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A re-evaluation of wetland carbon sink mitigation concepts and measurements: A diagenetic solution

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Abstract

The capacity of wetlands to mitigate greenhouse gas (*GHG*) emissions is the sum of two services: the protection of vulnerable organic stocks from remineralisation, and the capacity to sequester *GHGs* relative to their anthropogenic replacements. Organic carbon accumulation (*CA*) down through the sediment column is often taken as the measure of sequestration because of its capacity to record long-term variability and trends. However, we demonstrate that: i) *CA* is not equivalent to sequestration as net ecosystem production (*NEP*) for open systems; it requires the subtraction of the initial deposition rate of labile allochthonous carbon sources; ii) *CA* also requires subtraction of intrinsically allochthonous recalcitrants down through the sediment column, and together with subtraction of autochthonous recalcitrants from organic stock services; iii) *CA* as a climatic mitigation service also requires a diagenetic correction, as the annual deposition of labile organic carbon continues to remineralise over the long-term; and iv) preserving of a wetland has a significantly greater mitigation potential than restoring one. To address the above concerns, a global diagenetic solution is proposed, applied and tested for a tropical seagrass and mangrove. As expected, traditional *CA* estimates were disproportionately larger than their respective diagenetically modelled *NEPs*, and together with stocks fell within the ranges reported in the literature, with a final carbon accreditation highly dependent on the choice of their anthropogenic replacements. The review demonstrates that mitigation concepts and measurements for natural carbon sequestration solutions require re-evaluation to avoid *GHG* emissions above their capacity or reduce the ability to fulfil emission targets.

Keywords: blue carbon · teal carbon · black carbon · carbon accumulation · carbon sequestration · carbon stocks · net ecosystem production · allochthonous recalcitrants · autochthonous recalcitrants ·protection · restoration

Introduction

Anthropogenic greenhouse gas (*GHG*) emissions are largely responsible for climate change and will likely disrupt both society and global ecosystems (Hoegh-Guldberg et al. 2018). This has led to calls to mitigate these emissions (UNFCCC, 2015). Several mitigation practices have been suggested, ranging from increasing the efficiency and reducing reliance on the burning of fossils fuels, carbon capture storage, and the restoration and conservation of existing natural carbon sinks (Lal 2008). However, there is also a realisation that relying on altruism may not be sufficient and mitigation requires incentives. Carbon trading is one such mechanism. It can be used to fund projects and move the costs of carbon offsets to industry towards more efficiency or reliance on fossils fuels. In contrast, to engineered processes such as carbon capture and storage, natural solutions in the preservation or restoration of a healthy ecosystem and its associated carbon sinks have the necessary vigour, resilience, and self-organization to sustain themselves (Costanza & Mageau 1999; de Paoli et al. 2017). Consequently, there has been a push for extensive replanting and protection of terrestrial forests worldwide (Mackey et al. 2008).

While valuable, forests support a relatively low carbon production density (Lal 2008) and are vulnerable to fire. These constraints are not as apparent for flooded wetlands both inland marshes and coastal canopy systems – mangroves, seagrass, and salt marsh (Duarte et al. 2010). Although for peatlands, combustion of biomass and soils is of growing concern as their water tables fall (Gaveau et al. 2014). Furthermore, while inland wetlands — Teal carbon systems (Zinke 2020) – occupy an area between 2% to 5% of the land but store significant fraction of between 20% and 30% of organic carbon of the terrigenous landscape (Kayranli et al. 2010). Similarly, coastal canopy wetlands – Blue carbon systems (Zinke 2020) – occupy < 2% of the marine seascape, and are estimated to contribute around 50% of the ocean's organic carbon storage, largely within an accreting soil-sediment column (Duarte et al. 2005). Although the magnitude of this claim has been disputed (Johannessen and Macdonald, 2016). This makes wetlands valuable and potentially manageable but also vulnerable given their relatively small areas. Understandably, an accurate assessment of these ecosystems' mitigation potential is required for any carbon offset standard. This would seem a requirement to see real mitigation impacts but also for confidence in market-based and national compliance schemes. While ground-based surveys and satellite observations can provide accurate accounts of their extent, however, we contend that misunderstandings and errors have arisen in the foundations behind wetland mitigation services. First, there is an implicit and untested assumption that total sedimentary organic carbon accumulation is a proxy for carbon sequestration. Second, the majority of carbon sink assessments still fail to subtract a full range of possible intrinsically recalcitrant allochthonous or autochthonous carbon from stocks and sequestration estimates (Chew and Gallagher 2018; Rillig 2018). Third, the traditional conceptual model does not account for the remineralisation of its sedimentary organic carbon over climatic time scales (Chuan et al. 2020; Maher et al. 2017). Fourth, major reviews continue to equate an ecosystems' organic carbon stocks as the measure of its mitigation service without reference to a likely anthropogenic replacement (e.g. Macreadie et al. 2021; McLeod et al. 2011), although see wetland Verified Carbon standards (Needelman et al. 2018). Fifth, Verified Carbon Standards do not include the difference in their net carbon balances (e.g. Siikamäki et al. 2013; Tokoro et al. 2014), they appear to consider only the gain or loss of wetland carbon stocks, for their respective restoration or preservation, as a mitigation service (Needelman et al. 2018).

Aims

The article aims to review the measurement of carbon stock and sequestration concepts and clarify the additional constraints needed to quantify wetland carbon accumulation and stocks as a mitigation service. The currently-favored methods for stock and sequestration are outlined; constraints and methods discussed, and sequestration misconceptions addressed for different circumstances for systems both young and mature, while closed and open to allochthonous inputs. We then demonstrate what would be required to measure true sequestration and stock variability over climatic time scales from the sedimentary record, and argue for an additional mitigation concept for future discussion and testing. Finally, these concepts are implemented and evaluated using two disparate examples, namely, a tropical submerged seagrass meadow and an intertidal mangrove wetland; both open to different amounts and forms of allochthonous organic carbon.

