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Chapter 3

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ECOSYSTEM SERVICES AND DISSERVICES OF MANGROVE FORESTS AND SALT MARSHES

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Abstract Coastal wetlands such as mangrove forests and salt marshes provide a range of important benefits to people, broadly defined as ecosystem services. These include provisioning services such as fuelwood and food, regulating services such as carbon sequestration and wave attenuation, and various tangible and intangible cultural services. However, strong negative perceptions of coastal wetlands also exist, often driven by the perceived or actual ecosystem disservices that they also produce. These can include odour, a sense of danger, and their real or perceived role in vector and disease transmission (e.g. malaria). This review provides an introduction to the ecosystem services and disservices concepts and highlights the broad range of services and chdisservices provided by mangrove forests and salt marshes. Importantly, we discuss the key implications of ecosystem services and disservices for the management of these coastal ecosystems. Ultimately, a clear binary does not exist between ecosystem services and disservices; an ecosystem service to one stakeholder can be viewed as a disservice to another, or a service can change seasonally into a disservice, and vice versa. It is not enough to only consider the beneficial ecosystem services that coastal wetlands provide: instead, we need to provide a balanced view of coastal wetlands that incorporates the complexities that exist in how humans relate to and interact with them.

Keywords: blue carbon, coastal protection, coastal wetland, cultural ecosystem services, environmental policy, environmental service, wave attenuation

Introduction

Coastal wetlands are found along low-energy shorelines worldwide, with distinct but overlapping geographical distributions. Mangrove forests are restricted to the tropics, subtropics, and some warm temperate locations, covering 137,600 km^2 in 2010 (Bunting et al. 2018). Salt marshes are predominantly found in temperate and subarctic regions, though extensive salt marshes are also found in the tropics and subtropics, where they may form an ecotone with mangrove forests. The global area of salt marsh is poorly constrained, particularly due to uncertainty in the distribution of tropical salt marsh, though this ecosystem is conservatively estimated to cover 41,700–54,900 km^2 globally (Ouyang & Lee 2014, McOwen et al. 2017).

The distribution of coastal wetlands overlaps with a zone of disproportionately high population densities (Neumann et al. 2015); thus, many populations rely on coastal wetlands for the benefits or 'ecosystem services' that they provide. Ecosystem services are most commonly conceptualised into three groups of benefits to people: provisioning services (materials directly extracted from the ecosystem, such as timber and medicinal products), regulating services (the regulation of ecosystem processes such as wave attenuation and carbon sequestration), and cultural services (ranging from tourism and recreation to aesthetic and spiritual values). These services are sustained through a range of supporting ecosystem services, such as pollination and photosynthesis. More recently, ecosystem services have been further conceptualised as *Nature's Contributions to People,* with greater emphasis on the role of culture and local knowledge (Díaz et al. 2018).

While the ecosystem services concept has been successful in promoting the importance and value of the environment, it has often faced criticism for being too anthropocentrically focused, for overemphasising economic valuation, for oversimplifying complex ecosystem processes and functions (Schröter et al. 2014, Saunders & Luck 2016), and for not encompassing the real and perceived negative impacts that ecosystems can have on human wellbeing, termed 'ecosystem disservices' (McCauley 2006, Vaz et al. 2017). Disservices provided by coastal wetlands include being a source of pests and diseases (Claflin & Webb 2017) danger (Friess 2016), and odour (Knight et al. 2017). Disservices have received relatively little attention among coastal wetland scientists compared to ecosystem services (*sensu* von Döhren & Haase 2015). Incorporating disservices into the broader environmental policy and decision-making framework, however, allows for a more holistic understanding of a stakeholder's preference for and interactions with the environment. Studying disservices also encompasses a broader set of ecosystem processes and functions that may not be the same as those producing ecosystem services (Blanco et al. 2019).

This review takes a holistic view of human interactions with coastal wetlands that incorporates both ecosystem services and disservices. We do this by conducting an in-depth literature review of the broad range of ecosystem services and disservices produced by mangrove forests and salt marshes (as conceptualised in Figure 1). We also discuss how ecosystem services and disservices can be managed to achieve effective coastal wetland conservation outcomes.

