in carbon burial efficiency. An exception was higher carbon burial in the restored mangrove area since the 1960s, followed by a decline since the 1990s. Microalgae were a consistent source (more than 50%) of the buried organic carbon, except in the post-1960 restored mangrove sediments - where the majority of the carbon originated from the mangroves and estuarine particulates. This suggests a shift in ecological structure towards a ‘closed loop’ of carbon recycling. Carbon burial spatial and temporal dynamics were well aligned to coastal land-use change, upstream dam construction and dry-wet climatic variability. These results suggest that both old-growth and restored mangrove forests are important carbon stores. Importantly, we show that mangrove re-establishment can have positive effects on carbon storage – supporting

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the application of habitat restoration for blue carbon-based sustainable development. However, this benefit can be transient if environmental conditions (because of natural or human factors) do not facilitate the development of ‘normal’ ecological functioning, emphasising the need for a holistic approach towards multiple conservation objectives.

Introduction

Coastal marine ecosystems are known to sequester and subsequently lock up significant quantities of carbon in their underlying sediments [1]. This so-called “blue carbon” is becoming increasingly recognised as a potentially important nature-based solution (NBS) for climate change mitigation [2], facilitated by a wide range of coastal ecosystems [3,4]. Together with geoengineering, harnessing organic carbon sediment storage could help to reduce atmospheric carbon dioxide whilst also providing multiple ecological and socioeconomic co-benefits such as improved ecosystem health, increased biodiversity, and income diversification [5,6], progressing us towards sustainable development. Our understanding of blue carbon dynamics has significantly increased in recent years, but the majority of research remains focused on quantifying sediment standing stocks. However, we know that organic carbon sediment storage can be highly variable across time and space, driven by changes and differences in landscape morphology, ecosystem function, prevailing climates, and local to regional land-use (e.g., [7–12]). Whilst knowledge of carbon standing stocks is important for present-day carbon accounting, this information cannot resolve or explain spatio-temporal variability in organic carbon sediment storage [2], limiting our ability to refine and optimise blue carbon-based conservation objectives.

Mangrove forests are considered one of the most important blue carbon ecosystems, in terms of global distribution and their carbon storage capacity – with significant potential as a national and regional scale NBS tool [13]. In addition, the success of regeneration and restoration efforts is high compared to other marine ecosystems [14]. Mangroves are therefore a promising option for achieving local-regional blue carbon policy objectives [8,15] with significant blue carbon finance potential, with a return on investment of ~US$3.7 billion per year [16]. This may help to overcome the current community hesitancy towards mangrove restoration [17]. Southeast Asia is a global hotspot for mangrove forests (where 34% of the world’s mangroves are located, c.a. 45 500 km^2) [18]. However, the region has also experienced the greatest loss over the past few decades [18,19].

Vietnam historically supported some of the largest mangrove extents (408,500 ha in 1943), but over 60% of its past mangrove forest cover has been lost since then due to human conflicts, coastal development, and environmental change [20–22]. In particular, the second Indochina war (1955–1975) [22] and economic revitalisation initiatives for shrimp aquaculture development in the 1980s [23] has led to significant mangrove loss throughout Vietnam’s coastline. In recognition of the ecological and socioeconomic detriment of mangrove loss, community-led initiatives since the mid-1990s to conserve, regenerate and restore Vietnam's mangrove forests, have resulted in an annual 6.4% increase in mangrove distribution, bucking trends seen in other parts of the world [24]. This has created a heterogeneity in Vietnam's mangrove forest maturity and composition, with concomitant variability in blue carbon "value". Mangrove restoration and regeneration is known to promote positive outcomes for blue carbon-relevant metrics such as above-ground biomass [25,26], sediment accretion rates [27] and sediment organic carbon content [28,29]. Integration of shrimp farms within mangrove forests is known to have significant negative impacts on blue carbon release around the world [30,31]. However, this also holds potential for realising sustainability objectives, such as regenerating mangrove in abandoned pond areas [30], promoting local carbon capture [29,32] and reducing aquaculture-derived nutrient pollution [33]. The majority of the organic carbon stored within mangrove forests is within the underlying sediment (up to 98% and 90% of total storage in estuarine and oceanic sites, respectively), with temporal accumulations of sediment organic carbon driven by carbon inputs, sediment characteristics and above-ground productivity [34]. Therefore, knowledge of organic carbon burial dynamics over time and space is essential for blue carbon-based conservation of mangrove ecosystems [10,29]. However, this knowledge remains limited in southeast Asia, despite recent commitments to blue carbon policies [35].

To resolve these knowledge gaps, here we investigated variation in organic carbon sediment storage in mangrove sediments over the past 300 years within the Red River delta, northern Vietnam. We compared the source and quantity of carbon buried between areas with different land use histories – including past mangrove loss, aquaculture development, natural spontaneous regeneration and contemporary intentional habitat restoration. Sediment cores from different mangrove forest types were dated using ^{210}Pb and their sediment organic carbon (SOC) isotopic composition and carbon to nitrogen (C/N) ratios determined to generate spatially resolved time series of SOC stock, OC source and C/N time series, informed by contemporary terrestrial/marine source end-members.