We also acknowledge, but do not address other factors that can be considered as an adjunct or constraint to a wetlands' proclivity to sequester carbon. These include the uncertain fate of organic carbon export and its downstream impacts (Prairie et al. 2018), the uncertain roles of CO_2 production and loss during biogenic and geogenic calcium carbonate formation or redissolution between their respective canopy and anthropogenic non-vegetated replacements (Howard et al. 2018; Mitsch et al. 2013), and the biogenic production and emissions of other greenhouse gases other than CO_2. These have already been addressed as emission factors within the carbon accreditation process (Needelman et al. 2018). Finally, no consideration is given to the inputs of dissolved inorganic carbon (DIC) from adjacent rivers (Chapin et al. 2006), and coastal upwelling. While these may also play a role in the ecosystems' carbon balance to the atmosphere (Tokoro et al. 2014), they are also ecosystem site parameters. Consequently, they are largely independent of the differences between biological drivers of sequestration relative to their anthropogenic replacements.

The role of sediment carbon stocks in mitigation

Within the confines of the voluntary and compliance carbon markets, a wetland's ability to mitigate carbon emissions is considered as the loss of organic stock as a preservation service, or a gain of stock to justify restoration (Needelman et al. 2018). Translated as a sequestration service, that loss or gain is the time it takes for its anthropogenic replacement or restoration to establish itself at a steady-state. Or more succinctly, where any variance is described along a stationary time series, the length of which will ultimately respond to ongoing climate variability (Gallagher 2017; Marba and Duarte 1997). However, not all organic stocks are vulnerable to remineralisation (Jennerjahn 2020). Leaving aside a case-by-case fate and loss of biomass, for example, mangrove wood may be stored as a product or burnt for charcoal (Eong 1993). The fate of the remaining sedimentary stocks is largely determined not only by their intrinsic vulnerability to remineralisation, between 25% and 100% (Pendleton et al. 2012), but to a depth of disturbance and oxidation. Once the vulnerable stock fraction and its extent have been estimated (gC m^{-2}), the values are then transformed to a vector quantity (gC m^{-2} yr^{-1}) in the time it takes for the vulnerable fraction to be remineralised. The time has been set as a default of 20 years (IPCC 2014) to reflect the period over which the replacement ecosystem will establish itself at steady state equilibrium.

For a first-tier assessment, the depth of disturbance is commonly set to a maximum of 1 m irrespective of its total column depth. The depth of disturbance and the extent of remineralisation, however, will ultimately depend on its type of anthropogenic replacement (Siikamäki et al. 2013). For example, a mangrove forest can lose up to 1.5 m to 3 m of sediment to aquaculture ponds, piled up and directly exposed to the weather on top of its banks (Järviö et al. 2018). Alternatively, the loss of a natural seagrass meadow’s canopy to disease has directly progressed to a rocky barren ground state (Wilson 1949). It is also conceivable that the wetland’s anthropogenic replacement may support a larger organic stock inventory. For example, mangroves have encroached onto salt marshes due to sea-level rise (Rogers et al. 2005). Ironically, preserving the original salt marsh in the face of anthropogenically driven sea-level rise could potentially exacerbate and not mitigate *GHG* emissions. However, it could also be argued that the replacement mangroves are also potentially vulnerable to anthropogenic decisions, which could lead to their clearance (Chee et al. 2017). Under these circumstances, preservation of the threatened salt marsh (SMS) stock inventory as a means to justify preservation from an encroaching mangrove system could be justified. That is to say, the value in actively preserving the salt marsh is the difference in stocks gained as the salt marsh (SMS) is replaced by the encroaching mangroves (MS) and relative to a likely future disturbed mangrove state (DS) of a smaller stock inventory (i.e. (MS – DS) – SMS; where SMS < MS and DS < SMS). For mangroves, such pressures may come from the expansion of shrimp pond aquaculture with a loss of sequestration capacity and stocks to a depth of the ponds (Chen et al. 2016), or even a complete loss of an ecosystem sink to property development (Chee et al. 2017).

Measuring carbon stocks.

The attraction of focusing only on standing stock measurements comes from their relative simplicity. This is an important attribute to capture the inherent variability for some systems (Hu et al. 2021). For example, there is a simplicity in that estimates of tree biomass can be made using accessible allometric width, carbon content, and bulk density parameters. Furthermore, the timing for both biomass assessments is usually not that critical during the timespan of a year or more, with the possible exception of the faster-growing and reproducing r-strategist macrophytes adapted to a relatively high rate of disturbance. This is because it would take decades to a century for accumulation to contribute a significant fraction of the stock inventory to 1 m depth (Wilkinson et al. 2018), and are not likely to change significantly between years irrespective of the canopy species. Although, researchers must remain cognisant of deposition and erosion events (Gallagher and Ross 2017) or seasonal changes in the direction of allochthonous supply and export for coastal systems (van Keulen and Borowitzka 2003). Furthermore, after taking sediment cores, a selected number of horizons of a few cm thick can easily be sampled or physically mixed to integrate variation with depth. Either way, accurate dry bulk densities are required before the sediment samples are dried, should be also normalised for any intertidal variation for those coastal systems from either their water, mineral, and organic matter content (Binford et al. 1990) or volumetric determinations sampled to stop compression. Compression is typically minimised by using a cut off syringe resembling a piston corer. Only after drying the analysis for carbon content (i.e. the percentage dry weight) can be handled through more specialised laboratory services. Stocks then become the product of the dry bulk density and content. This service can be readily extended to the analysis of organic stable isotope signatures to estimate the fractions of different sources (Gonneea et al. 2004). Alternatively, a less resource-intensive approach uses organic carbon proxies. These can be previously determined dry bulk density (Callaway et al. 2012), gravimetric losses after combustion, chemical oxidation and titration (Byers et al. 1978; Heiri et al. 2001), or infrared reflectance (Bellon-Maurel and McBratney 2011). Except for chemical titration, the above proxies to some degree require a global (Fourqurean et al. 2012) or preferably regional (Craft et al. 1991) calibration with standard methods, with the inclusion of regression variance in the final estimate (Gallagher et al. 2021a).