History of the ecosystem services and disservices paradigms

History of the ecosystem services concept

The reliance of humans on the benefits of nature has long been known, with Plato (~400 BC) recognising spatial trade-offs between upstream deforestation for timber and downstream impacts on soil erosion and water scarcity (Daily 1997). Notions of this relationship were later introduced in the book *Man and Nature* (Marsh 1864) which by the 1960s spurred collaborative efforts between ecologists and economists leading to the use of terms such as 'environmental services' (Wilson & Matthews 1970), 'natural capital' (Schumacher 1973), and 'nature's services' (Westman 1977). In particular the term 'ecosystem services' (Ehrlich & Ehrlich 1981) quickly gained traction in the 1980s–90s, culminating in two seminal publications Daily (1997) and Costanza et al. (1997). The Millennium Ecosystem Assessment (MEA 2005) later defined categories of ecosystem services and mainstreamed the concept into national and international policy. In order to increase the utility of the concept, subsequent initiatives have refined the definitions and categories of ecosystem services (e.g. the European Union's Common International Classification of Ecosystem Services [CICES] and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [IPBES]) and

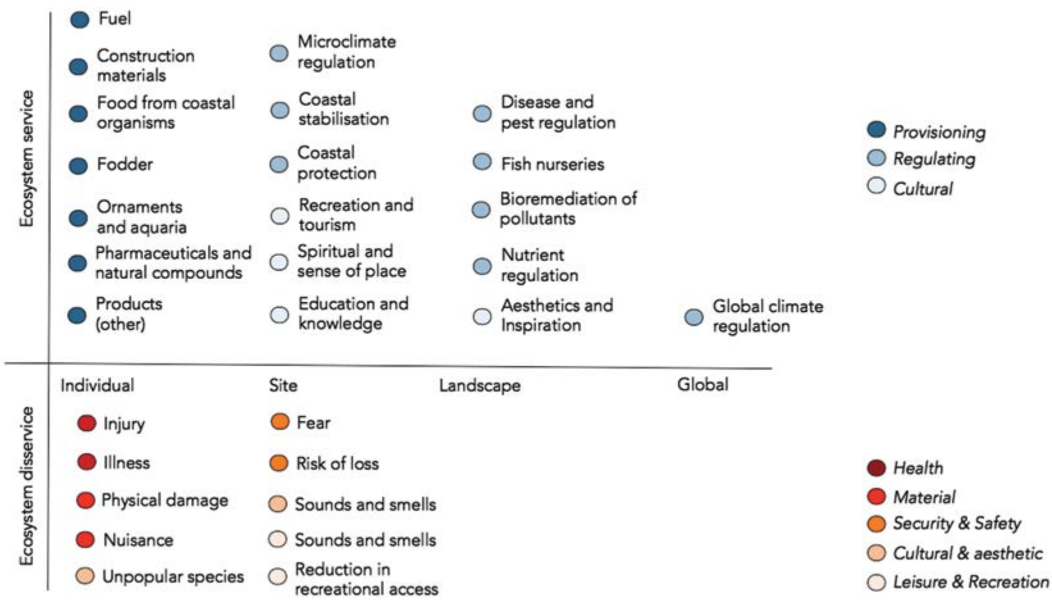

Figure 1 A conceptual diagram of ecosystem services and disservices in coastal wetlands.

communicated them to different audiences such as businesses (e.g. The Economics of Ecosystems and Biodiversity [TEEB] framework).

Critiques of ecosystem services and the emergence of the ecosystem disservices concept

The concept of ecosystem services has received considerable criticism (Schröter et al., 2014) for its anthropocentric view of nature, inconsistencies between valuation schemes (Gómez-Baggethun et al. 2010, Braat & de Groot 2012), and ethical issues related to the commodification and economic valuation of nature (McCauley 2006, Turnhout et al. 2013). Additionally, by focusing on ecosystem benefits, the concept of ecosystem services has also been critiqued for its positive bias and inability to reflect negative components i.e. ecosystem disservices (Lyytimäki & Sipila 2009, Dunn 2010, Lele et al. 2013).