Methods

Study system & core sampling

Sediment cores were collected from three sites (n = 3 cores per site) within the Red River delta, northern Vietnam in May 2019 (S1 Fig), representing three different mangrove types: (1) *old-forest:* established mangrove situated within intermediate-intensity shrimp farm ponds in the centre of the delta, (2) *regenerated*: mangrove that has undergone natural spontaneous regeneration (dominated by mangrove species *Kandelia obovata, Sonneratia caseolaris, Aegiceras corniculatum,* and *Rhizophora stylosa*) towards the east of the delta, and (3) *restored:* mangrove that has been artificially restored with active mangrove seedling planting of *K. obovata* intermixed with *S. caseolaris* and *R. stylosa* at a planting density of around 20,000 trees ha^{-1} on the southern edge of the delta. Sediment cores were collected at low tide using a two-metre Geo-slice NM4 corer [36]. Core lengths ranged from 85 – 135 cm (see dataset for details of each core length). Cores were taken with the permission and support from of Management Board of Xuan Thuy National Park, Vietnam.

Core horizon sample preparation

Cores were sectioned into 5 cm horizons in the field immediately upon collection and then frozen at -20°C within 1 hour of collection. On return to the laboratory, samples were dried at 60°C until mass loss was stable. Bulk density (g cm^{-3}) was calculated as: *mass of the dry horizon (g) / volume of the horizon (cm^3)*. Once dry, each horizon was ground into a fine powder.

Core chronologies

Core horizons were analysed for ^{210}Pb, ^{226}Ra, ^{137}Cs and ^{241}Am by direct gamma assay using ORTEC HPGe GWL series well-type coaxial low background intrinsic germanium detectors. Lead-210 was determined via its gamma emissions at 46.5keV, and ^{226}Ra by the 295keV and 352keV gamma rays emitted by its daughter isotope ^{214}Pb following three weeks storage in sealed containers to allow radioactive equilibration. Cesium-137 and ^{241}Am were measured by their emissions at 662 keV and 59.5 keV, respectively. The absolute efficiencies of the detector were determined using calibrated sources and sediment samples of known activity. Corrections were made for the effect of self-absorption of low energy gamma rays within the sample [37]. Lead-210 chronologies were calculated using the CRS (constant rate of ^{210}Pb supply) dating model [38], informed by the horizon bulk densities. Carbon accumulation rates were calculated by multiplying carbon concentrations (wt%) with sedimentation rates (g cm^{-2} y^{-1}; calculated from horizon dry mass, horizon year intervals and corer cross-sectional area). Poor ^{137}Cs and ^{241}Am records precluded their use for chronology validation.

End member sampling for isotopic calibration

End-member samples of mangrove roots (from the cores) and locally caught fish were collected at the sample sites to enable fingerprinting of the SOC source. Samples were frozen within an hour of collection and kept frozen for transport to the laboratory where they were freeze-dried and ground to a fine powder. Additional end-member data was acquired from previously published isotopic studies from the local area (references [39–43]) – see S1 Table for details.

Stable isotope analysis

δ13C and δ15N were determined for all end-member samples and for each 5 cm horizon, providing a resolution of 12.0 ± 8.2 years. δ^{13}C and δ^{15}N stable isotope analyses were conducted at the SUERC Stable Isotope Ecology Facility to identify the primary source(s) of organic matter in each sediment horizon. This enabled discrimination between terrestrial / marine inputs and trophic levels via comparison to the end-member samples. Since marine organisms are distinctly less cellulosic but more proteinaceous than terrestrials, C/N ratios have also been widely used for reflecting physical mixing of sediment material composited by distinct end-member sources. A marine algal dominated source of sediment material is represented by C/N ratios of 4–10 [44], while C/N ratios >12 represent terrestrial vascular plant end-member sources [45]. All analyses were carried out using a Delta Plus XP continuous flow isotope ratio mass spectrometer (Thermo Fisher Scientific) coupled to a Pyrocube Elemental Analyser (Elementar). Three laboratory standards (Gelatine, Alanine/Gelatine and Glycine/Gelatine) were analysed for every 10 unknowns and used to correct for instrument drift during each analytical run, which typically lasts 16 hr. Four aliquots of USGS40 were analysed each day (average and standard deviation (SD) for USGS 40 over six analytical runs, spanning a period of 6 months (n = 24) was δ13C −26.35 ± 0.13‰ and δ15N −4.60 ± 0.20‰, accepted values are −26.39 ± 0.04‰ and −4.52 ± 0.06‰ respectively).

Calculation of C stock

Sediment organic carbon accumulated in each horizon (g C m^{-2}) was quantified by:

$$SOC_{horizon} = Sr \times Hi \times C$$

Where S_r = sedimentation rate (g cm^{-2} y^{-1}), H_i = horizon year interval (years) (both obtained from core chronologies), and C = carbon content (wt%), scaled up to m^2. Total carbon stocks (kg C m^{-2}) were calculated by summing carbon accumulated within in each horizon, for the past 40 cm and the past 100 years of accumulation. Error in accumulated carbon was propagated from the vertical accretion rates, obtained from the Pb-210 chronologies.

