

All tidal wetlands are blue carbon ecosystems

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Abstract

Managing coastal wetlands is one of the most promising activities to reduce atmospheric greenhouse gases, and it also contributes to meeting the United Nations Sustainable Development Goals. One of the options is through blue carbon projects, in which mangroves, saltmarshes, and seagrass are managed to increase carbon sequestration and reduce greenhouse gas emissions. However, other tidal wetlands align with the characteristics of blue carbon. These wetlands are called tidal freshwater wetlands in the United States, supratidal wetlands in Australia, transitional forests in Southeast Asia, and estuarine forests in South Africa. They have similar or larger potential for atmospheric carbon sequestration and emission reductions than the currently considered blue carbon ecosystems and have been highly exploited. In the present article, we suggest that all wetlands directly or indirectly influenced by tides should be considered blue carbon. Their protection and restoration through carbon offsets could reduce emissions while providing multiple cobenefits, including biodiversity.

Keywords: carbon offsets, Cypruss, Melaleuca, peatlands, tidal freshwater wetlands

Reducing atmospheric greenhouse gas (GHG) concentrations and adapting to climate change have become one of humanity's biggest challenges. This global effort requires, primarily, reducing fossil fuel emissions and, secondarily, implementing nature-based solutions that can help offset peaks in projected global warming (Matthews et al. 2022). The Clean Development Mechanism, part of the United Nations Framework Convention on Climate Change, allows countries to purchase carbon credits for GHG emissions reduction projects to reach their targets under the Paris Agreement. Of the projects that can reduce emissions, restoring or managing coastal wetlands is one of the more promising activities, providing multiple cobenefits and significantly contributing to meeting the United Nations Sustainable Development Goals (Smith et al. 2019). Management of blue carbon is an integral part of carbon offset strategies but is usually limited to specific wetland types, including mangroves, marshes, and seagrass (McLeod et al. 2011).

Blue carbon ecosystems can be defined following the criteria by Lovelock and Duarte (2019) as marine or coastal wetlands that have long-term storage of fixed carbon dioxide; that can remove GHGs; that have been lost or degraded because of anthropogenic impacts; that can be managed to enhance carbon stocks, reduce

GHG emissions, and facilitate habitat persistence; that have the potential for interventions that have no or minimal social and environmental harm; and that can be managed in alignment with policies for mitigation and adaptation to climate change. Mangroves, saltmarshes, and seagrass meet all the conditions except that, in some cases, management interventions could have social and environmentally negative consequences—for instance, when competing with aquaculture that provides income to local communities or when their restoration results in the loss of freshwater wetlands. Other ecosystems, such as coral reefs, do not satisfy this blue carbon definition, because their carbon balance results in the net production of carbon dioxide through calcification.

Blue carbon projects are exponentially expanding worldwide; however, the demand for carbon credits greatly exceeds the offer. Currently, most blue carbon projects are conducted in mangrove forests. However, many other wetlands have characteristics that strongly align with the blue carbon definition and support expanding global initiatives for large and management actionable areas. The previous data gaps on carbon dynamics in these ecosystems are closing rapidly. In this review, we explore whether tidal wetlands other than mangroves, saltmarshes, and

Received: April 28, 2023. Revised: January 14, 2024. Accepted: February 6, 2024

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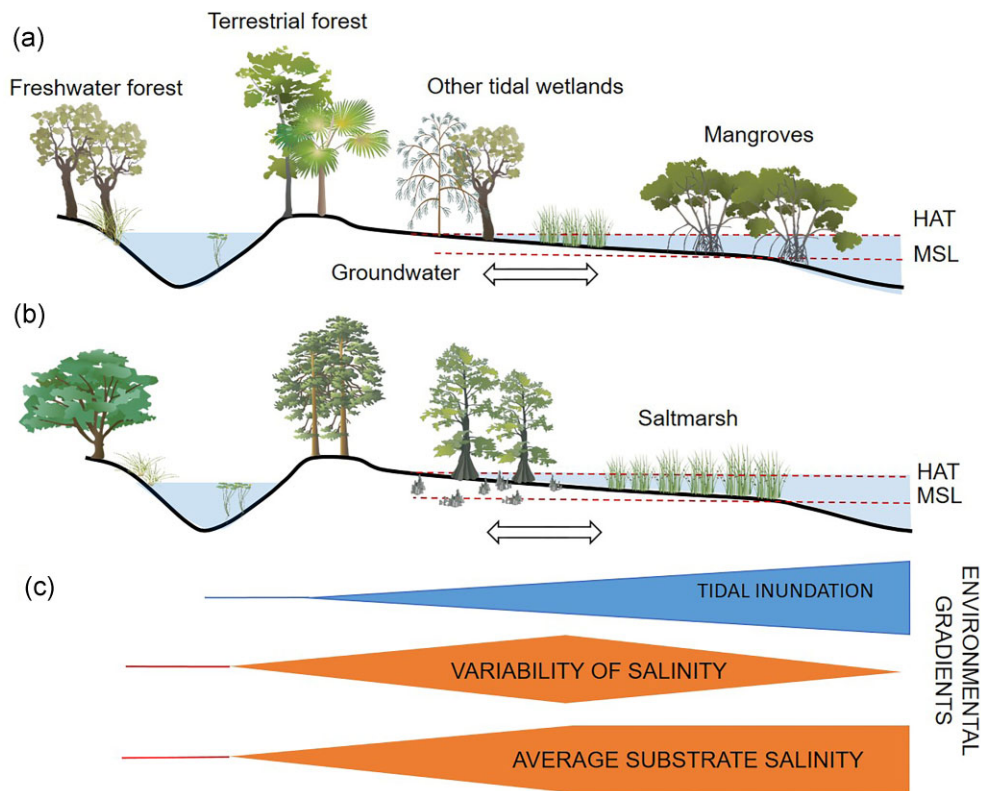


Figure 1. Typical distribution of vegetated communities in (a) tropical or subtropical *Melaleuca* and *Casuarina* forest in eastern Australia and (b) temperate cypress forest and tidal freshwater marshes in the Southeast region of the United States, (c) within a tidal inundation and salinity gradient. Abbreviations: HAT, highest astronomical tide; MSL, mean sea level. Graphics: Kim Kraeer, Lucy Van Essen-Fishman, Tracey Saxby, Annie Carew and Catherine Collier from the Integration and Application Network (ian.umces.edu/media-library).

seagrasses align with the current definition of blue carbon. To achieve this goal, we first revisit the biochemical and physical characteristics of wetlands that can be defined as blue carbon. Second, we provide a compilation of published and unpublished data on the capacity of these previously unaccounted tidal wetlands to sequester carbon while having relatively low GHG emissions. Third, we investigate the threats and management options for these tidal wetlands. Finally, we provide knowledge gaps and describe future opportunities. We propose that protecting and restoring all tidal wetlands could reduce GHG emissions while providing multiple cobenefits, including biodiversity.

Definition and classification

In the present article, we define blue carbon ecosystems on the basis of the biochemical and physical attributes that support the processes that result in climate change mitigation. These processes include long-term organic carbon storage, mostly in their soils, and low GHG emissions; the former results from high primary productivity and low soil decomposition rates in waterlogged soils. The latter is mainly caused by sulphates in marine water, which outcompete carbon as an electron acceptor, inhibiting methanogenesis. On the basis of these criteria, we suggest the definition of blue carbon include the following language: “ecosystems that are influenced by marine waters that fix carbon dioxide and that store and accumulate it as organic carbon. They are bounded by the highest levels of tidal inundation at the terrestrial edge and by the limits of the photic zone in the marine edge.”

This definition includes forested wetlands and those dominated by shrubs, grass, sedges, or microalgal mats. It excludes

temporal carbon storage, such as macroalgal beds that do not sequester carbon in their sediments and ancient peat formations, which are not currently fixing and accumulating carbon. The deep ocean is also excluded from this definition, because carbon is not fixed within these ecosystems (except for localized chemotrophic communities) but is transported from elsewhere.

Tidal wetlands are typically distributed along inundation and salinity gradients. Seagrass is found within the lower end of tidal inundation and can be permanently flooded. Mangroves and saltmarshes are usually found slightly below or above mean sea level. Above mean sea level and extending to the highest tide levels, other ecosystems can occur where geomorphology, climatic conditions, and water availability allow (figure 1). These wetlands are influenced by tides; their soils are directly flooded by the highest tides, or changes in groundwater levels during tidal fluctuations or affected indirectly through ocean waves or marine spray.