Limitations and misunderstandings of the stock concept

Intrinsically recalcitrant organic carbon produced within or outside the ecosystem is not vulnerable to remineralisation after disturbance and so, do not contribute to a carbon stock mitigation service. In other words, only sedimentary carbon stocks vulnerable to remineralisation will release *GHGs* after disturbance and thereby only their continued presence can be considered as a mitigation service, irrespective of their origin. However, removing intrinsically autochthonous from the stock calculus has not yet been adopted other than recognition by the IPCC for allochthonous recalcitrants (Bindoff et al. 2019). Arguably, the most ubiquitous of the intrinsic recalcitrants is black carbon (BC), for which, there is very little impediment to quantifying this component. The carbon content can be estimated after thermal or chemical isolation within a sedimentary matrix using standard laboratory equipment (Chew and Gallagher 2018). Carbon and N isotopic signatures can then be used to assist in identifying their allochthonous or autochthonous nature (Gallagher et al. 2021a; Leorri et al. 2018). Nevertheless, BC estimates across coastal wetland ecosystems remain globally under sampled'm. This is despite BC contributing substantial fractions to the sediments' total organic carbon (TOC) (means between 3-38%) (data from Chew and Gallagher 2018; Gallagher et al. 2021a; Gallagher et al. 2019). The variability reflects the supply of labile organic supply, relative to local pure BC supply from atmospheric deposition, advective loss from the canopy before deposition, and any intertidal loss after dissolution (Gallagher et al. 2021). Estimates across freshwater wetlands, however, are scarce but may still be significant. Sediment and soil BC fractions between 35.9% and 41% of its TOC have been reported within an industrial setting for ponds and paddies (Song et al. 2002).

Along with BC, other intrinsically recalcitrant forms may require evaluation. These are kerogens, geologically old organic materials washed in from catchment shales, and plastics. Unfortunately, wetland measurements of sedimentary kerogens are restricted to a shallow pond and paddy field located within an industrial region (reportedly 24% and 30.8% of TOC respectively; Song et al. 2002). Similarly, information on the importance of microplastics is limited. Nevertheless, Rillig (2018) pointed out that microplastics in soils are being disguised as carbon storage because of their allochthonous recalcitrant nature. Indeed, within industrial soils, the plastic content can be substantial (6.7% dry wt) and remains measurable between 0.1−5% of carbon content for soils away from direct human influence (Rillig 2018). However, for wetland sediments, no data is available. Nevertheless, during a sediment coring program, the author (JBG) has observed the presence of plastic bags buried within surface sediments of an intertidal urban seagrass meadow (Middle Bank, Penang, Malaysia). Within the same region, macroplastic debris from 162 to 6763 items 100 m^{-2} was also observed to both cover and be retained on the surface sediments of urban and peri-urban mangrove forests (Chee et al. 2020). Along with intrinsically recalcitrant material, labile material can become effectively recalcitrant from physical protection. This can occur within a sedimentary clay matrix. However, estimates put this association to < 5% of the TOC (Needelman et al. 2018). Furthermore, it is unclear whether such associations remain recalcitrant when subject to continuous resuspension (Cathalot et al. 2013). Physical protection is also manifested from occluded carbon contained within the glassy structures of plant phytoliths. Within non-alkaline soils, phytolith-occluded carbon can represent most of the remains of organic matter (Parr & Sullivan 2005). However, for aquatic systems, data is limited to two tropical river-estuarine systems, where biogenic silica within the water column or non-vegetated surface sediments is dominated by phytoliths (Cary et al. 2005; Zang et al. 2016).

Carbon sequestration

Sequestration occurs when the rate of ecosystem CO_2 fixation exceeds the community respiration from the consumption of available organic carbon. When the biomass is at a steady state, the excess organic carbon accumulates largely down a relatively rapid accreting sediment column (Prairie et al. 2018). The balance is referred to as the net ecosystem production (*NEP*). Indeed, the shallow canopy can stabilise and preserve these sedimentary remains as stocks, which makes these ecosystems so valuable in comparison to terrestrial forests (Mcleod et al. 2011). Like stocks, the amount sequestered should be compared to its anthropogenic replacement (Eq. 1). Together, both stock vectors and *NEP* concepts appear to describe how the preservation in avoiding the loss of these canopy wetlands can restrain *GHG* emissions (Siikamäki et al. 2013).

$$Cmit\ = (NEP + \frac{\theta}{\varphi_1} Cstock) - (NEPRr + \frac{\theta}{\varphi_2} Crstock) \tag{1}$$

Eq. 1 Where *Cmit* is the *GHG* carbon mitigation service and *NEP* the annual atmospheric carbon dioxide sequestration rate, positive for uptake; *Cstock* represents biomass and sedimentary organic carbon stocks that have accounted for BC, with θ the vulnerable fraction likely to be remineralised over a time φ to attain an equilibrium should the ecosystem be degraded or destroyed; *NEPRr* is the atmospheric carbon dioxide sequestration rate of the replacement ecosystem, and *Crstock* the organic carbon stock of the replacement ecosystem that has accounted for *BC,* with θ is the fraction vulnerable to remineralisation after destruction or disturbance, over time φ for each circumstance. As equation (1) stands, it provides no information on what determines the extent of the balance other than the net productivity of the plant. The carbon use or consumption will depend on the innate digestibility of the plant (Cebrian 2002) and any labile allochthonous material supplied to the ecosystem. Whereas for the flooded sediment deposits, the innate digestibility is constrained by the ‘aging’ of the organic mix within the sediments, and physical protection associated with the clay fractions (Burdige 2007). Clearly, unlike stocks, sequestration is more valuable for an ecosystem capable of both directing CO_2 towards the production of autochthonous recalcitrants and the protection of labile fractions. Although it should be noted that gaseous emissions in the production of autochthonous BC would constrain its mitigation service (Santín et al. 2015). As in the case of stocks, allochthonous recalcitrants continue to play no role in the sequestration calculus; and to include these recalcitrants would amount to double-accounting across ecosystems (Needelman et al. 2018).