Microalgal counts

Water samples (100 mL) were collected at the three sampling sites in glass bottles, preserved with 1% acidic Lugols solution and kept in the cool and dark until analysis around a week later. Sub-samples (1 mL) from the well mixed preserved samples were enumerated in a plastic Sedgewick Rafter counting cell with glass cover slip under an inverted light microscope (x200). Initial examination of the samples determined that the diatom *Skeletonema* was highly abundant and so this diatom was counted in either a single or double row of the counting cell (i.e., from 50 or 100 µL, respectively); all other taxa were counted from the full chamber.

Isotope mixing model and statistics

Data analysis was conducted in R version 4.3.2 [46] in RStudio version 2023.09.1 [47]. The relative proportion of the different carbon sources found in the sediments was quantified using "simmr" V0.5.1.216 – a Markov Chain Monte Carlo (MCMC)-based Bayesian Stable Isotope Mixing Model [48]. simmr was run in R version 4.3.1 in RStudio version 2023.06.2. Contributions are expressed as posterior probability distributions, with a generalist prior (1:1). Both $\delta^{13}C$ and $\delta^{15}N$ were used as isotopic markers. Successful MCMC convergence was confirmed using the simmr "diagnostic" statistics (all = 1), and model fit was confirmed using the "posterior predictive check". MCMC algorithm parameters employed were: iterations = 10,000; burn in = 1,000; thin = 10; and chains = 4. Faunal end-members (excluding polychaetes which had a distinct isotopic signature) were combined *a posteriori* into an 'Other fauna' end-member group for data interpretation.

One-way ANOVAs were used to investigate differences in carbon parameters between sampling sites. Where significant site effects were found, post hoc Tukey test comparisons were used to determine differences between sites. Data met assumptions for parametric testing. Volumetric counts of microalgae were Log(X + 1) transformed for calculation of Bray-Curtis similarity in PRIMER 7 (v.7.0.23) and analysed with a non-metric Multi-Dimensional Scaling ordination (2D stress = 0.08). Statistical differences between microalgae counts for abundant taxa were assessed in SigmaStat v4.0 through Holm-Sidak t-tests.

Results

Sedimentation rates

Vertical accretion rates in the regenerated and restored mangroves were stable at <1 cm y^{-1} until around the year 2000, after which vertical accretion rates accelerated to >1 cm y^{-1} (Fig 1a). Vertical accretion rates in the shrimp farm-associated old-forest mangroves were similarly low and stable until ~1980, after which vertical accretion rates accelerated to >4 cm y^{-1}. Contemporary vertical accretion rates were higher in the old-forest mangrove than the restored or regenerated mangroves. A similar pattern of recent increases in sedimentation rates were also observed (Fig 1b), preceded by decadal-scale 'waves' of higher sedimentation rates around 1940 (regenerated mangrove area only) around 1990 at all three sites (Fig 1b). No significant differences in sedimentation between mangrove types was identified, for the top 40 cm of sediment ($F_{2,21} = 0.634$, $p = 0.540$) nor the past 100 years of deposition ($F_{2,21} = 0.904$, $p = 0.414$) (Fig 1c).

Organic carbon content

Total organic carbon (TOC) was relatively stable at 0.6-0.9 wt% in the old-forest and regenerated mangroves throughout the time series, excepting an enrichment in the surface horizon of the regenerated mangrove (Fig 2a). The restored mangrove had comparable TOC content until ~1960, after which TOC rapidly increased to a peak of 1.8 wt% in the 1990s (Fig 2a), coinciding with the mangrove restoration efforts. The elevated TOC content translated to higher carbon accumulation in the restored mangrove that were maintained for around 20 years (Fig 2b). Total carbon accumulated in recent years was significantly higher in the restored mangroves compared to the other mangrove types, both in the top 40 cm ($F_{2,21} = 6.297$, $p = 0.007$) and over the past 100 years – where the regenerated mangroves were also significantly lower than the old-forest mangrove ($F_{2,21} = 98.72$, $p < 0.001$; Fig 2c).

Source of organic carbon

C/N elemental ratios lay at the transition between terrestrial plants and marine algal signals with no clear temporal patterns (Fig 3a). Average C/N ratios were slightly elevated in the restored mangrove (throughout the whole time series; Fig 3a), suggesting a higher contribution of plant-derived carbon. $\delta^{13}C$ variation over time was generally stable around -20‰