Tidal wetlands at the highest end of the intertidal zone are dominated by grasses, sedges, or woody plants forming shrubs, thickets, or tall forests, usually of a single or a few tree species. They have primarily organic unconsolidated soils, with different compositions and grain sizes. Their inundation can be frequent or infrequent, regular or sporadic, depending on their location within the landscape and their association with river channels. Their inundation regime and, consequently, their soil salinity depend on the tidal amplitude, the occurrence of storm surges, groundwater flows, and wind and wave intensity. Many of these tidal wetlands are oligohaline (0.5–5 practical salinity units).

We provide a classification of types that can be applied to tidal wetlands globally following the key attributes of blue carbon ecosystems (figure 2). These attributes are divided into four

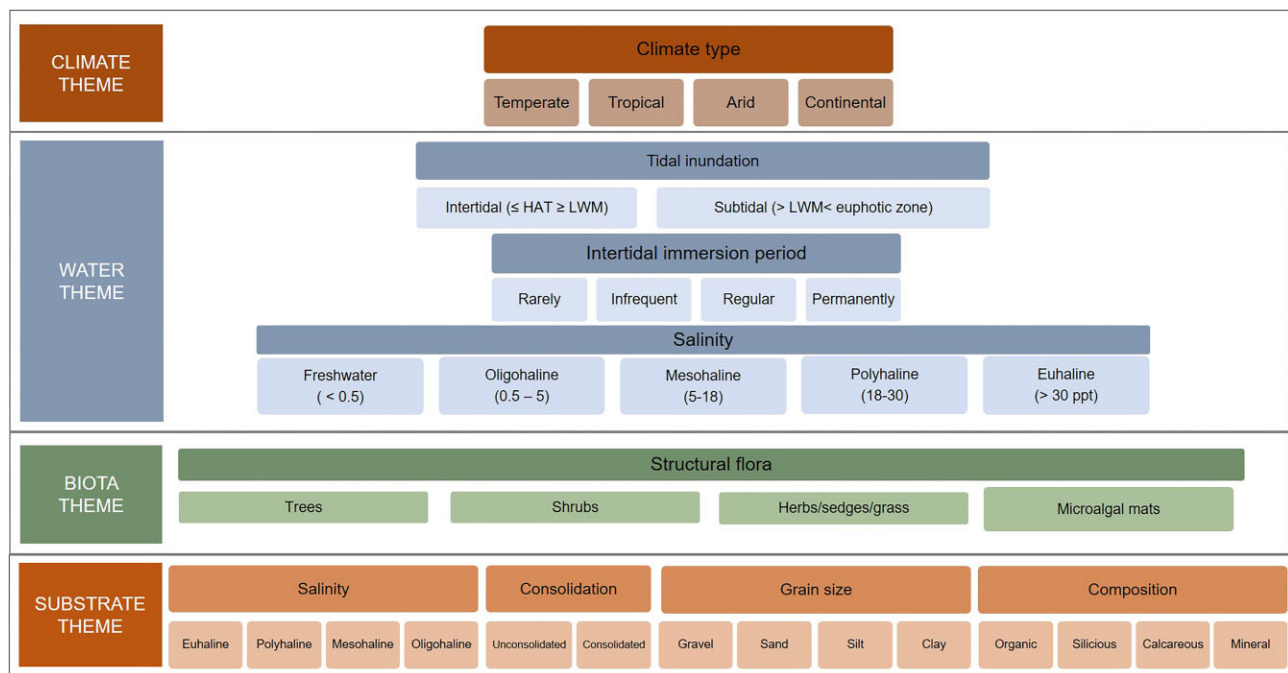


Figure 2. Attribute classification for blue carbon ecosystems, including functional typologies (climate), attributes (dark) and categories (light) within themes (climate, water, biota, and substrate).

themes: climate, water, biota, and substrate. Within each theme, there are attributes subdivided into categories. For instance, for the water theme, one attribute is tidal inundation, with intertidal and subtidal being categories. Other attributes within the water theme are the intertidal immersion period or the frequency of tidal inundation and salinity. For biota, the attributes essential for blue carbon are structural flora, which includes trees, shrubs, sedges and microalgal mats (e.g., cyanobacteria mats). The substrate attributes include the dominant grain size, soil salinity, and the composition of the sediment, whether organic, silicious, calcareous, or mineral. For example, coastal *Melaleuca* forests in northern Australia are tropical oligohaline wetlands, which are infrequently inundated, are dominated by trees, and have a substrate that is saline, organic, or mineral and composed of mostly silt and clay.

Geographical location

Tidal wetlands, other than mangroves and saltmarshes, have been reported worldwide. Some of these are called *supratidal forests* in Australia (Iram et al. 2022); *tidal freshwater wetlands*, *tidal swamps*, *tidal forested wetlands*, or *brackish tidal wetlands* in the United States (Conner et al. 2007, Duberstein et al. 2014); *transitional forests* in South East Asia (Aslan et al. 2016); and *coastal wetlands* or *swamp forests within estuary boundaries* in South Africa (figure 3, table 1; Van Deventer et al. 2021, Riddin and Adams 2022).

In Southeast Asia and throughout the Pacific, common tidal wetlands at the terrestrial edge are forests of *Melaleuca* or paper bark tree swamps (CABI 2019). For instance, extensive forests of *Melaleuca viridiflora* or *Melaleuca quinquenervia* cover the coast of Australia. Similarly, *Casuarina* forests are found on the southern Australian coast. *Melaleuca* and *Casuarina* trees tolerate acidic and oligohaline conditions. They typically form dense monospecific stands of fast-growing trees, and both have become invasive species in many regions outside their native ranges. For instance, in the Everglades of Florida, *M. quinquenervia* currently occupies an

area larger than that of native mangroves (Turner et al. 1998). *Casuarina* is also widely distributed, because it has been introduced into many countries for wood production. It has become invasive in many coastal wetlands, such as in South Africa, Brazil, India, and the Caribbean (CABI 2019).

In the southeastern United States, tidal freshwater forested wetlands are influenced by river flows and flooded by spring tides during high river but not flooded during low river stages or neap tides. These tidal wetlands include forests of bald cypress (*Taxodium distichum*), shrubs of twinberry (*Lonicera involucrate*), and mixed bottomland hardwood forests (*Nyssa*, *Fraxinus*, *Alnus*; Conner et al. 2007, Duberstein et al. 2014). In the northwestern United States, tree-, grass-, and sedge-dominated tidal wetlands may be brackish, with salinities ranging from fresh to mesohaline (5–18 practical salinity units; Brophy et al. 2011). These wetlands are located from just above the mean higher high water up to the highest tide level (Brophy et al. 2011).

Forests in southern Africa can be found within the estuarine functional zone, which has been classified as the habitat below the 5-meter topographical contour (Veldkornet et al. 2015, Adams et al. 2016). They are associated with freshwater lakes and coastal drainage areas extending from the subtropical to tropical areas of South Africa to Mozambique. These tidal wetlands can be found at the fresher upper reaches of estuaries but typically occur in temporarily closed estuaries where the connection to the sea can be interrupted by sandbars forming across the mouth during low river flow under high coastal wave conditions (Van Niekerk et al. 2020). These small estuaries can be perched above normal tidal levels, resulting in brackish conditions because of little tidal exchange. These tidal wetlands are dominated by *Hibiscus tiliaceus* (lagoon hibiscus) and *Barringtonia racemosa* (powder puff tree) or freshwater swamp tree; figure 3). The endemic *Raphia australis* (raffia, giant palm) can also form tidal wetlands in Maputaland, South Africa, at Kosi Bay and the Siyaya estuaries.