Similar to ecosystem services, ecosystem disservices have been described for centuries through various historical descriptions (e.g. Friess 2016 for coastal wetlands). Ecosystem disservices, however, are a much more recent academic concept compared to ecosystem services (Blanco et al. 2019), so typologies and frameworks are not as clearly defined. Disservices were first categorised according to aesthetic, safety, security and health, economic, and mobility disservices (Lyytimäki et al. 2008) and then financial costs, social nuisances, and environmental pollution (Escobedo et al. 2011). Shackleton et al. (2016) have undertaken one of the more rigorous ecosystem disservice typologies, defining disservices as the 'functions, processes, and attributes that resulted in perceived or actual negative impacts on human wellbeing and describing many of the important considerations for their categorisation. Others have subsequently expanded this and categorised disservices into health, material, security and safety, cultural and aesthetic, and leisure and recreation disservices (Vaz et al. 2017).

Ecosystem disservices have themselves been criticised for oversimplifying complex ecosystem processes, hampering conservation efforts, and potentially leading to undesirable economic outcomes and justifications (Dunn 2010, Lyytimäki 2015). A desire to consider disservices within the dominant ecosystem services framework, however, represents a fundamental paradigm shift in understanding human-environment interactions, recognising that nature can have both beneficial and harmful impacts on human wellbeing, both of which must be managed (Shackleton et al. 2016, Schaubroeck 2017). In reality, ecosystem services and disservices are not binary but can influence stakeholders at the same time or in the same location (Saunders & Luck 2016).

Ecosystem services of coastal wetlands

Coastal wetlands provide a range of ecosystem services that support human wellbeing in a number of ways. These include provisioning services such as food from fisheries and plant products, fuels, and fibre; regulating services such as coastal protection through wave attenuation, water quality improvements to nearby coastal areas through nutrient assimilation and sediment trapping, and climate regulation via carbon sequestration and storage; and cultural ecosystem services such as recreation, education, and spiritual value (Table 1). Cultural ecosystem services are particularly understudied in coastal wetlands, in part because they are non-material, often intangible, and rarely remain constant (Thiagarajah et al. 2015, Queiroz et al. 2017).

Provisioning ecosystem services

Construction materials

Coastal wetlands are an important source of materials for construction. This ecosystem service is particularly provided by mangrove forests, as their durability, hardness, and resistance to rot and pests make trees such as *Rhizophora* spp. a highly desirable source of timber for subsistence and commercial purposes (Uddin et al. 2013, Friess 2016). At the subsistence level, mangrove forests provide timber for the construction of houses, fencing, and boats (Knox & Miyabara 1984, Palacios & Cantera 2017). *Rhizophora* spp. are commonly used for home construction in South and Southeast Asia and South America, though *Heritiera fomes* and *Excoecaria agallocha* were also historically used in the Sundarbans of Bangladesh and India (Bandaranayake 1998). *Avicennia* spp., *Xylocarpus* spp., and *Barringtonia asiatica* are preferred for boat building in the Pacific islands, while *Sonneratia alba* is preferred in Madagascar (Bandaranayake 1998). Similarly, mangrove-associated plants (often shrubs), and many salt marsh species (e.g. *Juncus kraussii*, *Spartina alterniflora*, and *Phragmites spp.*) provide thatch used in the construction of farmhouses and homes (Russell 1976, Köbbing et al. 2013, Cunningham 2015). Fronds of the palm *Nypa fruticans* are a common roofing material in Indonesia and Malaysia, known as *attap* (Baba et al. 2013).

Commercially, the large-scale mangrove forestry trade was instrumental in the expansion of Spanish naval fleets in Central America in the 19th century (López-Angarita et al. 2016). Mangrove trees were the primary material to construct telecommunication poles, without which the reach of telecommunications in some parts of East Africa and Asia would have been limited (Semesi 1998). Despite the wider availability of timber resources today, mangrove timber continues to be extracted, often for fencing posts. Poles may be the product of thinning during mangrove forestry operations for the production of charcoal.