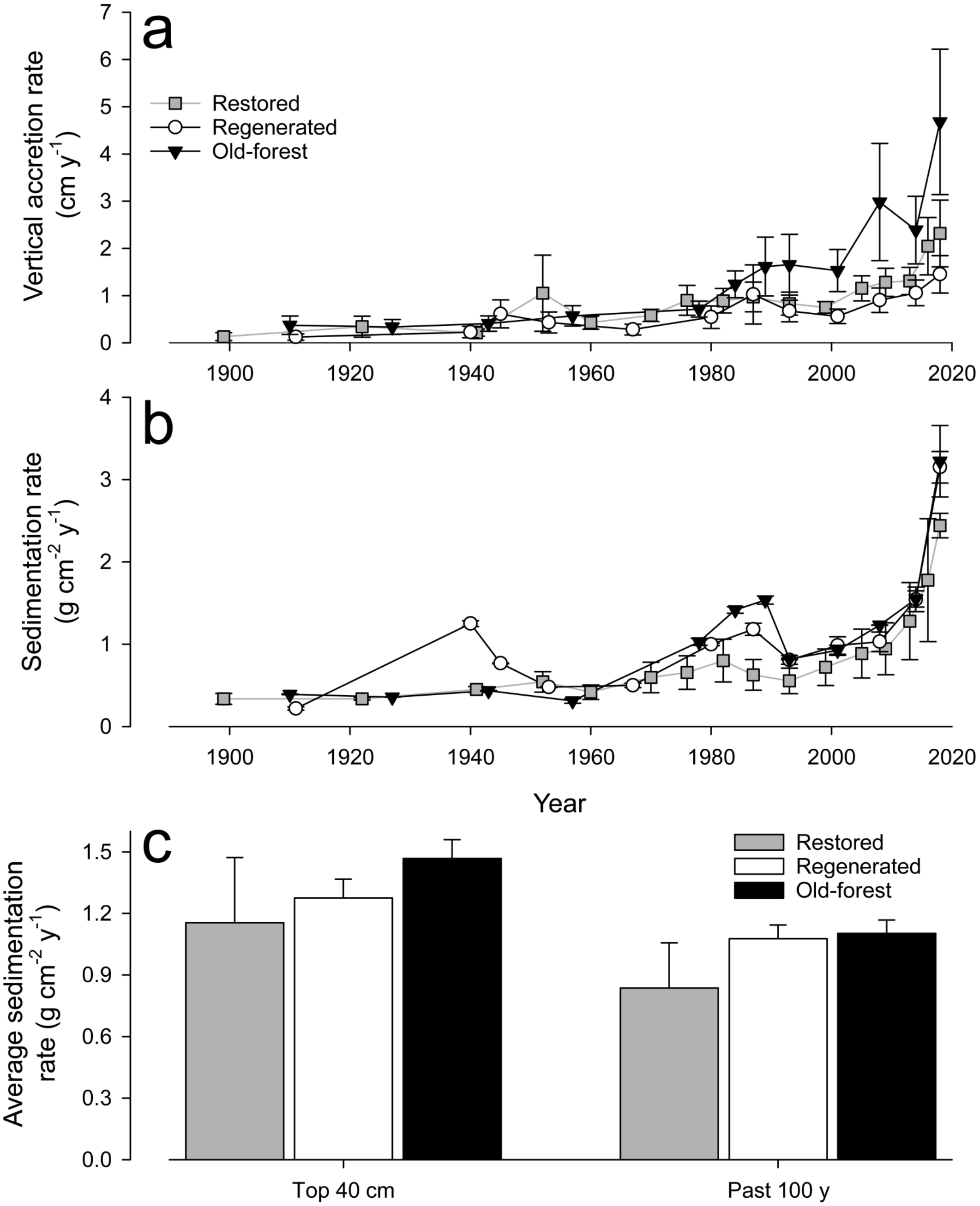

Fig 1. Sedimentation in the mangroves over time. (a) vertical accretion rate (cm y^{-1}) and (b) sedimentation rate (g cm^{-2} y^{-1}) over time from restored, naturally regenerated and old-forest mangrove types (n = 3 cores per mangrove type), and (c) average sedimentation rate in the top 40 cm (n = 8 horizons; black bars) and over the past 100 years (n = 10-16 horizons; grey bars). Data presented as mean±range (a,b) and mean±SE (c).

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in both the old-forest and regenerated mangroves (Fig 3b). $\delta^{13}C$ was similar in the restored mangrove until ~1960, after which a distinct decline in $\delta^{13}C$ was observed; this temporal shift was consistent across all cores (Fig 3b). $\delta^{15}N$ variation over time was generally stable over time (between ~3–5‰ at all sites) with no clear temporal groupings (Fig 3c). Baseline end-members provided clear $\delta^{13}C$/ $\delta^{15}N$ distinction between marine/ terrestrial inputs, and across trophic levels (Fig 3d).