There are also various reports of tidal wetland forests in Indonesia that are not mangroves. In Papua, transitional forests, or

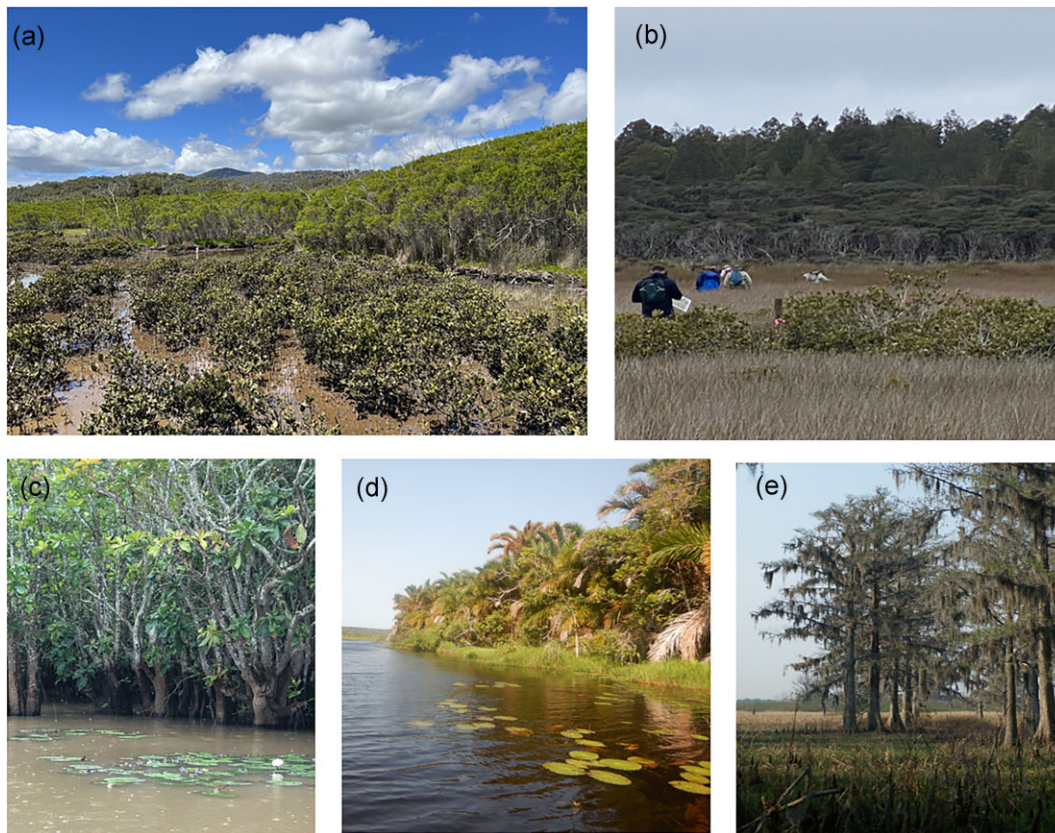


Figure 3. (a) *Melaleuca* forests fringing mangroves in Australia, (b) kānuka and mānuka forests fringing saltmarsh or mangroves in New Zealand, (c) *Barringtonia racemosa* and (d) *Raphia australis* forests in South Africa, (e) cypress swamp in the United States. Pictures: JK (a), KWK (b, e), JBA (c, d).

those fringing mangroves in the landward zone, cover large areas (Aslan et al. 2016). In Central Kalimantan, riverine mangrove forests are bordered by wetlands in the upper tidal reaches of the river dominated by *Ganua motleyana* (Sapotaceae) and *Gluta walichii* (Anacardiaceae), among 49 other tree species (Murdiyarto et al. 2009).

In New Zealand, forests located inland of tidal marshes were initially dominated by kānuka (*Kunzea ericoides*) and mānuka (*Lepidospermum scoparium*) forests. Other unique tidal wetlands occur in tropical America, such as zapotonales (*Pachira aquatica*; Adame et al. 2015) in the Mexican Pacific, the *Mora* forests (*Mora olifera*) of central and South America (Palacios Peñaranda et al. 2019), and the mixed mangrove and rainforest of the Amazon River in Brazil (table 1; Bernardino et al. 2022). Many other tidal wetlands probably have not been considered blue carbon worldwide, which could include significant areas.

All tidal wetlands are important for carbon storage but also for supporting biodiversity (figure 4). For instance, in *Melaleuca* forests in Australia, at least 642 species of mammals, birds, fungi, amphibians, reptiles, and plants have been recorded (WetlandInfo 2024). These include the endangered spectacle flying fox (*Pteropus conspicillatus*), the eastern curlew (*Numenius madagascariensis*), and the critically endangered orange-bellied parrot (*Neophema chryso-gaster*). In the Mekong Delta, Vietnam, 159 bird species have been recorded in *Melaleuca* forests, 15 of them listed under the Global and Regional Endangered List (Tran and Matusch 2017). In peat swamps in Southeast Asia, 2236 species of plants, mammals, reptiles, amphibians, and fish have been recorded, with 252 species restricted to these habitats (Posa et al. 2011). In some coastal peat swamps, such as those in Borneo, orangutans (*Pongo pygmaeus*)

and proboscis monkeys (*Nasalis larvatus*) are commonly found (Posa et al. 2011). Forested tidal wetlands of the northwestern United States provide foraging habitats for juvenile salmon (*Oncorhynchus tshawytscha* spp.; Davis et al. 2019). Therefore, all tidal wetlands support national and global biodiversity, and they share characteristics closely aligned with what is considered blue carbon as outlined in the following sections.

Tidal wetlands have long-term storage of fixed carbon dioxide

Many tidal wetlands have carbon stocks comparable to or exceeding those traditionally considered blue carbon ecosystems (figure 5). For instance, in Australia, *Melaleuca* forests have aboveground carbon stocks between 57 and 430 megagrams (Mg) of carbon per hectare (ha) and soil carbon stocks between 23 and 230 Mg of carbon per ha (0–50 centimeters [cm] deep; Tran et al. 2015, Tran and Dargusch 2016, Adame et al. 2019b). *Casuarina* forests have 143 Mg of carbon per ha, with a standard error of 61, for aboveground and 241 Mg of carbon per ha, with a standard error of 136, for belowground stocks (at a 1 meter depth; Kelleway et al. 2022). Cypress swamps and mixed (*Nyssa*, *Fraxinus*, *Alnus*) forest stands have 115 and 560 Mg of carbon per ha for aboveground and belowground stocks, respectively, with respective standard errors of 20 and 125 (1.4 meter depth; Krauss et al. 2018). However, some sites can reach at least 800 Mg of carbon per ha of belowground stocks (Krauss et al. 2018). High carbon stocks have also been measured in forests of *Picea* and *P. aquatica*, with 95.1 and 220 Mg of carbon per ha for aboveground and 614 and 844 Mg of carbon per ha for belowground carbon stocks, respectively, with

Table 1. Global dominant species of tidal wetlands that are not mangroves, seagrass or saltmarshes.

Species	Distribution	Reference
<i>Melaleuca viridiflora</i>	Northeast Australia	CABI 2019
<i>Melaleuca quinquenervia</i>	Southeast Australia	CABI 2019
<i>Melaleuca</i>	South Vietnam	Tran et al. 2015
<i>Casuarina glauca</i>	Southeast Australia	Kelleway et al. 2022
<i>Taxodium distichum</i>	Southeast United States	
Mixed forest: <i>Nyssa</i> , <i>Fraxinus</i> , <i>Alnus</i>	Southeast United States	
<i>Juncus roemerianus</i> , <i>Zizaniopsis milacea</i>	Southeast United States	Conner et al. 2007, Duberstein et al. 2014
<i>Picea sitchensis</i>	Northwest Pacific United States	Christy and Brophy 2007
<i>Mora olifera</i>	Colombia Pacific coast	Palacios Peñaranda et al. 2019
<i>Pachira aquatica</i>	Mexico Pacific coast	Adame et al. 2015
Mixed forest	Papua Indonesia	Aslan et al. 2016
<i>Leptospermum scoparium</i> (manuka)	New Zealand	Stephens et al. 2005
<i>Dacrydium dacrydioides</i> (kahikatea)	New Zealand	Robertson et al. 1991
<i>Ganua motleyana</i>	Central Kalimantan, Indonesia	Murdiyarsa et al. 2009
<i>Gluta wallichii</i>		
<i>Pternandra azurea</i>		
<i>Diospyros maingayi</i>		
<i>Ixora tenelliflora</i>		
Mixed forest: <i>Macaranga motleyana</i> , <i>Vatica oblongifolia</i> , <i>Diospyros siamang</i> , <i>Nypa fruticans</i>	Western and Southern Kalimantan, Indonesia	Saragi-Sasmito et al. 2019
<i>Metroxylon sagu</i>	West Papua, Indonesia	Taberima et al. 2014
<i>Syzygium cordatum</i> , <i>Barringtonia racemosa</i> <i>Ficus trichopoda</i>	South Africa	Veldkornet et al. 2015, Adams et al. 2016

respective standard errors of 16, 45, 86, and 38 (Adame et al. 2015, Kauffman et al. 2020b).