Fuel

Many mangrove species, particularly those in the *Rhizophora* genus, are highly valued as a source of fuelwood and charcoal, because their high calorific value makes them a preferred fuel source compared to other trees (Bandaranayake 1998). *Rhizophora* spp. are slow-burning and release a high amount of heat with little smoke (Walters et al. 2008). Historically, mangroves were used as fuel for trade ships connecting European and Asian markets (Friess 2016), and naval fleets in Latin America (López-Angarita et al. 2016). Their importance to the Spanish empire was such that mangrove wood

Table 1 Summary of major ecosystem services provided by coastal wetlands

Ecosystem service	Description	Examples of benefit
Provisioning ecosystem		
Construction materials	Timber and construction materials derived from coastal wetland trees, shrubs, and associated flora	Mangroves such as *Rhizophora* spp. are used as a source of timber due to their hardness, pest-resistance, and rot-resistance properties (Palacios & Cantera 2017) The common reed, *Phragmites australis,* has long been used as both roof and wall thatch (Köbbing et al. 2013). Other saltmarsh species have been used for a similar use (Russell 1976)
Fuel	Energy sourced from coastal wetland biomass	Mangroves such as *Rhizophora* spp. are used directly as fuel or converted to charcoal due to their high calorific value (Malik et al. 2015) The common reed, *Phragmites australis,* is used as a source of fuel, burned after pelletisation, or used as a raw fuel source for methane production (Köbbing et al. 2013, Wichmann 2017)
Products (other)	The extraction of wild or farmed biota for human use	Tannins are extracted from mangrove bark used in the preservation of leather (Higake 1987) and dyeing textiles and fabrics (Punrattanasin et al. 2013) The mat rush, *Juncus kraussii*, is used in basket weaving, bedding, and the production of various wedding crafts and handicrafts for export in southern Africa (Cunningham 2015)
Food from coastal wetland organisms	Extraction of coastal wetland faunal biomass	Numerous fish and shellfish species are caught or gleaned from mangroves and salt marshes for food (Carney 2017, Castagno 2018)
Ornaments and aquaria	Extraction of biota for decoration, fashion, handicraftand souvenirs,, and so on or for display in aquaria	Juvenile fish, crustaceans, amphibians, and snakes can be extracted from mangroves and salt marshes for use in aquaria (Sandiliyan 2016)
Fodder	Biomass used to feed domesticated animals	Mangrove biomass is used for cattle fodder in East Africa (Semesi 1998). Saltmarsh grasses are used for cattle fodder in Australia and Canada (Rogers et al. 2006, Gedan et al. 2009)
Pharmaceuticals and natural compounds	Extraction of coastal wetland biota in order to produce medicines or pharmaceuticals	Ecteinascidin 743, derived from the mangrove ascidian *Ecteinascidia turbinata*, is used as an anti-cancer drug in the treatment of soft tissue sarcoma (Newman & Craig 2004). Compounds extracted from mangrove tissue can also be used as an insecticide (Bandaranayake 1998) Saltmarsh spp. have been used to treat gastrointestinal issues (especially *Salicornia* spp.) and are recognised for their potential anti-inflammatory, anti-oxidant, and anti-cancer properties (Rhee et al. 2009, Oueslati et al. 2012)

(*Continued*)

Table 1 (*Continued*) Summary of major ecosystem services provided by coastal wetlands