Measuring sequestration

Direct measurements of *NEP* are resource and knowledge-intensive. Carbon sink assessment programs require either numerous spatial and seasonal diurnal deployments of benthic chambers or measurements of the water columns’ metabolic gases (Maher and Eyre 2012; Gruber et al. 2017). Alternatively, continuous eddy covariance deployments on single benthic frames or atmospheric towers have been deployed (Lu et al. 2017; Rodil et al. 2019). The eddy covariance footprint can be more than 200 m long, but changes with wind direction require statistical imputation methods to effect a contiguous time series. However, while valuable, these approaches do not function on the time scale needed for estimating variance and trends over climatic scales. Furthermore, across intertidal systems, *GHG* fluxes immediately above the macrophyte assemblage fail to account for the lateral loss of CO_2 from the deeper parts of the sediment column (Fig. 1c) (Maher et al. 2018).

In place of direct measurements, annual sedimentary organic carbon accumulation has been touted as a measure of sequestration. It reduces the need for specialised equipment and variability can be estimated down the sediment column over decades while accounting for any lateral losses of CO_2. Photorespirometry chambers and eddy covariance flux experiments cannot detect lateral emissions during intertidal exchange (Maher et al. 2018). Furthermore, sediment cores have the advantage of appearing to integrate accumulation rates extensively beyond their diameters. One core can closely represent the mean for a wetland, and wetlands across a region (Callaway et al. 2012). Once the core has been extracted, traditional estimates of carbon accumulation are then calculated as the product of horizon carbon concentration and their sediment accumulation rate (velocity) below a diagenetically active surface layer (Cebrian 1999; Johannessen and Macdonald 2016). Accumulation rates are usually calculated using global geochronological models from the supply and decay of the natural radioisotope ^{210}Pb over the last 100 to 150 years (Lavelle et al. 1986). While historically known markers can be employed, a ^{210}Pb geochronology has the advantage of providing sedimentation rates over climatic scales without the knowledge of its depositional history and is widely accessible as a laboratory service.

Limitations and misunderstandings of sequestration down the sedimentary record

Sequestration calculations using the product of the sediment accumulation rate and organic concentration appear to solve the issues associated with direct measurements of sequestration as their *NEP*. Indeed, carbon accumulation termed as sequestration is pervasive across wetland and estuarine science (e.g. Carnell et al. 2018; Chmura 2013; Forbrich et al. 2018; Mitsch et al. 2013; Murray et al. 2011; Murray and Vegh 2012; Troxler 2013; Villa and Bernal 2018; Weston et al. 2014). Likely as a consequence, the term sequestration within Verified Carbon Standards has also been given to the accumulation rate of sedimentary carbon stocks (Needelman et al. 2018). However, no consideration is given to the extent and fate of export, and doing so implicitly assumes export has been consumed. The consequence is that accumulation rates will differ between similar ecosystems with similar inputs but subject to different rates of export. Eventually, the fate of this export across the landscape (e.g. Cole et al. 2007) and seascape will need to be addressed for a global balance. Indeed, for the extreme cases where no significant local accumulation occur within coastal seaweed ecosystems, the export of deposits sequestered to the deep ocean is considered as the sole sequestration metric (Krause-Jensen et al. 2018; although see Gallagher et al. 2021b). It must be said, that some articles use the term sequestration with accumulation without explicitly stating an equivalency with *NEP* (e.g. Carnell et al. 2018; Chmura 2013; Murray et al. 2011; Murray and Vegh 2012; Villa and Bernal 2018). Nevertheless, equivalency is often implied when comparisons are made with carbon accumulation or used to estimate lateral exchange of CO_2 from vertical *NEP* measurements (e.g. Forbrich et al. 2018; Mitsch et al. 2013; Troxler 2013; Weston et al. 201).

The equivalency of sequestration with sediment organic carbon accumulation, however, is mistaken from three standpoints. First, most teal and some blue carbon ecosystems are heterotrophic from the consumption of allochthonous subsidies, yet continue to increasingly accumulate organic carbon down the sediment column (Prairie et al. 2018; Duarte and Prairie 2005). Indeed, the only known example that we are aware (Tokoro et al. 2014) showed that recent annual sedimentary organic carbon accumulation of boreal seagrass meadows was on average around 35 times greater than their moderately autotrophic *NEP* rates (*recal.* 5.5 and 0.16 gC m^{-2} yr^{-1} respectively). Second, remineralisation of non-recalcitrant allochthonous and autochthonous organic carbon is not only confined to the first year or so of surface deposition. Evidence suggests that mineralisation continues over the decades and even a century in a significant and measurable manner (Chuan et al. 2020; Gälman et al. 2008; Maher et al. 2017; Zimmerman and Canuel 2002). The consequences are threefold. First, respiration from the sediment community continues to erode the impact of annual *NEP* as a carbon sink over climatic scales. Second, before accumulation can be calculated, organic carbon concentrations will need to be corrected for losses to mineralisation after 100 years of deposition (Chuan et al. 2020). Third, while not conceptual, there are also considerations of applying best practices to determine accurate sedimentation rates from a ^{210}Pb geochronology. These applications have several built-in assumptions for both their mapping methodologies and inductive models (Abril 2014). However, these are not always or sufficiently addressed within blue carbon research (Johannessen and Macdonald 2016), or through evaluation with one or a number of independent markers and presentation of the data set (Smith 2001). As a consequence, untested overestimates can arise from neglecting the effects of surface mixing/bioturbation of deeper older deposits (Johannessen and Macdonald 2016). Although for practical purposes, this is only when the ^{210}Pb inventory of the surface mixed layer is > 15% of the columns' total (Lu and Matsumoto 2005), a calculation that does is not always carried out. Not accounting for the loss of the sedimentary record due to erosion or massive deposition events (e.g, floods, storms, or tsunamis) will obfuscate natural radioactive decay depth variance and give erroneous results (Gallagher and Ross 2017). Nevertheless, scenario analysis suggested that the average sedimentation rate within a dynamic blue carbon ecosystem may be sufficiently robust from a shortened *ln*-linear portion of a complex ^{210}Pb decay profile, as determined by eye

(Arias-Ortiz et al. 2018). Whether that approach is justified in the field (Gallagher and Ross 2017), the averaged sediment accumulation rate cannot capture variability, a prerequisite for carbon accreditation (Needelman et al. 2018).

Is sequestration carbon accumulation?