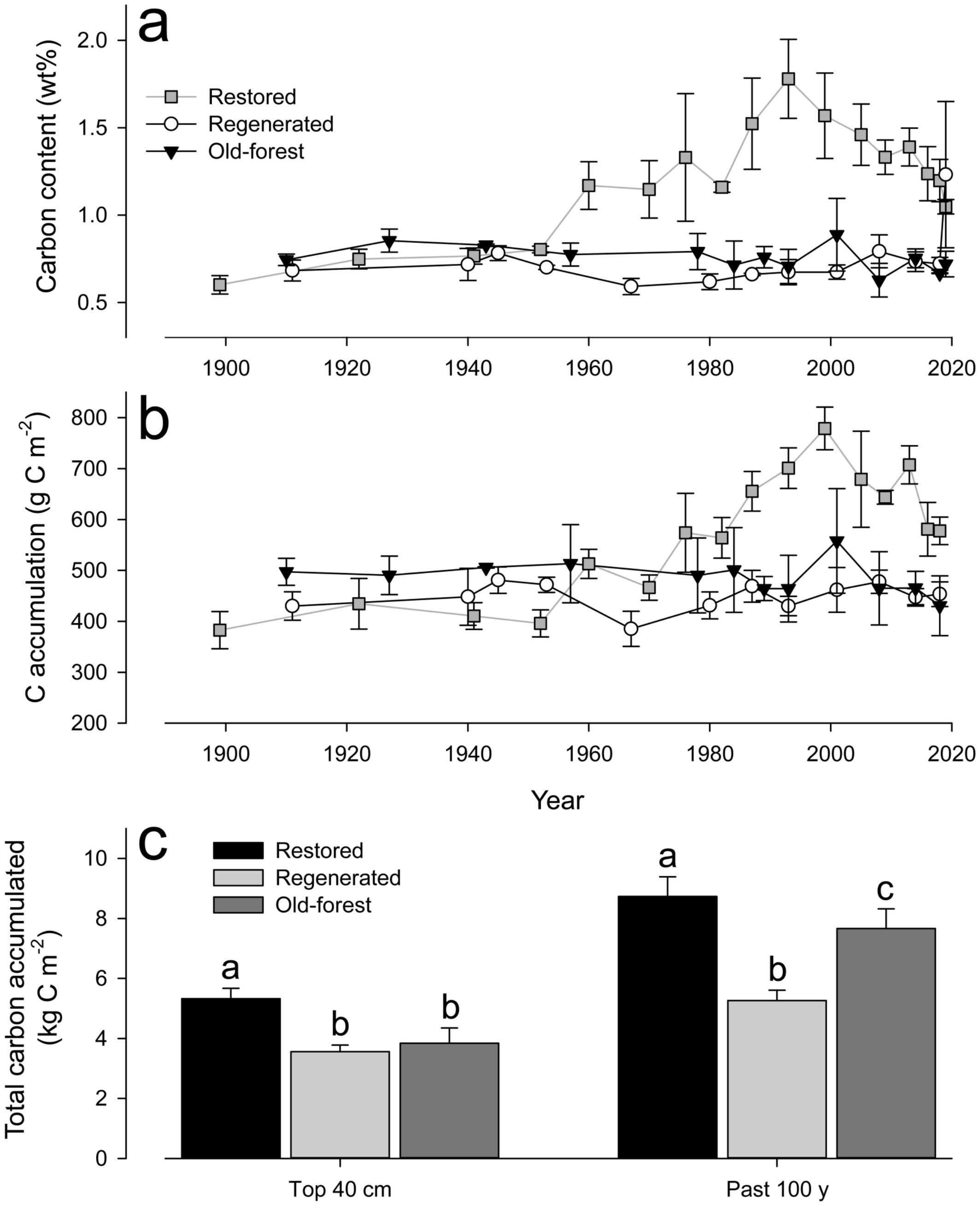

Fig 2. Total organic carbon (TOC) within mangrove sediments. (a) sedimentary TOC (wt%) over time and (b) carbon accumulation (g C m^{-2}) over time from restored, naturally regenerated and old-forest mangrove types (n = 3 cores per mangrove type), and (c) total TOC in the top 40 cm (n = 8 horizons; grey bars) and over the past 100 years (n = 10-16 horizons; black bars). Letters over the bars in (c) represent significantly different groupings. Data presented as mean±SE.

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In the old-forest mangrove, regenerated and pre-1960 restored mangroves, benthic microalgae contributed >50% of the sedimentary organic carbon (Figs 4 and S2). Non-polychaete fauna and macroalgae accounted for another 35–37% of the organic carbon in the old-growth and pre-1960 artificially restored mangroves (Figs 4