The highest carbon stocks have been measured in the peat swamps of Indonesia, with 168 Mg of carbon per ha for above-ground, with a standard error of 31, and 1526 Mg of carbon per ha for belowground carbon stocks, with a standard error of 126 (Novita et al. 2020), and total ecosystem carbon stocks ranging from 558 to 1213 Mg of carbon per ha (Murdiyarsa et al. 2009). In Borneo, coastal peat swamps have 168 and 1526 Mg of carbon per ha for above and belowground carbon stocks, respectively, with respective standard errors of 31 and 126 (Saragi-Sasmito et al. 2019). Therefore, the total ecosystem carbon stocks of these tidal wetlands range between 358 and 1694 Mg of carbon per ha, well within the ranges of mangroves (79–2208 Mg of carbon per ha; Hutchison et al. 2014, Atwood et al. 2017), seagrasses (9–830 Mg of carbon per ha; Fourqurean et al. 2012), and saltmarshes (100–800 Mg of carbon per ha; figure 5; Chmura et al. 2003).

The depth of organic matter (42 to at least 300 cm) in many of these tidal wetlands is comparable to that of mangroves, which have a mean of 216 cm, ranging from 22 to 600 cm (Kauffman et al. 2020a). The shallowest organic matter layers are found in *Melaleuca* forests in tropical Australia, with mean values of 41.7 cm, with a standard error of 4.4 (Adame et al. 2019b). In the peat swamps of Indonesia, organic matter is deeper, around 200 cm (Murdiyarsa et al. 2009). Comparatively, in *Picea* and sago forests, soil depth ranges from 100 to at least 300 cm (Jones et al. 2017, Kauffman et al. 2020b).

The processes responsible for accumulating and storing soil carbon in all tidal wetlands are similar. These include relatively high productivity (Finlayson et al. 1993, Srivastava and Ambasht 1996) and slow organic matter decomposition (Wallis and Raulings 2011, Middleton 2020), which are favored in waterlogged soils (Spivak et al. 2019). For instance, the organic carbon concentration of *Melaleuca* forest soil is typically associated with its water

content. Therefore, links between water content and organic soil carbon reflect seasonal inundation and the depth of the groundwater table. In addition, high carbon stocks in many forested tidal wetlands result from the layering of local production and mineral carbon transported from the catchment (Noe et al. 2016, Jones et al. 2017, Adame and Reef 2020). The sequestered carbon in the soil in these tidal forested wetlands is relatively stable and, if undisturbed, may persist for centuries (Adame et al. 2019b).

Carbon preservation in the soil of tidal wetlands is favored where litter and roots are recalcitrant and have high carbon-to-nitrogen and lignin-to-nitrogen ratios (Srivastava and Ambasht 1996, Stagg et al. 2017). Preservation varies with the type of organic matter and inundation. For labile organic matter, frequent inundation can increase decomposition through leaching; for recalcitrant organic matter, increased inundation facilitates carbon accumulation due to slowed anaerobic decomposition (Wallis and Raulings 2011, Stagg et al. 2017). Decomposition can also be enhanced by fluctuating water regimes compared with permanent waterlogged conditions (Ozalp et al. 2007). In cypress and mixed hardwood forests (*Nyssa*, *Fraxinus*, *Alnus*), the decomposition of litter and roots is significantly reduced by flooding and the incursion of saline water (Weston et al. 2006).

The net capacity of tidal wetlands to sequester carbon can only occur when the decomposition of organic matter is lower than accumulation. This is common for many tidal wetlands, where decomposition rates are low (Trevathan-Tackett et al. 2021). For example, in forests of *T. distichum*, *Paquira aquatica*, *Melaleuca*, and *Casuarina* spp. soil decay rates range between 0.001 and 0.008 per day. These values are lower than those for seagrasses (0.0002–0.03 per day; Trevathan-Tackett et al. 2020), mangroves and saltmarsh (mean ranges of 0.015–0.06 per day; Middleton 2020, Trevathan-Tackett et al. 2021, Ouyang et al. 2023) (figure 6), highlighting their capacity for soil carbon sequestration.



Figure 4. Fauna inhabiting tidal wetlands: (a) orangutans (*Pongo pygmaeus*) in peat swamps of Central Kalimantan, Indonesia; (b) frogs (*Litoria peronii*, Peron's tree frog); (c) cassowary (*Casuarius casuarius johnsonii*); (d) wallaby (*Notamacropus* sp.) and (e) flying fox (*Pteropus* sp.) in *Melaleuca* forests in Australia; (f) American alligator (*Alligator mississippiensis*) and (g) water moccasin snake (*Agkistrodon piscivorus*) in cypress or mixed forest swamps, in Louisiana, in the United States. Pictures: DM (a); LJ (b, d); Gary Cranitch, Queensland Museum (c); and KWK (f, g).

Tidal wetlands may have significant GHG removals

The accumulation of soil organic carbon in many tidal wetlands is similar to or exceeds that in blue carbon ecosystems. In Australia, *Melaleuca* wetlands accumulate soil organic carbon at a rate of 0.55 Mg of carbon per ha per year, with a standard error of 0.05. Cypress and mixed forests (*Nyssa*, *Fraxinus*, *Alnus*) have long-term accumulation rates (millennia) of between 0.07 and 3.4 Mg of carbon per ha per year (Krauss et al. 2018, Adame et al. 2019b) and decadal rates of between 1.1 and 1.8 Mg of carbon per ha per year (1963–2012; Ensign et al. 2015). These values are close to or exceed those of saltmarshes and mangroves (e.g., 0.60 and 0.55 Mg of carbon per ha per year, respectively; Chmura et al. 2003, Alongi 2020). Similar grass-dominated wetlands have higher carbon accumulation rates of 1.24 Mg of carbon per ha per year compared with saltmarshes with 0.040 Mg of carbon per ha per year (Loomis and Craft 2010). Tree uptake and carbon dioxide fixation that results in wood accumulation occur at similar rates to those of mangroves, averaging 4.0 Mg of carbon per ha per year, with a standard error of 0.2 (Xiong et al. 2019), compared with rates of *Melaleuca* and cypress or mixed forests (*Nyssa*, *Fraxinus*, *Alnus*) with mean values of 5.0 and 1.1 Mg of carbon per ha per year, respectively, and with respective standard errors of 2.1 and 0.3 (Krauss et al. 2018, Adame et al. 2019b).

GHG emissions (atmospheric plus lateral flux) must be lower than carbon sequestration for an ecosystem to be a net carbon sink. To be a net radiative carbon sink, the summed radiative forcing of carbon dioxide and methane, a potent GHG, cannot exceed their sink potential. The latter is not likely the case in many freshwater wetlands, where methane emissions counteract the atmospheric cooling effect of carbon dioxide fixation (Hemes et al.

2018). Methane is produced by methanogens during the anaerobic breakdown of organic matter and is commonly emitted from wetland soils (Al-Haj and Fulweiler 2020). However, in many tidal wetlands at or near maximum tide levels, where the soils are saline, the methane fluxes have, thus far, differed from freshwater wetlands, having lower methane emissions (Holm et al. 2016). For instance, in oligohaline *Melaleuca* forests, soil GHG emissions are lower than neighboring mangroves and those reported from other forests around the globe (table 2).

The low soil methane emissions measured in tidal wetlands could partially explain their long-term capacity to store soil carbon (Holm et al. 2016). The soils of tidal wetlands are generally enriched with marine sulphate deposits or receive marine water, at least periodically. As sulphate reduction outcompetes methanogenesis (Burdige 2012), sulphate-enriched soils produce low methane emissions. In addition, low methane emissions could result from soil uptake (e.g., Krauss and Whitbeck 2012). Although the mechanisms of these processes are unclear, changes in tidal fluctuations affecting the water table may simultaneously support anaerobic zones where methane is produced by methanogens and aerobic zones, where it is consumed or converted back to carbon dioxide by methanotrophs (Megonigal and Schlesinger 2002).