The issues of *NEP* as sequestration relative to organic carbon accumulation and continued remineralisation are expanded and formalised with the use of three hypothetical scenarios **(**Fig. 1). Case I, II, and III develop from a simple closed immature system to a more complex mature open system, whereupon the use of uncorrected carbon accumulation becomes increasingly divergent from *NEP*. A means is then discussed on how to obtain the required additional information contained within the sedimentary record with a robust decomposition model.

Case I: An immature closed wetland.

This type of ecosystem approximates the beginning of an established closed wetland pond, or perhaps the start of restoration (Fig. 1a). This scenario was also set out by Prairie et al. (2018) but modified hereafter one further year of deposition. Primary production and consumption are at a steady state and confined within the wetland. The resultant *NEP* then becomes the difference between the annual CO_2 drawdown as it is fixed by the autotrophic assemblage (*GPP*) over the amount respired, as it is consumed by the autotrophic assemblage (*Pr*) and the heterotrophic community. The community is composed of herbivores (*Hr*), benthic detritivores (*Dr*), and decomposers largely confined to sedimentary micro-flora (*Br*). Once organic matter is deposited within the sediments, the microflora continues to consume and remineralise the remaining labile fraction leftover from detritivores over the following year (Cebrian 1999). Thus, the net rate of the remaining labile autochthonous organic carbon accumulated within the sediments after 1 year of deposition (*CAautl*) is equivalent to the annual *NEP* for that previous year (Eq. 2). *Under these conditions, carbon accumulation does indeed appear to be a proxy for NEP*.

$$NEP = GPP - Pr - Hr - Dr - \int_{1y}^{0y} Br = CAautl \tag{2}$$

Case II: An immature open wetland.

Most teal and blue carbon ecosystems are not isolated (Fig. 1b). Allochthonous organic carbon is added to wetlands either constantly or sporadically. For freshwater wetlands, this can come from rivers as components of soil and plant debris. On the other hand for coastal wetlands, adjacent seagrass, salt marsh, and mangroves can also supply material (Chuan et al. 2020; Gonneea et al. 2004). However, recalcitrant BC delivery can be from both soil washout and aerial deposition (Chew and Gallagher 2018). Importantly, unlike a wetlands' innate primary production, labile allochthonous carbon has been fixed outside as a separate ecosystem service. Consequently, the effect on the *NEP* is then constrained by the amount of labile allochthonous carbon remineralised (*Ir*) within the ecosystem and not included in the carbon balance (Eq. 3). In other words, allochthonous production is not an ecosystem service of the receiving ecosystem only its remineralisation. This error appears to have been propagated from the terrestrial assessments, where allochthonous inputs rates were included in the *NEP* term to describe carbon accumulation (Lovett et al. 2006). Assuming most of the allochthonous inputs are rapidly deposited by the canopy into the surface sediments (Bos et al. 2007), their remineralisation is most likely confined within those sediments. The *NEP* (Eq. 3) then becomes the subtraction of the allochthonous remineralisation over a year of deposition (*Ir*) from the remains of autochthonous production ($CAautl_{1y}$) after macrophyte respiration (*Pr*), its consumption by the fauna (*Hr* and *Dr)*, and decomposition within the sediments (*Br*) (Eq. 3). In terms of what can be conceivably measured, *Ir* is equivalent to the difference between the input of allochthonous carbon (*I*) and what remains after 1 year of deposition (Il_{1y}). This illustrates a clear contrast to *CA* which is the sum of the remains of autochthonous production $CAautl_{1y}$ and the remains of allochthonous carbon Il_{1y} (Eq. 4).

$$NEP = GPP - Pr - Hr - Dr - \int_{1y}^{0y} Br - \int_{1y}^{0y} Ir = \ CAautl_{1y} - (I - Il_{1y}) \tag{3}$$

$$CA_{1y} = \ CAautl_{1y} + Il_{1y} \tag{4}$$

By subtracting Eq. 4, the elements of carbon accumulation from the sedimentary and input terms in Eq. 3, show that the total organic carbon accumulation will overestimate *NEP* by the rate by which allochthonous organic carbon is initially supplied to the sediments before it had been consumed (II_{0y}, see Eq. 5). To state it in another way, the fraction of allochthonous carbon that has been decomposed is built into the amount remaining for accumulation. For example, should the allochthonous fraction be composed only of recalcitrant BC, then $Il = BC$. Consequently, *NEP* can be calculated by subtracting the sedimentary *BC* fraction from the total organic carbon accumulation rate, because *BC* is not consumed.

$$NEP = \ CA_{1y} - II_{0y} - BC \tag{5}$$

Case III: A mature intertidal wetland.

Most canopy ecosystems are both open and have been sufficiently established to deposit an extensive sediment column (Fig. 1c). The depth of age of the column produces a legacy of sedimentary organic decay over the past century, albeit faster over decades, thus, further constraining Eq. 3 (Eq. 6). Indeed, for mangroves, it has been found that the mineralisation from century-old carbon deposits is tidally advected into adjacent waters at a rate equivalent to the CO_2 vertical flux (Maher et al. 2018). The result is a similar construct and description of input and output terms as in Eq. 3 for annual rates of production and deposition. The difference is in the projection of what remains of organic carbon accumulation after a century of deposition, as seen from the subscripts and integrals (0y and 100y).

$$NEP = GPP - NPP - Hr - Dr - \int_{100y}^{0y} Br - \int_{100y}^{0y} Ir = CAautl_{100y} - \left(I_{0y} - Il_{100y}\right) \qquad (6)$$

By substituting (5), after correcting for remineralisation over climatic scales (100 years) into Eq. 6, and subtracting the allochthonous recalcitrants inputs such as BC (CA_{BC}), the *NEP* becomes the difference between accumulated organic carbon after it has been deposited for 100 years (CA_{100y}) from the sum of the initial deposition rate for allochthonous carbon before consumption (I_{0y}) and the accumulation of allochthonous recalcitrants likely dominated by BC (Eq. 7).

$$NEP = CA_{100y} - (II_{0y} + CA_{BC}) \qquad (7)$$

Surface horizons will thus require substantial corrections in the amount of carbon lost after 100 years of deposition, decreasing as the horizons become older. The true rate of mitigation through accumulation then becomes the product of the accretion rate and remaining concentrations after 100 years since it was deposited. Furthermore, the variability and trends for *NEP* can be extracted throughout the sediment column irrespective of the degree of mineralisation or lateral exchange.