Ecosystem service	Description	Examples of benefit
Regulating ecosystem services		
Global climate regulation	Carbon sequestration and storage by salt marshes and mangroves (known as 'blue carbon'), particularly in their soils	In mangroves, mean carbon sequestration rates range from 174–224 gC m^{-2} $year^{-1}$ (Alongi 2012, Hopkinson et al. 2012), and carbon stocks are estimated to be on average 956 TC ha^{-1} (Alongi 2014) In salt marshes, carbon sequestration rates range from 57–218 gC m^{-2} $year^{-1}$ (Chmura et al. 2003, Hopkinson et al. 2012), and carbon stocks are estimated to be on average 593 TC ha^{-1} (Alongi 2014)
Microclimate regulation	Cooling of local climate by evapotranspiration and changes in albedo	Mangrove vegetation can cool local temperatures (Beserra de Lima & Galvani 2013) through evapotranspiration, and to a lesser extent, this effect may be expected in salt marshes
Coastal protection	Dampening or reduction in the intensity of hydrodynamic energy by vegetation and topography	Coastal wetlands contribute to coastal protection by attenuating wave energy, though this is non-linear with coastal wetland width (Koch et al. 2009). Mangrove vegetation can reduce incoming wave heights by 80% (Brinkman 2006) Saltmarsh vegetation can reduce storm surge wave height by ~20% (Möller et al. 2014)
Coastal stabilisation	Soil conservation within the ecosystem	The complex above-ground root structures of many mangrove species contribute to sediment trapping (Krauss et al. 2003) and reduced erosion Similarly, in salt marshes, high above-ground shoot density and dense rooting networks result in both trapping of sediments and limited erosion (Stumpf 1983)
Bioremediation of pollutants	Contribution of coastal wetland biota to the removal of pollutants through storage, dilution, transformation, and burial	The nutrient and heavy metal retention ability of mangrove forests contributes to their use in the treatment of wastewater (Tam & Wong 1993) Similarly, salt marshes are also able to improve coastal water quality by removing heavy metals from industrial wastewater, including Zn, Pb, and Cu (Caçador et al. 1996)
Nutrient regulation	Storage, internal recycling, processing, and acquisition of nutrients	Mangroves such as *Sonneratia* spp. can reduce nutrients such as nitrites, ammonia nitrogen, and reactive phosphorus from water by as much as 89%, 86%, and 38%, respectively (Huang et al. 2012) Nutrient cycling and N and P assimilation by *Spartina* and *Sarcocornia* saltmarsh spp. contributes to reducing eutrophic conditions (Sousa et al. 2010)
Fish nurseries	Provision of refuge habitat and nursery grounds which are crucial for survival in early life stages of commercial, recreational, and other ecologically important species	Mangrove forest extent is linked to offshore fisheries catches (Whitfield 2017) and are particularly important nurseries for emperor fish (*Lethrinus* spp.) in many parts of Indonesia, Japan, and west Africa (Nakamura & Sano 2004, Unsworth et al. 2009) Salt marshes have historically been important nurseries for blue crab (*Callinectes sapidus*) fisheries in the Gulf of Mexico (Rees 1963)

(*Continued*)

Table 1 (*Continued*) Summary of major ecosystem services provided by coastal wetlands

Ecosystem service	Description	Examples of benefit
Cultural ecosystem services		
Recreation and tourism	Provision of natural landscapes that supply an appropriate space for recreational activities, which can enhance physical and mental health and wellbeing	Mangrove forests provide opportunities for leisure activities such as fishing, boating, and social gatherings (Queiroz et al. 2017) and are hotspots for nature-based tourism (Uddin et al. 2013) Common recreational activities in salt marshes include bird watching, hiking, watersports, and wildfowling (Feagin et al. 2010, Jones et al. 2011)
Aesthetic appreciation and artistic inspiration	Provision of visual, aural, tactile, or olfactory enjoyment and/or a source of inspiration for artistic creations from natural landscape scenery and features	Mangrove forests have provided artistic inspiration for paintings (Cumming 2008) The tranquillity of salt marshes has been expressed via sculpture, poetry, and books (Wiegert & Pomeroy 1981, Seabrook 2012)
Scientific and educational knowledge	Ecosystems (or their features) which provide the opportunity for the development of scientific research, knowledge systems, and environmental education practices	Mangrove forests and salt marshes support a variety of opportunities to further the understanding of wetland ecosystems on a local and regional level, such as educational school visits and the engagement of local communities through an expanding network of wetland education centres (Ramsar Convention Secretariat 2010), and on a global scale through the publication of scientific research papers and other forms of scientific value (Greenberg et al. 2006)
Spiritual and cultural heritage and sense of place	Provision of specific areas of religious, spiritual, or cultural significance or social value, including World Heritage Sites, folklore, lifestyle values, media specifically associated with culture and heritage	Spiritual festivals such as Rush Mela (Uddin et al. 2013) and celebrations of other deities (Jalais 2014) are held within mangroves by local communities Areas in South Carolina, US, are revered as a 'sacred' by Gullah Geechee people, who hold strong cultural and heritage ties to tsalt marshes (Seabrook 2012)