Another GHG emitted from wetlands is nitrous oxide, a product of nitrification and incomplete denitrification. Emissions of nitrous oxide are higher in warm climates and wetland soils rich in nitrifying archaea (Bahram et al. 2022). In addition, higher nitrous oxide emissions are found where nitrogen, primarily dissolved inorganic nitrogen, is high (Murray et al. 2015). However, some tidal wetlands, such as mangroves can be sinks of nitrous oxide if nitrogen concentrations are low (Maher et al. 2016). Other tidal wetlands such as *Melaleuca* forests in Australia and peat swamps in

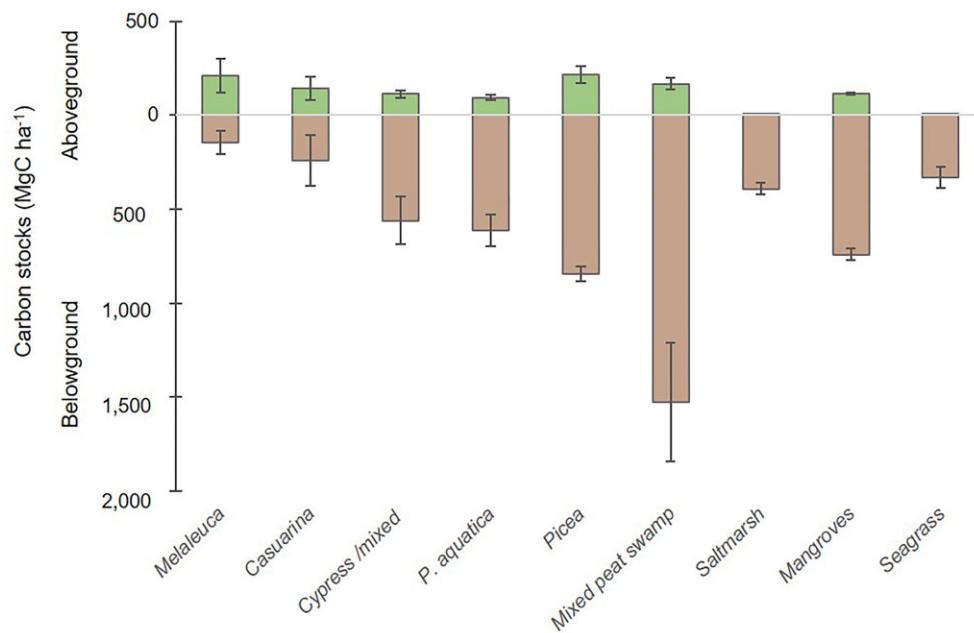


Figure 5. Carbon stocks (aboveground and belowground) of *Melaleuca* (Tran and Dargusch 2016, Adame et al. 2019b), *Casuarina* (Kelleway et al. 2022), cypress and mixed forests (Krauss et al. 2018), *P. aquatica* (Adame et al. 2015), *Picea* (Kauffman et al. 2020b), and peat swamps (Novita et al. 2020) compared with global means of saltmarsh (Alongi 2020), mangroves (Kauffman et al. 2020a) and seagrass (Fourqurean et al. 2012). Soil carbon stocks were measured for most of their organic matter layer, which ranged between 50 centimeters (in *Melaleuca* and *Casuarina* forests) and 3 meters (mangroves and peat swamps).

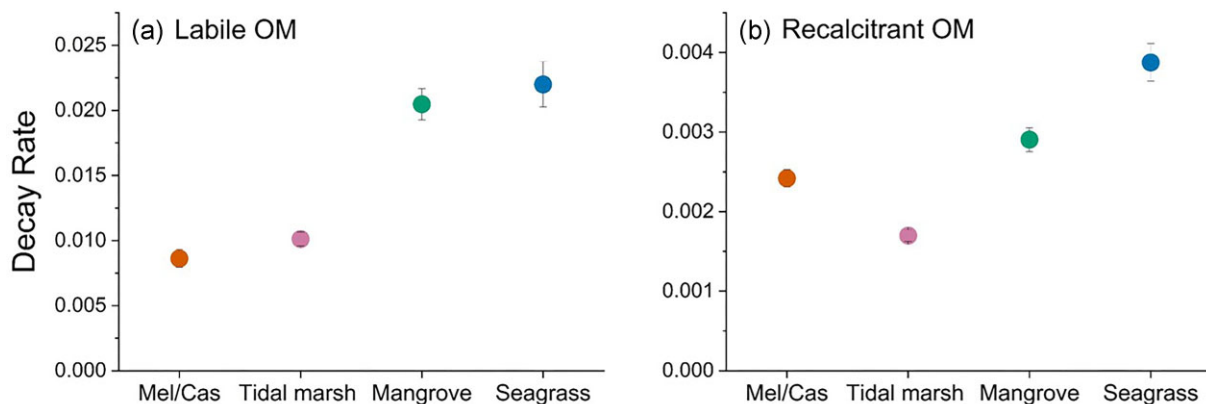


Figure 6. Organic matter decay rate (k) for standardized (a) labile and (b) recalcitrant organic matter (OM) for saltmarsh, mangroves, seagrass, *Melaleuca* and *Casuarina* (Mel/Cas) forests in Australia. The data are mean and standard error. Source: The figures are modified from Stacey Trevathan-Tackett and colleagues (2021).

Indonesia have shown similar patterns with low nitrous oxide soil emissions (-1.1 to 2.6 kilograms [kg] per ha per year), which are comparable to those of mangrove forests and saltmarshes (-0.73 to 1.2 kg per ha per year; table 2).

Although soil emissions in tidal wetlands may be low, their trees and understory vegetation may be substantial GHG sources. The methane generated in deep soils can be transported to the atmosphere via tree roots, stems and bark, potentially bypassing consumption in aerobic surface soils (Vann and Magonigal 2003, Jeffrey et al. 2020). Indeed, tree methane fluxes from some freshwater-flooded wetlands contribute to 10%–50% of their total ecosystem methane emissions (Pangala et al. 2017, Jeffrey et al. 2020, Sjögersten et al. 2020). However, some trees have communities of methanotrophs within their bark, which can consume

a third of the potential vegetation methane emissions (Jeffrey et al. 2021a, 2021b). Similar microbial communities have also been found in dead trees and branches of tidal wetlands in the Southeast United States (Martinez et al. 2022). Although further data are required to generalize on the extent of methane emissions from the trees of tidal wetlands close to maximum tide levels, so far, the data suggest that they have similar rates to those of mangroves (Jeffrey et al. 2020, Zhang et al. 2022), which are lower than for trees in freshwater wetlands (table 3).

Finally, lateral carbon movements are common in blue carbon ecosystems. For instance, about 39% of the carbon (leaf litter, wood, and sediments) stored in tidal forested wetlands in the United States is exported (Krauss et al. 2018). Dissolved forms of carbon are also exported through tidal pumping, driven by the

Table 2. Soil fluxes of methane and nitrous oxide (in kilograms per hectare per year) from tidal wetlands paired with adjacent mangroves and saltmarshes and compared with global data.

Study site	Ecosystem	Condition	Climate	Methane fluxes	Nitrous oxide fluxes	Reference
Insulator Creek, Australia	<i>Melaleuca</i>	Natural	Tropical	0.73 to 2.2	−0.73 to 2.6	Iram et al. 2022
	Saltmarsh			0.29 to 0.51	0.8 to 1.6	
	Mangroves			6.0 to 8.6	−0.08 to 0.22	
Ninds Creek, Australia	<i>Melaleuca</i>	Restored	Tropical	−3.5 to 2.2	−1.1 to 0	This study
		Natural		−1.8 to 1.1	−0.73 to 0.37	
	Mangroves	Natural		2.6 to 8.8	0 to 1.8	
Yandina Creek, Australia	<i>Melaleuca</i>	Natural	Subtropical	−0.73 to 2.9	−1.1 to 0.73	Iram et al. 2022
	Saltmarsh	Restored		2.9	−0.73 to 0.73	
	Mangroves			43.8	−1.5 to 29	
Mornington Peninsula, Australia	<i>Melaleuca</i>	Natural	Temperate	−3.0 to −1.9	0.07 to 0.26	Livesley and Andrusiak 2012
	Saltmarsh			0.15 to 0.37	−0.15 to 0.22	
	Mangroves			0.07 to 5.5	−0.07 to 0.26	
Papua, Indonesia	Peat swamp	Natural	Tropical	11.2 to 16.8	1.5 to 8.9	Swails et al. 2021
Global	Mangroves		Global	−3.7 to 4266		Al-Haj and Fulweiler 2020
	Saltmarsh			−5.4 to 5511		
	Seagrass			0.07 to 23		
	Mangroves		Global	−2.6 to 39	−6.2 to 17	Rosentreter et al. 2021

Note: Positive fluxes are emissions, and negative ones are uptakes.