Is there a mitigation role for total organic carbon accumulation?

The concept of locking away the remains of organic carbon produced and supplied to the ecosystem within the sediments still ‘feels’ like a sequestration concept. That is true, but only in the sense of what would have not been locked away if the canopy ecosystem had not been there. Some insights into the differences imposed by the presence of a canopy may be gauged from the population organic *CA* medians between vegetated and non-vegetated ecosystems–62% of coastal wetland accumulation rates (Wilkinson et al. 2018). However, this comparison may need to be tempered as non-vegetated ecosystems will support a different mix of allochthonous and autochthonous carbon sources (Macreadie et al. 2014). Nevertheless, in some situations, annual *CA* may still provide additional information on ecosystem form or function. A larger *CA* rate can separate a near equivalent *Ir* term between an ecosystem assemblage supplied by large amounts of relatively recalcitrant allochthonous carbon from another supplied by smaller amounts of more labile allochthonous carbon. Likewise, is also conceivable that they can also support similar standing stocks, should the larger amount of inorganic mineral material be associated with the supply of the more recalcitrant allochthonous sources. Whether annual *CA* can be as a conditional vector stock service, perhaps another vector as a useful descriptor of the position of the wetland and various replacement scenarios in mitigation 3D phase space with *NEP* and vulnerable stock service is an avenue of further discussion and testing.

Putting it all together

Preservation

It is proposed that the full mitigation potential in preserving a wetland carbon sink has three elements (Eq. 8). First, a sequestration service from differences in carbon balance (*NEP*) between the wetland and its anthropogenically driven replacement (*NEPR*). Second, a vulnerable stock service as the fraction (θ) of labile organic carbon stock vulnerable to remineralisation. The extent of the vulnerable stock ($Cstock_{1\,m}^{y=1}$) projected to a depth of 1 m as the default value (Pendelton et al. 2012) and after subtraction of intrinsically allochthonous and autochthonous recalcitrant carbon, primarily BC (TBC). This fraction is then given dimensional integrity with sequestration as the time (φ) which the vulnerable fraction was remineralised after disturbance. In other words, the term is equivalent to the difference in what remains over the time it takes to establish its anthropogenic replacement towards a steady-state equilibrium. Third, the possible inclusion of the rate of organic carbon accumulation that would be vulnerable without the protection of a canopy ($\theta CA_{y=1} - CA_{BC}$), and after subtraction of allochthonous recalcitrants, again, primarily BC.

$$Annual\ C\ offset\ credit = (NEP - NEPR) + (\frac{1}{\varphi}(\theta Cstock_{1\,m}^{y=1}) - TBC)) + (\theta CA_{y=1} - CA_{BC}) \qquad (8)$$

Restoration

The value of a carbon mitigation service for restoring, however, must be less than preserving a wetland. That is to say, restoration is a measure of only the difference between the *NEPR* of the previous anthropogenic state and the *NEP* of a restored version of its natural state. Thus, the Annual C offset credit = *NEP-NEPR* without the remaining terms of Eq. 8. Any perceived gain of *CA* in the restored system is only an expression of its *NEP* as the rate of accumulation of autochthonous material over the consumption of its labile allochthonous fractions (Eq. 5). It is true, that restoration will lead to a larger organic carbon stock. However, if these stocks are to be considered as a mitigation service, they need to be vulnerable to remineralisation from business as usual anthropogenic pressures. For a successful restoration, however, anthropogenic pressure has been addressed, resulting in an ecosystem that is ecologically and economically robust over the long term (Thorhaug et al. 2020).

The above process of diagenetic projection, hindcasting for sequestration, and stocks would seem at a first inspection to be less than parsimonious. However, the apparent complexity only arises from the number of arithmetic operations. Notwithstanding accurate geochronologies, in the final analysis, accuracy only requires a robust sedimentary decomposition model for the different sources of organic carbon and a means to estimate the fractions of the autochthonous and allochthonous labile and recalcitrant sources. The model being used to both project remineralisation and hindcast allochthonous remains to the original concentration and inputs rates (I_{0y}) (Fig. 2a). For many programs, accurate geochronology and determination of organic sources are part of a best practice (Bindoff et al. 2019). To be useful, however, the decomposition model must account for or dismiss confounding differences in sediment types, redox conditions, temperatures, and organic sources across all time scales. This describes the relative precise and robust power model of Middelberg (1989) (r = 0.987) as determined down a range water saturated sediments and its application to a mixed organic estuarine sediment column ((Zimmerman and Canuel 2002). The model describes how the organic mix becomes increasingly recalcitrant over time. The only requirement is a starting point for a continuously recalcitrant first-order decay constant characteristic of the organic mixture, or its organic components (Gallagher 2015). How this model responds to wetlands that frequently flood and dry is uncertain. A major review by Brinson et al. (1981) could draw no conclusions that increased frequency or duration of flooding leads to increased decomposition rates. An alternative relies on a collection of cores (> 20) sufficient to normalise models for variability in space as a proxy for time (Strayer et al. 1986; Johannessen et al. 2021), or identifying asymptote concentrations with depth. However, the former is resource-intensive and requires a gradient of sedimentation and source supply rates. For the latter, simple decay to an asymptote with depth requires a constant rate of deposition and unchanging proportions of all organic sources. This is not usual for canopy ecosystems. Concentration profiles are often characterised by a series of complex broad peaks and troughs, and in many cases, surface concentrations are less than their centennial counterparts (Callaway et al. 2012; Ellison and Beasy 2018; Gonneea et al. 2004; Rozaimi et al. 2017; Serrano et al. 2014).