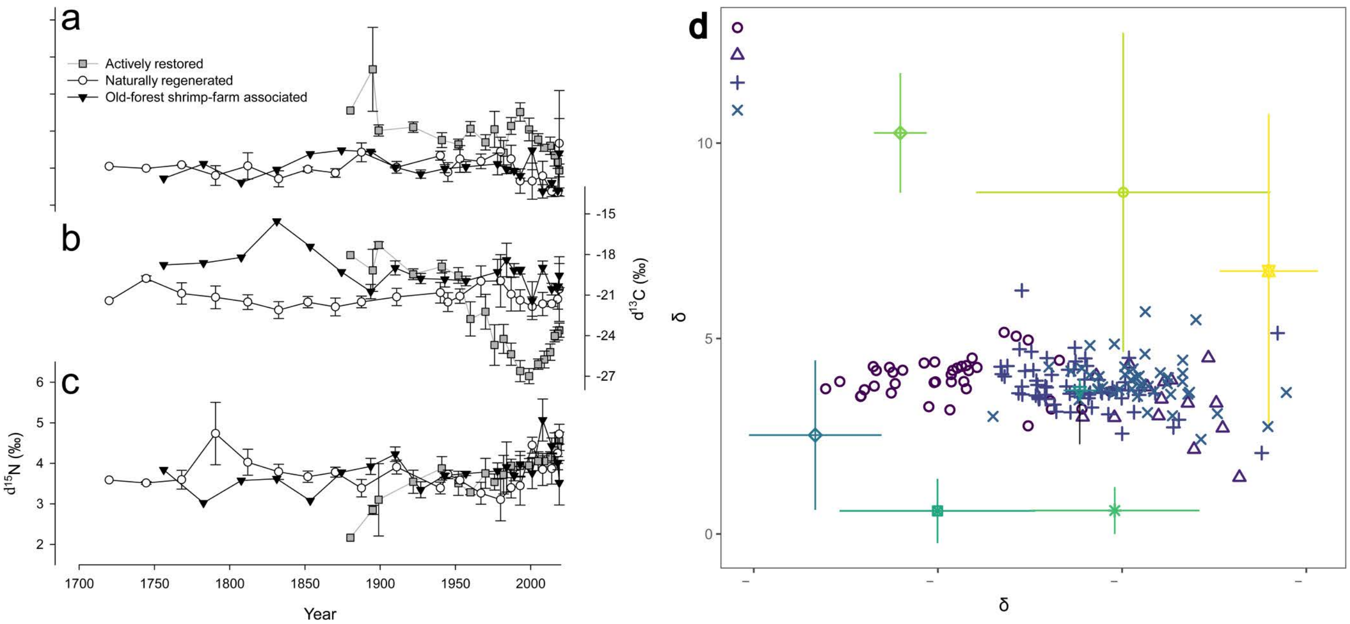

Fig 3. Source of sedimentary organic carbon. Time series of (a) C/N ratio, (b) δ13C (‰) and (c) δ15N (‰), and (d) δ13C against δ15N (‰) of sedimentary organic material from each core horizon from restored, naturally regenerated and old-forest mangrove types (n = 3 cores per site), and contemporary end-members (isotopic values from the literature and this study - see S1 Table). POM = particulate organic matter. Data presented as mean±SE.

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and S2). In the spontaneously regenerated mangroves, fauna (polychaetes + other) accounted for another 33%, with mangrove (6%) and macroalgae (4%) providing minor contributions (Figs 4 and S2). However, organic carbon from post-1960 sediments in the restored mangrove was characterised by a distinctly different source origin (Figs 3, 4 and S2). The source of this organic carbon was more diverse, consisting of: inland-derived particulate organic matter (28%), mangrove (21%), benthic microalgae (21%), polychaetes (19%) and other fauna (9%) (Figs 4 and S2).

Microalgae and protist communities

Site-specific microalgal and protist communities were observed at all sites (Fig 5). The greatest distinction was observed in the old-forest mangrove (Fig 5), driven by a significantly lower abundance of diatoms (Fig 5). *Skeletonema* spp. accounted for 99% of the diatom microalgae at all sites; other marine and freshwater diatom taxa (e.g., *Navicula, Cyclotella, Achanthes*) represented <2% of total counts. Other numerically abundant taxa included small protists such as planktonic ciliates (mostly *Strombidiida*), and the Euglenoids (freshwater flagellates) (S3 Fig). These microalgae groups did not statistically differ between mangrove types.

Discussion

The storage of organic carbon in coastal ecosystems is becoming recognised as a useful nature-based tool to achieve sustainability, with substantial co-benefits. Here, we identify that the high spatiotemporal variation in carbon stored in mangrove forests in northern Vietnam is driven by local coastal development, upstream dam construction and long-term climatic change. We identify two priority areas for blue-carbon focused sustainability policy: the preservation of old-forest mangrove as a persistent organic carbon sediment store, and the sensitivity of mangrove restoration success to external human or climatic forcings.

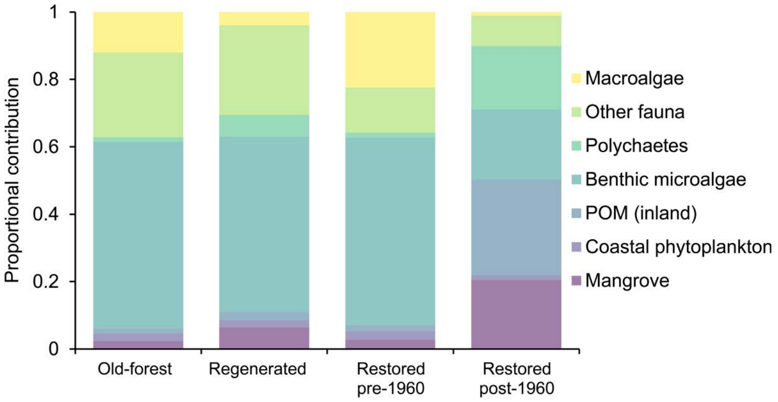

Fig 4. Mean proportional contribution from each end-member group across the mangrove types.