Figure 2 Local-scale mangrove harvesting for charcoal, Tanakeke Island, Indonesia. (Photo by Jared Moore [National University of Singapore].)

became part of the tax or 'tribute' that indigenous communities had to pay the Spanish king (López-Angarita et al. 2016). Today, some small-scale charcoal production is conducted at the community level, which can have negative impacts on local mangroves if not regulated effectively (Brown et al. 2014; Figure 2). Most charcoal production, however, is produced through large forestry concessions, with complex supply chains that produce charcoal for national and regional markets. For example, Matang Mangrove Forest Reserve in Malaysia has been managed for forestry purposes since 1902, and produces as much as 179 tonnes of biomass per hectare each year from harvested plots (Ismail et al., 2017). Forestry production in Matang is not without consequence, however, as species diversity and wood yields have declined over time (Goessens et al. 2014).

Few saltmarsh species are used as fuel sources, but the common reed, *Phragmites australis,* is commonly used as a source of fuel by direct burning, being made into fuel pellets, or used to produce biogas via anaerobic digesters (Köbbing et al. 2013, Wichmann 2017). This is most popular in northern Europe and north-eastern North America.

Food from coastal wetland organisms

Many coastal communities depend on coastal wetlands for subsistence, owing to the wide variety of biodiversity they support, including offshore fisheries, invertebrates (Figure 3), mammals, birds, and plants (Hutchison et al. 2014). For many coastal communities, fish and shellfish derived from these ecosystems are the main source of dietary protein (e.g. Carney 2017). Mangroves and salt marshes provide fish and other marine species with vital spawning grounds and nurseries to raise their young and provide a habitat for shellfish, such as oysters and snails, thus supporting a highly productive and diverse food source. Historically, the food security afforded by mangrove forests may have led to the settlement of nomadic Middle East communities along the coast ~6500 years ago, as interior areas became more arid and less productive (Biagi & Nisbet 2006). More recently, fisheries derived from mangrove forests (e.g. Aburto-Oropeza et al. 2008) and salt marshes (e.g. Castagno 2018) constituted significant contributions to subsistence and commercial markets, such that coastal wetlands are

Figure 3 Examples of coastal wetland organisms used for food, including dried fish in Sulawesi, Indonesia (a); prawns in Sumatra, Indonesia (b); mangrove crabs in New Caledonia (c); and octopus in Madagascar (d). (Photos by authors.)

valued at over USD \$1,000 ha^{-1} yr^{-1} for fisheries alone (De Groot et al. 2012, Costanza et al. 2014). In many instances, the abundance and exploitation of food resources creates a number of livelihood opportunities (Siar 2003, Glaser & Diele 2004, Magalhães et al. 2007; Figure 3).

Plant resources extracted from coastal wetlands are also an important food source. In mangrove forests, sap from the Nypa palm (*Nypa fruticans*) is commonly tapped to produce sugar, vinegar, or alcohol, and its fruits are used for food in both raw and processed forms (Hamilton & Murphy 1988). The fruits and propagules of *Bruguiera* spp., *Sonneratia* spp., and *Avicennia* spp. are all used to produce flour for baking, and the leaves of *Acanthus* spp. are used for tea (FAO 1996). In salt marshes, *Salicornia* spp. are collected for use as a vegetable or the base for vinegar and fermented beverages (Patel 2016).