Table 3. Tree stem methane fluxes extrapolated to annual aerial emissions from forest density (in kilograms per hectare per year, as in Jeffrey et al. 2020) from intertidal *Melaleuca*, *Casuarina*, and *Taxodium distichum* forests compared with mangroves and freshwater forested wetlands.

Study site	Number of trees measured	Ecosystem	Climate	Methane fluxes	Reference
Pottsville, Australia	15	Tidal <i>Melaleuca</i>	Subtropical	0.001 to 0.003	This study
		Tidal <i>Casuarina</i>	Subtropical	0.0004	
Karumba, Australia	46	Mangroves	Tropical	0.14	Jeffrey et al. 2019
Ogeechee River, United States	21	<i>Taxodium distichum</i>	Temperate	0.51	Pulliam 1992
Zhangjiang River, China	24	Mangrove	Subtropical	3.9 to 9.9	Zhang et al. 2022
Cattai Wetland, Australia	30	Freshwater <i>Melaleuca</i>	Subtropical	0.04 to 102	Jeffrey et al. 2020
		Freshwater <i>Casuarina</i>	Subtropical	8.1 to 593	

changes in hydraulic head between surface and groundwater, which results in surface water infiltrating the groundwater table during flood tides and being released back during ebb tides (Santos et al. 2012). Surface water generally has lower concentrations of dissolved GHG than groundwater (e.g., Sadat-Noori et al. 2016); therefore, the net result is the export of GHG from the forest soils to the adjacent river or estuary (figure 7; Rosentreter et al. 2021). Some of the particulate and dissolved exported carbon will be released into the atmosphere as GHG (Bogard et al. 2014), whereas the rest will be buried or exported to the ocean (Maher et al. 2018). The carbon export can also occur in the form of carbonate alkalinity (mostly as bicarbonate at a pH greater than 8), reducing acidity in the coastal ocean (Maher et al. 2018). This process provides additional benefits for climate change regulation and adaptation.

The exchange of GHG through tidal pumping is well studied in mangroves and saltmarshes (Maher et al. 2018, Schutte et al. 2020) and is an essential pathway in those blue carbon ecosystems (Alongi 2014, Chen et al. 2022). However, this process has yet to be measured in other tidal wetlands. Nevertheless, the lateral exchange is likely less critical in tidal wetlands close to maximum tide levels, which have only sporadic inundation events compared

with wetlands that are frequently flooded, such as mangroves (figure 7).

Tidal wetlands have been lost or degraded by anthropogenic impacts

Tidal wetlands are located near the coast, where human population density is high, and agriculture and other land uses are widespread and intensive (Barendregt and Swarth 2013). Therefore, they are often affected by both terrestrial and marine threats, including pollution, deforestation, land-use cover change, changes in sedimentation and hydrology, and, recently, changing climate and sea-level rise (Barendregt and Swarth 2013, Jones et al. 2017). However, the impacts of these multiple threats on many tidal wetlands are challenging to quantify. First, there is no consensus on the definition and classification of tidal wetlands, and second, their historical and current distribution is largely undescribed.

Australia has about 6.4 million ha of native *Melaleuca* forests, most in the tropical north (ABARES 2024); however, only some of these forests are located within the intertidal. Most tidal *Casuarina* forests are found in the temperate south, with an estimated

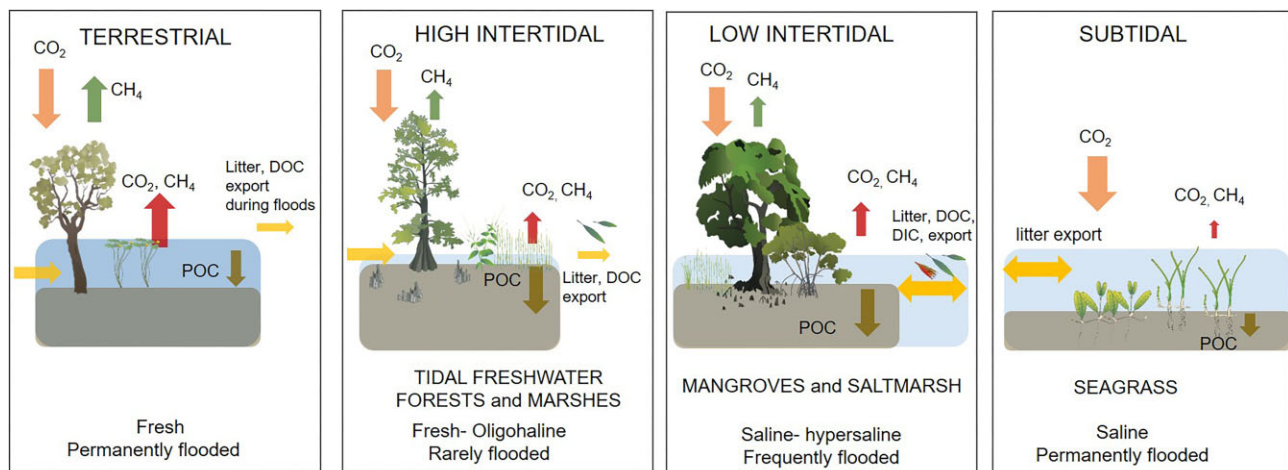


Figure 7. Carbon fluxes of freshwater terrestrial wetlands compared with intertidal wetlands (high, low, and subtidal). Source: Modified from Barendregt and Swarth (2013). Graphics: Tracey Saxby, Integration and Application Network (<https://ian.umces.edu/media-library>).

historic area in New South Wales and southeast Queensland of between 89,000 and 152,000 ha (DEE 2018). Deforestation rates of these tidal wetlands in Australia were likely to be high, because these areas are in prime agricultural land in fertile floodplains. For example, in the Herbert River catchment in northern Australia, 80% of all *Melaleuca* forests were converted to sugarcane farms in the last century (Johnson et al. 1999). The deforestation of these wetlands caused severe problems with the acidification of streams, the release of heavy metals, and the loss of biodiversity. Similar losses have occurred for tidal *Casuarina* forests, with the current distribution being almost half their historical area (more than 50,000 ha; DEE 2018). More accurate and recent data for *Melaleuca* forest distribution in Australia exists only for the state of Queensland, where 11,900 ha have been mapped and classified as *natural coastal and subcoastal floodplain and nonfloodplain Melaleuca or Eucalyptus forests*. In Queensland, the annual deforestation rate is 0.2% (2013–2017; WetlandInfo 2024).

In temperate North America, cypress or mixed forests (*Nyssa*, *Fraxinus*, *Alnus*) have had significant losses since the turn of the last century. Still, less intense land-use changes began as far back as 400 years ago, during European colonization. During this time, the development of port cities and agriculture were the leading causes for the drainage and filling of many wetlands to accommodate human infrastructure, pasture, or crops. In the seventeenth century, along the southeast coast of the United States, much of the tidal wetland area was converted to rice agriculture (Smith 2012). In the northwestern United States, most tidal wetlands were historically forested. However, over 90% of these forested tidal wetlands have been lost to diking by levees and alternate vegetation conversion, and the losses have been as high as 99% in some major estuaries (Marcoe and Pilson 2017, Brophy et al. 2019). Later, because this land was abandoned, years of land subsidence resulted in emergent freshwater or low-salinity marshes (Smith 2012). Currently, the primary threats to tidal wetlands in the United States include salinity intrusion due to sea-level rise, changing river flow patterns, damming of rivers, dredging, and other localized land-use changes, such as urban development (Jones et al. 2017, White et al. 2022).

Similar to Australia and other nations, the exact historical and current national area of tidal wetlands in the United States is unknown. However, there is information available for some regions

within the country. For instance, wetland timberland in the southeast United States, including cypress and mixed forests (*Nyssa*, *Fraxinus*, *Alnus*), was estimated at 3.9 million ha in 1990 (Tansey and Cost 1990). More recently, the total historical area of vegetated tidal wetlands for the US West Coast has been estimated at 335,230 ha, from which 85% have been lost (Brophy et al. 2019).