Testing the accreditation model

Two examples were chosen from the available literature to calculate the annual *NEP* variability over decadal to centennial time scales as determined from (Eq. 7). A mangrove sediment containing moderately labile seagrass litter (Gonneea et al. 2004), using BC estimates from similar systems. A seagrass sediment dominated by relatively recalcitrant mangrove detritus and moderate fractions of BC (Chuan et al. 2020). The sites were also selected to bookend expected rates of supply and retention of organic carbon to their sediments (Li et al. 2018; Gallagher et al. 2020). The mangrove was situated near the entrance of its lagoon’s marine tidal delta region, a region of relatively high rates of exchange with coastal waters. The seagrass was situated near the head of its lagoon within a relatively enclosed embayment. Full details of the site descriptions can be found in Supplementary Information along with details and support behind the theory of the diagenetic models, their evaluation, and *NEP* estimates can be found in Supplementary Information (S1). Decompositional decay curve model used for the hindcasting and projection of variables (Eq. 7) and in stability tests can be found as Excel™ files in Supplementary Information (S2), located at https://doi.org/10.25959/NNHB-T463. Data for first-order component decay parameters for the decay curve model, along with the template and results for hindcasting, projection of individual organic sources as weighted their total organic carbon, along with the evaluation of the model used can be found in Supplementary Information (S3) located at https://doi.org/10.25959/NNHB-T463.

Results of case studies

Net ecosystem production and carbon accumulation

The sediment accumulation rates were an order of magnitude higher in the mangrove forest than in the seagrass meadow (Fig. 3a, b). There was a considerable disparity between total organic carbon accumulation rates and the calculated *NEP* for both systems. For the mangrove forest, the median *CA* rate was over 33.5 times greater than *NEP*. This was not much less than *CA* rates after correction for decomposition losses (27.5 x) (Fig. 3a). This relative difference, however, was reduced for their respective median *NEP* rates. The seagrass meadow *NEP* (7.2 gC m^{-2} yr^{-1}) was around 4 times less than the mangrove forest *NEP* (27.6 gC m^{-2} yr^{-1}) (Fig. 3a, b). Indeed, from the limited data available, it appears that the seagrass had been essentially carbon-neutral over the last 15 years (geometric mean = 0 gC m^{-2} yr^{-1}). For the seagrass meadow, this likely reflects a respiration assault from the consumption of high rates of supply of relatively labile allochthonous mangrove litter (75.4% to 80.4% of organic carbon, Supplementary Information (S3) at https://doi.org/10.25959/NNHB-T463). However, the differences for the mangrove reflect the smaller allochthonous supply of a more labile phytoplankton assemblage.

Interestingly, the overall variability seen in *CA* rates was notably less than the relative variability in their respective *NEP* (Fig. 3a, b). Unlike the mangrove forest, the seagrass meadows' *NEP* switched between heterotrophy (-72.5 gC m^{-2} yr^{-1}) and autotrophy (7.2 and 8.5 gC m^{-2} yr^{-1}). These values are within the range reported from direct measurements also across a lagoon system (-92 to 227 gC m^{-2} yr^{-1}) for the boreal region, and their shallow subtropical counterparts immediately outside a lagoon (-539 to 215 gC m^{-2} yr^{-1}) (Tokoro et al. 2014). For the mangrove forest, the *NEP* maintained a degree of autotrophy through the century of deposition, ranging from 16.3 to 55.5 gC m^{-2} yr^{-1} (median 26.9 gC m^{-2} yr^{-1}). This median and range are notably smaller than reported for global medians of between 628 gC m^{-2} yr^{-1} (Alongi 2020) and 221 gC m^{-2} yr^{-1} (Duarte et al. 2005). However, the estimates were largely based on measurements of gas emissions emanating from the canopy system and could account for the significant lateral transport of remineralisation products during tidal exchange from the deeper parts of the sediment column (Fig. 1d) (Maher et al. 2018). Nevertheless, the examples suggest that *CA* augmented with allochthonous inputs would likely lead to major overestimates of sequestration, irrespective of decompositional corrections (Fig. 3). However, this hierarchy could conceivably be reversed during the early to mid-stages of a restoration. Stocks would not have had sufficient time to accumulate and the canopy and root system may not have been sufficiently developed to fully promote carbon accumulation.

Carbon accredited offsets, stocks, and sequestration

The current methodology for wetland carbon mitigation offsets is determined from the difference in vulnerable carbon stocks between its baseline or replacement ecosystem as the time it takes for the baseline to establish itself towards a steady-state equilibrium (Needelman et al. 2018). For seagrass and mangroves, this was calculated as typically 732 and 636 gC m^{-2} yr^{-1} respectively. This assumed that the sedimentary standing stock today is a good representative of the last century median (see section “Standing stocks over time” Supplementary Information (S1) for details) and losses to remineralisation were on average around 75% (Pendleton et al. 2012) as the anthropogenic replacement established itself over the next 20 years. In addition, corrections for recalcitrants were made as an allochthonous BC fraction, measured at 11% for the seagrass and estimated at 5% for the mangrove (Supplementary Information (S1)). Although notably for salt marsh and coastal seagrass meadows, BC fractions can reach as high as a third and close to half of their carbon stocks respectively (Gallagher et al. 2021a). However, we did not include the suggested additional prospective stock vector, which describes the difference in annual carbon accumulation without a canopy service to otherwise increased net deposition rates (in Eq. 8: ($\theta CA_{y=1}$ – .CA_{BC})). Whether its addition is justified will depend on its ability to separate form and function within similar wetland categories; a classification system yet as not explored or its need tested.

Along with stocks, it has been argued that sequestration (as its annual *NEP*), is required to assess the wetlands’ full mitigation potential (Eq. 8) relative to its anthropogenic replacement. For comparison with the mangrove system, Jobo Bay was chosen (Caffrey 2004) as typical for the seagrass replacement The bay occupies a similar niche of tropical enclosed embayment supporting a non-vegetated bottom surrounded by fringing mangroves. For the mangrove, the annual average *NEP* of mangrove shrimp aquaculture ponds was chosen as its anthropogenic replacement (Chen et al. 2016).

$$\text{Seagrass preservation} \qquad +1921.2\ \text{gC m}^{-2}\ \text{yr}^{-1} = (+7.2NEP_{seagrass} - (-450NEP_{\text{baseline}})) + 732C_{stock} \tag{9}$$

$$\text{Mangrove preservation} \qquad +690.6\ \text{gC m}^{-2}\ \text{yr}^{-1} = (+27.6\ NEP_{seagrass} - (+5NEP_{\text{baseline}})) + 668C_{stock} \tag{10}$$