Pharmaceuticals and natural compounds

Chemical extracts from coastal wetland organisms are widely used in many parts of Asia, Africa, Latin America, and the Caribbean, both in traditional and modern medicine, to treat a range of ailments including asthma, skin diseases, diabetes, cancer treatments, inflammation, tumours, viruses, ulcers, and animal venom. The medical properties of coastal wetland vegetation are typically concentrated in their leaves, fruits, flowers, roots, seeds, and resins, but recently, biomolecules are also being identified and extracted from otherwise overlooked components of the coastal wetland ecosystem, including microbes, fungi, algae, insects, and herpetofauna (Bandaranayake 1998, Cunningham 2015). Within traditional medicine, extracts from *Bruguiera* spp. are used by local communities in the treatment of tumours and viral infections (Knox & Miyabara 1984). Extracts from *Xylocarpus* spp., *Ceriops* spp., and *Rhizophora* spp. have also been used in the treatment of diarrhoea and haemorrhaging (Bandaranayake 1998). In pharmaceuticals, HIV-1 inhibitors have been characterised from the mangrove associate, *Calophyllum inophyllum* (Patil et al. 1993). Anti-viral, analgesic, and anti-parasite biomolecules have been identified from *Avicennia* spp. and used in the treatment of leprosy, hepatitis, and smallpox (Majumdar & Patra 1979, Sharma & Gard 1996, Ito et al. 2000).

The use and exploration of saltmarsh vegetation for medicinal biomolecules is not as advanced as for mangrove forests. However, recent biomolecular studies have highlighted the potential of saltmarsh flora as a resource for biomolecules with broad application in modern medicine. For instance, extracts from the saltmarsh *Salicornia herbacea* show potential application as an antibacterial, antidiabetic, antiproliferative, antioxidant, anti-inflammatory, and in diabetes treatments (Patel 2016). *Salicornia herbacea* extracts have also been traditionally used in the treatment of gastrointestinal ailments and obesity (Rhee et al. 2009). *Suaeda fruticosa* has also been evaluated for a variety of antioxidant, anti-inflammatory, and anti-cancer compounds (Oueslati et al. 2012).

Regulating ecosystem services

Global climate regulation

Coastal wetlands contribute to the regulation of the global climate through their ability to sequester and store carbon dioxide from the atmosphere. High productivity (Odum 1959) coupled with low decomposition rates (Patrick & DeLaune 1977) in their anoxic soils results in a predominantly net positive balance between aboveground and belowground tissue, litter production, and organic matter decomposition (Charles & Dukes 2009). This high productivity results in mangrove forests and salt marshes sequestering and storing 3–5 times more carbon per hectare than other vegetated ecosystems (Chmura et al. 2003, Donato et al. 2011).

In mangrove forests, mean carbon sequestration rates range from 174–224 gC m^{-2} $year^{-1}$ (Chmura et al. 2003, Alongi 2012, Hopkinson et al. 2012), and carbon stocks are estimated to average 956 Mg ha^{-1} (Alongi 2014). In salt marshes, carbon sequestration rates are estimated to be slightly lower, ranging from 57–218 gC m^{-2} $year^{-1}$ (Chmura et al. 2003, Hopkinson et al. 2012), with their resulting carbon stocks estimated to average 593 Mg ha^{-1} globally (Alongi 2014). Several factors contribute to the global variation in carbon sequestration rates and stocks observed in both ecosystems. At the largest scale, climate (temperature, precipitation and potentially extreme weather events) determines the productivity of the wetland ecosystem and the amount of biomass that is produced (Sanders et al. 2016, Feher et al. 2017, Simard et al. 2019). More locally, coastal geomorphology is a key factor in determining carbon sequestration rates and stocks through the import of nutrients from rivers or other sources, tidal regime, and underlying substrate (Rovai et al. 2018, Twilley et al. 2018). Carbon sequestration rates and carbon stocks are thus highly variable across space, and their quantification requires a sound understanding of large-scale climatic influences and local-scale edaphic conditions (e.g. geomorphology, temperature, freshwater availability) and species composition.