A similar story can be found in South Africa, where sugar cane farming, industrial development, roads, and bridges have extensively removed and degraded tidal wetlands. Altered soil conditions have encouraged habitat invasion by terrestrial and exotic invasive plants such as *Chromolaena odorata*, *Lantana camara*, and *Pereskia* species. In rural and highly populated areas, tidal wetlands are cleared illegally through unsustainable slash-and-burn practices to provide subsistence farming of bananas and vegetables (Van Deventer et al. 2021, Riddin and Adams 2022). Desiccation, burning, and erosion of the peat of these wetlands destroy the carbon sink function of these ecosystems and may cause significant emissions. Water extraction is also a threat; *Eucalyptus* and pine plantations lower the groundwater table and reduce freshwater inflows to downstream estuaries and wetlands (Bate et al. 2016). Extreme climate events, such as droughts and storms, are also of concern. Because of the low salinity tolerance of some tidal wetlands close to maximum tide levels, storm surges can cause catastrophic damage. For instance, at Mgobezeleni Estuary, in South Africa, storm swells after two cyclones caused strong winds and waves that scoured open the usually closed estuary mouth. Water stress caused by marine water intrusion into the oligohaline forest of *Ficus trichopoda* caused its death (Taylor 2016). As a result of these human and climatic pressures, 20% of the area (12,000 ha) of these wetlands was lost between 2000 and 2011 (Van Deventer et al. 2021). If this trend follows, these tidal wetlands in South Africa will likely be lost by 2060 and are, therefore, classified as Critically Endangered (Van Deventer et al. 2021).

The area of most tidal wetlands in other parts of the world is highly uncertain; however, they could exist in most coastal countries and would have likely experienced high deforestation rates. For instance, a vast 680,000 ha, classified as *transitional forests*, has been identified in Papua Indonesia alone (Aslan et al. 2016). In the Mekong Delta, in Vietnam, 99% of *Melaleuca* swamps (about 4 million ha) have been lost within 200 years of human expansion (since 1816); the loss has been exceptionally high in the past few

decades because of conversion to rice fields, urbanization, dike construction, and deforestation (Huu Nguyen et al. 2016). In New Zealand, kākūka and mānuka forests have become rare, because they were heavily deforested for agriculture (Elser and Astridge 1974). Since Māori settlement, these forests have been converted by repetitive fire and then, during colonization, by agricultural expansion (Burrows 1973). Most kahikatea forests have been lost, with only small patches remaining (Smale et al. 2005). Overall, if all tidal wetlands follow the same trend as global wetlands, more than half of their area, especially those within Asia, will probably have already been lost (Davidson 2014).

Management of tidal wetlands is practical and possible

Management of tidal wetlands is conducted through conservation and restoration. Conservation includes protection, preventing overuse, and limiting development. In contrast, restoration includes hydrological reconnection, provisioning for sea-level rise migration, removing invasive species, revegetation, managing nutrient and sediment fluxes, and restoring natural floods of saline tidal water to reduce methane emissions (Kroeger et al. 2017, Krauss et al. 2022). The management of tidal wetlands may not have previously included consideration of their ecosystem services, including carbon storage and biodiversity. However, because most tidal wetlands are affected by anthropogenic change, restoring them through carbon offset projects provides an opportunity to recover and enhance their multiple values.

In Australia, most tidal wetlands are offered legislative protection, although their protection varies within States and regions. For instance, coastal *Casuarina* forests are listed as an Endangered Ecological Community under the New South Wales Threatened Species Conservation Act 1995 and the Commonwealth Environmental Protection and Biodiversity Conservation Act 2016 (DEE 2018). For wetlands in the Great Barrier Reef region, legislation, policy, and management programs provide strong protection (Adame et al. 2019a). However, most *Melaleuca* forests (75%) in Australia are on leasehold and private land (1 million ha; ABARES 2024) presenting challenges for their management but also opportunities for landholders to participate in carbon offset restoration programs for projects within their properties.

Carbon offsets are unlikely to financially outperform urban developments, and therefore, additional benefits markets and offset schemes, such as nitrogen markets and biodiversity credits, may need to be considered together (Mack et al. 2022). There is potential for restoring intertidal *Melaleuca* and *Casuarina* forests in agricultural land that is no longer productive. For instance, in the Maroochy floodplain, large-scale restoration of previous sugarcane fields into coastal wetlands is being trialed as part of Australia's national program to boost blue carbon activities. This project aims to test the implementation of the first Australian carbon market methodology for blue carbon, which consists of reintroducing tidal inundation to restore hydrologically altered landscapes used for agriculture to coastal wetlands, including *Melaleuca* and *Casuarina* forests (Lovelock et al. 2022). In Vietnam, a successful restoration program of a *Melaleuca* forest that was lost because of a fire in 2002 resulted in the recovery of the forest and return of 156 bird species, 15 of which are important for the East Asian Australasian Flyway (Tran and Matusch 2017).

In the United States, wetland protection legislation was established under the 1972 Clean Water Act, and since then, efforts have been undertaken to manage and protect freshwater tidal

wetlands (Mihelcic and Rains 2020). For instance, cypress or mixed forests (*Nyssa*, *Fraxinus*, *Alnus*) have been treated with insecticide to protect the dominant *Fraxinus* spp. trees from invasive insects such as the Emerald ash borer (Dr. Andy Baldwin, University of Maryland, College Park, Maryland, US, personal communication, 29 May 2019). On a larger scale, restoring freshwater tidal wetlands in the United States in catchments where water flows are regulated could be possible through dam management for sediment and water delivery (Weston 2014, Ensign and Noe 2018). Historically, changes in sediments have dramatically affected tidal wetlands. Following European colonization, the intensification of agricultural practices caused erosion and sediment delivery, allowing tidal freshwater wetland areas to expand (Noe et al. 2020). Later, in the twentieth century, the implementation of soil conservation efforts reduced erosion and decreased sediment loads downstream, reducing wetland expansion (Noe et al. 2020). Therefore, dam management could be conducted to balance the necessary sediment loads to allow tidal wetlands to persist while maintaining good water quality, which requires low sediment.

The coastal wetlands of South Africa face a conservation conundrum; 62% of their area occurs within protected areas (Van Deventer et al. 2021). However, there has been degradation (measured as metrics of fragmentation and transformation) within these wetlands for the past two decades. Although legislation and management measures have been implemented, this trend has not stopped or reversed (Van Deventer et al. 2021). Slash and burn agriculture and lowering the water table from surrounding timber plantations have been identified as the primary threats. Draw-down of the groundwater table has resulted in the exposure of the soils and oxidation of the organic material (Grundling et al. 2021). These findings indicate that managing water and agricultural practices in the catchment or increasing protection in nature reserves could improve conservation outcomes for tidal wetlands in South Africa.

Finally, in Mexico, in La Encrucijada Biosphere Reserve, fires are a threat to zapotonales and brackish marshes (Adame et al. 2015). Funding from carbon projects that support fire brigades and fire management activities in the reserve could help reduce the intensity and frequency of fires in these wetlands. Many of these restoration projects are still ongoing. Studies suggest that many species of tidal wetlands, such as *Melaleuca* are fast colonizers, especially in low salinity conditions (Johnston et al. 2003, Iram et al. 2022), as has been shown in Australia and Indonesia (supplemental figure S1). Protection and management activities could be conducted globally throughout these tidal wetlands, depending on the local context and the threats these forests face.

Despite the opportunities, there are still challenges to managing and restoring tidal wetlands worldwide. Many of these challenges extend beyond knowledge gaps or technical difficulties, such as the country's political landscape and regulatory capacities. However, there is also immense potential, given the extent and global distribution of many of these previously unaccounted wetlands in blue carbon projects.

Management interventions of tidal wetlands may have no social or environmental harm

The multiple benefits of restoring and protecting wetlands are well known and accepted as a no-regrets option for GHG removal (Smith et al. 2019). However, any management activity always has trade-offs that must be considered and, if necessary, managed.

For example, carbon offset programs that include tidal wetlands can benefit farmers with land that is not profitable to cultivate; however, the inundation of agricultural land may be irreversible, and the land may no longer be suitable for most crops. Many previously tidal wetlands have been converted to agriculture, such as the Mekong Delta, in Vietnam, where rice cultivation is critical for the country's economy (Huu Nguyen et al. 2016). Similarly, tidal wetland restoration in agricultural landscapes of the US West Coast can generate social and political controversy (Breslow 2014). In these situations, the restoration of tidal wetlands could directly conflict with the immediate needs of the local farmers without adequate landscape planning (Huu Nguyen et al. 2016). In other regions, restoring or managing tidal wetlands would not typically affect ongoing agricultural interests. For example, in the southern United States, coastal rice agriculture was abandoned after the Civil War (1861–1865; Smith 2012).