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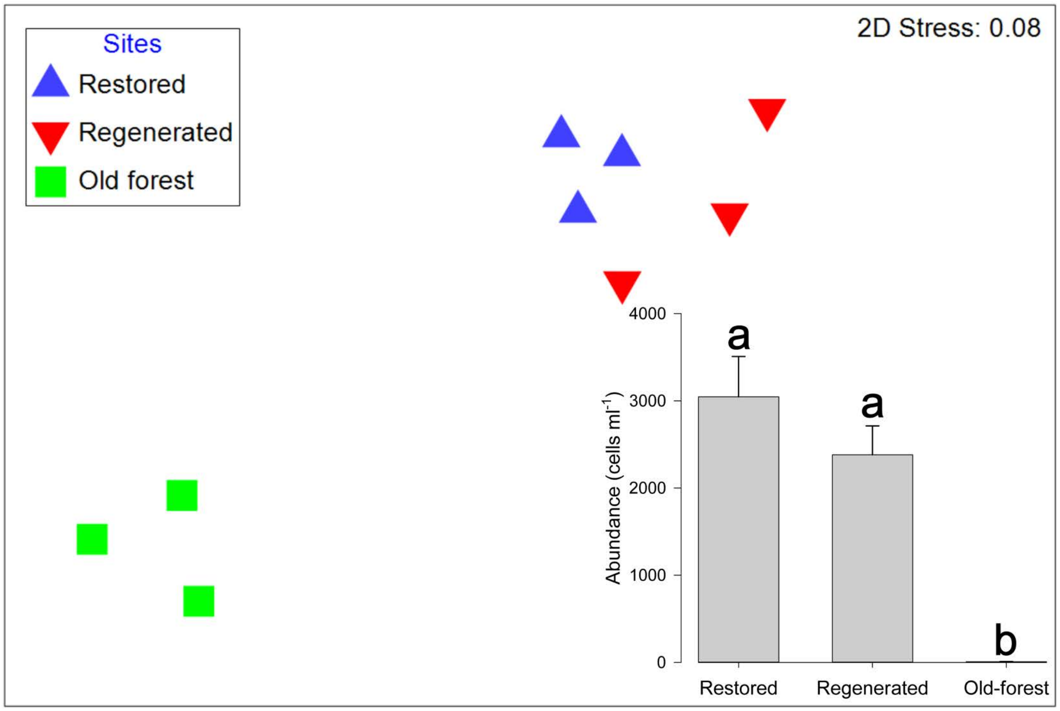

Fig 5. Microplankton community composition of surface waters. Ordination (nMDS) of community composition by mangrove type, and (inset) total diatom abundance (99% Skeletonema spp.). Letters above bars indicate significantly ($p < 0.01$) different groups based on Holm-Sidak t-tests. Note that the relative distance between samples in the nMDS relates to the percentage (dis-)similarity in community composition.

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Delta-wide increase in sedimentation rates over the past 50 years

Sediment accretion is a major factor for future survival of mangrove forests due to submergence risk from sea level rise [49]. For the majority of our record, sedimentation rates in the mangrove sites were consistently less than 1 cm y^{-1}, which is comparable to previous studies on bulk carbon storage in Vietnam mangroves [50]. From around 2010, sedimentation rates in the regenerated and restored mangroves have accelerated, aligning with mangrove re-establishment in these areas. Mangrove re-establishment therefore appears to have had a positive effect on sediment stabilisation and particulate entrainment. This effect is amplified in the shrimp-farm associated old-forest mangrove: sedimentation rates began to increase in the 1980s (when shrimp farm intensification started), and sedimentation rates have since been consistently double that of the regenerated and restored mangrove areas – suggesting that the shrimp farms provide a higher source of particulates for entrainment within the mangroves. Sustainable associations between sediment-producing aquaculture and mangrove forests may therefore provide respite from the submergence threat posed by projected climate change [49]. This supports the emerging integration of mangroves with aquaculture for ecosystem-based management with greater ecological co-benefits [51–53].

The source and storage of carbon is mostly stable over time

Although sedimentation rates in the shrimp-farm associated mangroves have been consistently higher since the 1980s, this did not translate to higher amounts of stored organic carbon – suggesting the shrimp farm-derived particulates are low in carbon content and/ or there was lower carbon burial efficiency, perhaps because of higher mineral proportions in the sediment, surface remineralisation and carbon dioxide flux [54]. This provides local support to global observations that mangroves with high sediment loads and/ or strong tidal flushing accumulate smaller carbon stocks [55]. Sediment carbon storage was comparable to other areas in Vietnam [25,50] and was within the global range (86–729 Mg C ha^{-1}) [55]. Surprisingly, the development of shrimp farms did not change the source of the buried carbon, suggesting 'natural' carbon burial dynamics were maintained. The source of organic carbon, as determined by C/N ratios and $\delta^{13}C$ - $\delta^{15}N$, were consistently characteristic of a marine flora/fauna mix for at least the past 125 years. The isotopic signatures were particularly characteristic of microalgal-derived carbon (albeit with differing contemporary compositions between sites), accounting for around half of the organic carbon origin. Contemporary microalgal abundance suggests this may derive especially from the bloom-forming diatoms *Skeletonema* spp., at least in the restored and regenerated mangroves. The only deviation to this was a peak in $\delta^{13}C$ in the early-mid 1800s (estimated from chronology extrapolation; S4 Fig) in the old-forest mangrove, suggesting a temporary increase in macroalgal-derived carbon and accompanied by lower carbon accumulation. This occurred after a prolonged period of reduced rainfall and repeated drought events in northern Vietnam [56,57] – driven by the 18th century Indochina "mega-drought" and El Niño-like elevation of sea surface temperatures in the tropical Pacific [58]. A return to lower sea surface temperatures and higher rainfall (and concomitant increase in river discharge) may have (1) promoted favourable conditions for macroalgal blooms within the Red River delta and (2) reduced allochthonous carbon supply due to reduced river flow.