Interestingly, while the sum of stock and individual *NEP* services stocks converge, what ultimately determines the differences appears to be the *NEP* of their anthropogenic replacements (Eqs. 9, 10). Although, one may have to consider that the anthropogenic replacement may support a larger *NEP* rate and reduce the overall accreditation service. This would be the case for mangroves replacing salt marsh (Alongi 2020) and conceivably for the above seagrass example, should the supply of mangrove detritus be lost from shoreline developments. The calculation also highlights, indirectly, the importance of neglecting the potential fate of a considerable stock housed within mangrove trees and root biomass. No information is available for this mangroves' biomass. Nevertheless, a mangrove biomass for a similar niche and species assemblage close to the entrance of its lagoon (Gallagher et al. 2020), suggests an additional mitigation service of around 891 gC m^{-2} yr^{-1}, should the mangrove be consumed or burnt over 20 years. This is in contrast to the largely insignificant contribution of seagrass biomass relative to its meadow's total organic carbon stock (Gallagher et al. 2020). Together, there appears to be a surprising convergence between the mitigation potential of these two disparate tropical ecosystems (i.e. 1581.6 gC m^{-2} yr^{-1} and 1923.4 gC m^{-2} yr^{-1} for the mangrove and seagrass respectively). Although, it must be said that the convergence was enhanced by the relative positions within their lagoons. The mangrove forest is located close to the entrance where carbon biomass is just over half of the upper lagoon counterparts (Gallagher et al. 2020). In contrast, the seagrass meadow is located in the upper and muddier parts of its lagoon, where sedimentary carbon stock is more than 10 greater than their lower lagoon counterparts (Gallagher et al. 2020). Either way, the significant fraction as stocks (Eqs 9 and 10) for both systems suggests that restoration is likely to be around a third less valuable than preservation as a mitigation service.

Conclusions

A full carbon accrediting assessment for the preservation of wetlands requires estimates of both sequestration and stocks that are vulnerable to remineralisation relative to their anthropogenic replacements. However, only relative sequestration need only be considered for wetland restoration. Continued remineralisation of the annual deposition of sedimentary organic stocks suggests that any estimate of sequestration should be calculated over climatic scales. This is after significant fractions of allochthonous recalcitrants such as black carbon are subtracted from that service. However, total *CA* is not a measure of sequestration, as previously used to account for centennial variability, for it will significantly overestimate this service, be it uncorrected or corrected for continued remineralisation. Nevertheless, with the use of a robust decomposition model and a means to untangle organic source contributions, the sedimentary record contains sufficient information to determine reasonable estimates of sequestration as the *NEP*. The review shows that conceptually unbiased and complete measurements of mitigation services will assure i) confidence across both voluntary and compliance carbon markets in that *GHG* offsets do not permit emissions above the capacity of a wetland; ii) underestimate the ability for regions to reach zero net carbon emission targets.

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Figure and equation captions

Fig. 1 Conceptual models for ‘a’ Case I; an immature subaquatic wetland pond after one year of deposition and closed to inputs and outputs but impacted by atmospheric deposition of black carbon; ‘b’ Case II; an immature aquatic wetland after one year of deposition, open to river and atmospheric inputs, and outputs; ‘c’ Case III; a mature intertidal mangrove with more than 100 years of deposits and open to atmospheric inputs, litter deposits from an adjacent seagrass ecosystem, with net losses of CO_2 from the canopy from shallow and deep parts of the sediment column vertically, and laterally during tidal exchange. The figure was produced in Abode Illustrator CS6™ with components taken from Media Library Integration and Application Network (https://ian.umces.edu/media-library/).

Fig. 2 Hypothetical sedimentary decomposition curves required to estimate *NEP* and carbon accumulation at climatic time scales. Curves within ‘a’ illustrate the process used to estimate the *NEP* (Eq. 5) from carbon accumulation rates, after correcting the component's sedimentary content over times of deposition. The allochthonous content I_y sits on its remineralisation curve as an example of a relatively recalcitrant form. It represents what remains after ‘*y*’ years of deposition from its original content I_{0y} deposited and mixed into the surface sediment layer. The total organic carbon content mixture (C_y) sits on a relatively labile remineralisation curve as the sum of individual organic components decomposition curves, weighted for their respective fractions. It represents what remains after ‘y’ years of deposition and projected to what would remain over climatic time scales (i.e. 100 years). Curve ‘b’ represents the used to hindcast an example of the content *C*y to the time of the original annual deposition C_y=1 (see section “Is there a mitigation role for total organic carbon accumulation?”). The figure was drawn within Microsoft PowerPoint™ 2013.

Fig. 3 Carbon sink concepts estimated down the sedimentary record for ‘a’ the seagrass meadow at Salut lagoon; and ‘b’ the mangrove forest at Chelem lagoon. The points (●) represent *CA* concepts at different depositional ages down their respective sediment columns (see Supplementary Information (S3) at https://doi.org/10.25959/NNHB-T463). The net ecosystem production after 100 years of deposition (*NEP*(100yrs)); Sediment carbon accumulation after 100 years of deposition (CA (100yr)) and without decomposition corrections (CA); 62% of carbon stock accumulation hindcasted to one year of accumulation (CA(1yr)) if hypothetically it was deposited within a non-vegetated patch (see section “Is there a mitigation role for total organic carbon accumulation?”). The figure was created in the statistical software package PAST™, exported as a svg file then modified and converted to an eps file within Abode Illustrator CS6™.

Declarations

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Conflict of interest/competing interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Ethics approval

Not applicable, no animal or plant materials were disturbed or manipulated during sampling or analysis

Guidelines on ethical review or waiver

Australia: https://www.arc.gov.au/policies-strategies/policy/codes-and-guidelines (accessed November 2021).

Consent to participate

Not applicable.

Consent for publication

Not applicable.

Availability of data and material.

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CRediT authorship contribution statement

JBG: Conceptualization, Validation, Formal analysis, Investigation, Writing - original draft, Writing - review and editing, Visualization, KZ: Conceptualization Writing - review and editing, CHC: Validation, Visualization, Writing - review and editing. All authors read and approved the final manuscript.

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