There are also other local issues and values to consider that are specific to each country and region. For example, in Australia and New Zealand, tidal wetlands tend to attract undesirable nonnative animals, such as wild pigs and buffalo, that devastate biodiversity and can carry and spread diseases (Mihailou and Massaro 2021). In contrast, there are also additional benefits to restoring tidal wetlands. For example, in the Mekong Delta, in Vietnam, natural tourism has become a strong incentive and driver of restoration (Tran and Matusch 2017). It has been estimated that the annual value of protecting the Mekong Delta wetlands is US\$0.5 to US\$1.8 million (Do and Bennett 2009). Tourism of *Melaleuca* wetlands may drive local economies and increase awareness of conservation activities; however, if tourism is not well controlled, the activity could result in forest degradation (Tran and Matusch 2017). The trade-offs of restoring tidal wetlands can also be considered alongside the project goals, which may extend further than carbon—for example, for fowl hunting, birdwatching, tourism, or cultural activities.

Management interventions of tidal wetlands are aligned with policies for mitigation and adaptation to climate change

Restoration and improved management of wetlands are some of the most effective land-management options for achieving the United Nations Sustainable Development Goals (Smith et al. 2019), Aichi targets, and the Ramsar Convention. Their management may also help nations achieve their carbon emission targets under the Paris Agreement, especially in small countries with low fossil fuel consumption and high deforestation rates (Taillardat et al. 2018). Restoration and improved management of these poorly recognized tidal wetlands could also be important to offset residual emissions—that is, emissions that cannot be practically reduced, such as those created by the aviation industry.

In Australia, restoring tidal wetlands can generate carbon offsets through the Australian blue carbon methodology (Lovell et al. 2022). The offset emissions will count toward Australia's commitment to reducing GHG emissions from land conversion. The management and protection of all types of tidal wetlands are also aligned with the Reef 2050 long-term sustainability plan (Commonwealth of Australia 2021) and the State of Queensland Wetland Policy of no wetland loss (WetlandInfo 2024). Strengthening the protection and restoration of all types of tidal wetlands requires addressing gaps in legislation for blue carbon

projects (Bell-James 2022). Classifying and mapping tidal wetlands are essential to any project with crucial management implications. For instance, in Queensland, environmental values are managed as maps of matters of state on environmental significance. These maps do not always align with current tidal wetland distributions.

Wetland protection in the United States is often related to water quality maintenance, with state-level regulations often adding additional specificity by wetland type. In the United States, the most robust protections for tidal wetlands are initiated from section 404 of the Clean Water Act, passed by the US Congress in 1972. Embedded within this legislation is a provision to limit sediment discharge into aquatic habitats, including wetlands, and preserve water quality on the basis of nutrient limits. Wetland conservation and restoration are recognized strategies for climate change mitigation in the United States (Needelman et al. 2018). Several coastal states are developing natural and working lands policies incorporating GHG inventories for tidal wetlands and other ecosystems (e.g., Oregon Global Warming Commission 2021).

Although there are good examples of alignment of tidal wetland management with climate change policies, there are challenges to address. In most countries, national policies do not distinguish between wetland types. For instance, in Vietnam, high-level policies are relevant to all wetlands, with more specific regulations established in the Forest Law, Land Use Law, Fishery Law and Environmental Protection Law (Nguyen et al. 2017). In Mexico, only maps and deforestation rates for mangroves are available at the national and state levels (CONABIO 2022). Long-term monitoring for other wetlands distinguishes only *marsh* from *open water* and *other wetlands*, which include inundated rainforests and other tidal and nontidal peat swamps (CONABIO 2022). More recent proposals for wetland classification in Mexico, following the Ramsar guidelines, include estuarine versus palustrine wetlands (CONAGUA 2017). Within palustrine wetlands, the saline swamps subcategory may include some tidal wetlands that could be incorporated into blue carbon projects.

This lack of specificity on wetland types and the overlapping of legislation is a global challenge and can hamper implementation and monitoring of conservation and restoration practices. Despite barriers, protecting and restoring all types of tidal wetlands in many countries and regions will likely align with state, national, and international policies to reduce GHG emissions, adapt to climate change, and provide multiple social and environmental cobenefits.

Conclusions

We provide compelling evidence that tidal wetlands beside mangroves, saltmarsh, and seagrass have characteristics aligned with blue carbon. There is strong evidence that most tidal wetlands, even those near the highest tides, have long-term storage of fixed carbon dioxide in their soils and the aboveground biomass. In addition, there is some evidence that GHG emissions from these tidal wetlands are low, although studies of emissions from trees and lateral carbon export are scarce and a significant knowledge gap. Furthermore, most tidal wetlands near the highest tides have suffered immense losses because of ongoing anthropogenic impacts. However, some examples exist where management and restoration have resulted in enhanced carbon sequestration and positive effects on biodiversity and the local communities.

Despite the potential of including all tidal wetlands in blue carbon projects, pressing issues to be addressed. First is a globally applicable consensus definition of blue carbon and a classification of tidal wetlands. From our experience, we have provided a definition of blue carbon and a classification of tidal wetlands based on attributes that make them important for carbon sequestration and emission reductions. This classification could be applied to blue carbon wetlands globally. Our review suggests that limiting wetlands by the limits of tidal influence in the terrestrial edge provides adequate inclusion of the ecosystems that possess the biophysical attributes of a blue carbon ecosystem. This delimitation is conservative as, by definition, it excludes all wetlands that do not have marine influence. Some wetlands above the highest astronomical tides could have a positive GHG balance, where emissions are higher than their sequestration potential, and therefore will not have a net cooling effect in the atmosphere. However, some of these freshwater wetlands may not produce high levels of methane because of other processes, such as microbial methane consumption or sulphates provided by remnant marine sediments. We propose that future studies address this knowledge gap. An agreed definition and appropriate classification are key steps for overcoming the second most crucial problem: mapping their distribution. Improved mapping at appropriate scales and regular updates of all tidal wetlands are essential for protecting and managing them.

Despite these limitations, including all tidal wetlands in blue carbon projects could accelerate the restoration and protection of these ecosystems and significantly expand the scale of carbon credits generated. Many tidal wetlands have been scarcely studied, and most have not been considered for blue carbon projects. However, tidal wetlands, even those close to the tidal inundation limits, are one of the most carbon-dense ecosystems on the planet. Mechanisms that finance their protection and restoration, such as carbon and biodiversity crediting, could significantly accelerate their conservation and recovery for the benefit of humanity.

Acknowledgments

MFA was funded by an Advance Queensland Industry Research Fellowship by the Queensland Government, Australia. STT was supported by the Australian Research Council DECRA Fellowship (grant no. DE210101029). Additional support was provided by the New Zealand Ministry of Business Innovation and Employment Future Coasts Aotearoa Contract C01x2, and the US Geological Survey, Climate Research and Development Program (to KWK and GBN). Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the US government. NI was supported by funding from Temasek Holdings, Singapore.

Supplemental material

Supplemental data are available at [BIOSCI](#) online.

Author contributions

M.F. Adame (Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Writing original draft, Writing review & editing), J. Kelleway (Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Writing original draft, Writing review & editing), K.W. Krauss (Data curation, Formal analysis, Investigation,

Methodology, Writing original draft, Writing review & editing), C.E. Lovelock (Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Writing original draft, Writing review & editing), J.B. Adams (Data curation, Formal analysis, Investigation, Methodology, Writing original draft, Writing review & editing), S.M. Trevathan-Tackett (Data curation, Formal analysis, Investigation, Methodology, Writing original draft, Writing review & editing), G. Noe (Investigation, Methodology, Writing original draft), L. Jeffrey (Data curation, Formal analysis, Investigation, Methodology, Writing review & editing), M. Ronan (Conceptualization, Data curation, Formal analysis, Methodology, Writing review & editing), M. Zann (Data curation, Formal analysis, Methodology, Writing review & editing), P. Carnell (Conceptualization, Investigation, Writing review & editing), N. Iram (Data curation, Formal analysis, Investigation, Writing review & editing), D.T. Maher (Conceptualization, Investigation, Methodology, Writing review & editing), D. Murdiyarto (Investigation, Methodology, Writing original draft), S. Sasmito (Investigation, Methodology, Writing original draft), Da. B. Tran (Investigation, Writing original draft, Writing review & editing), P. Dargusch (Investigation, Writing review & editing), J.B. Kauffman (Data curation, Investigation, Writing review & editing), and L. Brophy (Investigation, Writing review & editing)

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Received: April 28, 2023. Revised: January 14, 2024. Accepted: February 6, 2024

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