Mangrove re-establishment increases autochthonous carbon storage

A distinct and more persistent shift in carbon dynamics was identified in the regenerated mangrove from the 1960s – characterised by an increase in the amount of carbon stored and a shift in where that carbon came from (evidenced from both the C/N and stable isotope ratios). This indicates an increase in carbon burial efficiency due to increased recalcitrant carbon supply [59], perhaps due to (1) an increase in autochthonous carbon supply (mangroves) and (2) an increase in mineral-associated organic matter (inland POM). Post-1960 carbon standing stock in the restored mangrove area was subsequently higher than the other mangrove regions, suggesting a temporarily positive effect on carbon storage. However, the elevated carbon trend has been lost since the 1990s – coinciding with the operation of the Hoa Binh dam and a 61% decline in sediment flux towards the Red River delta [60]. By 2021, upstream damming had contributed to a 91% reduction in annual sediment load [61]. Although re-establishment of mangroves seems to result in comparable carbon storage [11], this can take 10 + years [62]. However, our results show that decadal persistence of mangrove carbon can be highly affected by aquatic developments hundreds of kilometres away (e.g., upstream river damming), placing the carbon benefits of re-establishment efforts at risk. The increased proportion of mangrove-derived carbon suggests that the ecological dynamics of this area may have been affected, creating a 'closed loop' of carbon recycling, with more plant-derived carbon and less external carbon input.

Conclusions: Implications for habitat restoration and blue carbon policy

Spatiotemporal variability in carbon storage in blue carbon ecosystems, and the drivers of those changes are important considerations for current and future blue carbon sustainability policies. In particular, our results indicate that land-use change appears to have significant effects on sedimentation rates, the amount of carbon to be stored, and the source of that carbon. Importantly, these effects appear to be persistent to at least decadal timescales. Quantification of the complete carbon budget would place this carbon storage in a wider biogeochemical context – although the impact from methane emissions appears to be less important in mangrove forests than previously thought [8]. Most notably though is the potential transiency of carbon storage benefits from mangrove restoration because of external drivers. This has important implications for restoration strategies and blue carbon sustainability policy. Firstly, our results support the continued conservation of old-forest mangrove as a carbon store that is stable and persistent over decadal-centennial scales, accompanied by a myriad of other co-benefits. Secondly, a closure of the carbon recycling loop to external inputs during mangrove restoration creates an opportunity for well-defined conservation objectives that work with local to cross-border development. However, this apparent change in ecological habitat function also raises questions about the scope of associated co-benefits. In the context of blue carbon sustainability policy, this should be taken into account when, for example, valuing blue carbon credits for areas of new mangrove restoration vs preservation of old-forest mangrove. Consideration alongside other blue carbon habitats, and also taking into account their long-term spatiotemporal dynamics, will provide a holistic coastal-wide opportunity for widescale implementation of blue carbon policy with environmental, societal and economic benefits.

Supporting information

S1 Fig. Core sampling locations in the Red River delta, northern Vietnam. Base map and data from OpenStreetMap and OpenStreetMap Foundation.
(PDF)

S2 Fig. Boxplots of the proportional contribution from each end member group in each mangrove type.
(PDF)

S3 Fig. Microplankton abundance of surface waters from the three mangrove types. Two microalgal groups are shown: Ciliates (left) and Euglenoids (right). No significant difference between mangrove types for either group.
(PDF)

S4 Fig. Source of sedimentary organic carbon. Time series of (a) C/N ratio, (b) δ13C (‰) and (c) δ15N (‰) of sedimentary organic material from each core horizon from restored, regenerated and shrimp farm-associated old-forest mangrove types (n = 3 cores per site), and contemporary end-members (isotopic values from the literature and this study - see S1 Table). Pre-1900 dating extrapolated from the post-1900 ^{210}Pb chronologies. POM = particulate organic matter. Data presented as mean±SE.
(PDF)

S1 Table. Contemporary material δ^{13}C/ δ^{15}N isotope signatures used as end-member carbon sources in the simmr Stable Isotope Mixing Model. Data presented as mean±SD.
(PDF)

Author contributions

Conceptualization: Heidi Burdett, Michel J. Kaiser, Nguyen T.K. Cuc.

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