At national and international policy levels, mangrove forests and salt marshes have been described as ‘blue carbon’ ecosystems, alongside seagrasses (Lovelock & Duarte 2019) and tidal freshwater forested wetlands (Krauss et al. 2018). Blue carbon ecosystems have received a large amount of attention globally for their ability to sequester and store carbon. However, scale is crucial in assessing the contribution of blue carbon to global climate regulation. The ability of coastal wetlands to regulate carbon is highest at the plot scale (Figure 4a) but largely insignificant at the

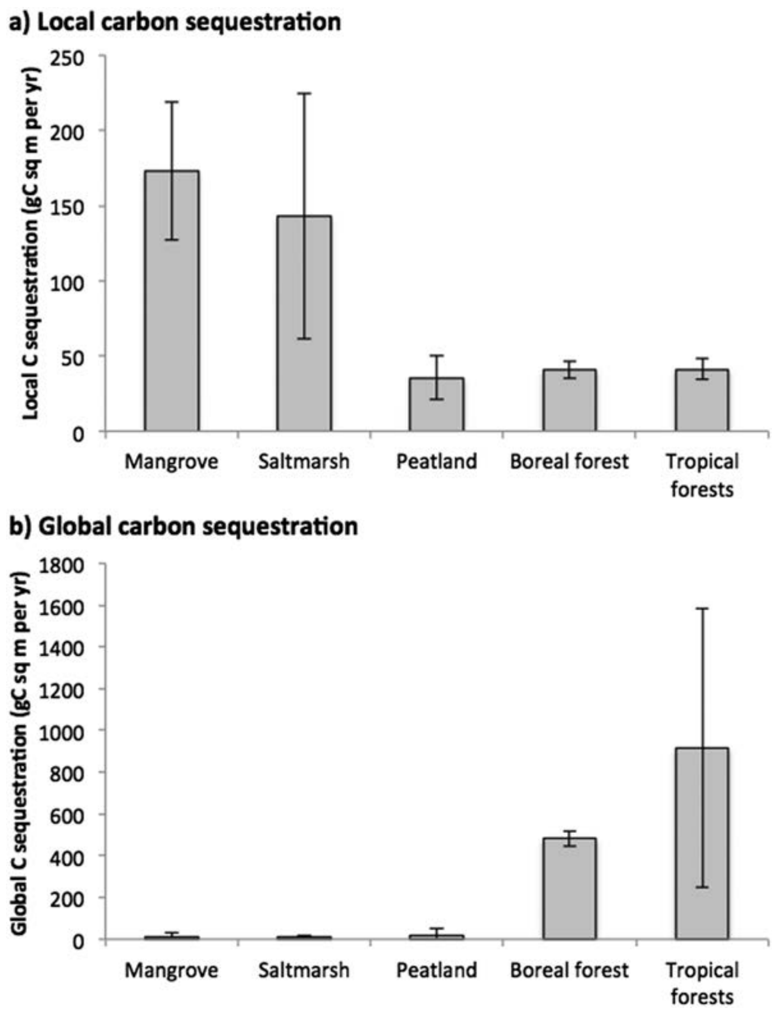

Figure 4 Comparison of carbon density for mangroves, salt marshes, and selected terrestrial ecosystems at the plot scale (a) and the global scale (b). (Data from Taillardat et al. 2018. *Biology Letters* 14, e20180251.)

global scale (Figure 4b). Combined, mangrove forests and salt marshes only account for ~0.8% of global carbon sequestration by vegetated ecosystems, due to their smaller global extent compared to other terrestrial ecosystems with a lower per hectare carbon density (Taillardat et al. 2018). It is at the national scale (for countries with long coastlines) where mangroves and salt marshes may have the most impact on climate regulation.

Coastal protection

Coastal wetlands provide protection to people and property by buffering the impact of storm surges and coastal flooding (Guannel et al. 2016, Hochard et al. 2019). This is particularly important given that over 625 million people lived in the coastal zone in 2000 with an expected increase to more than 1 billion by 2060 (Neumann et al. 2015). Mangrove forests and salt marshes protect shorelines by reducing incoming wave energy through reflection and dissipation. Dissipation occurs largely as a result of the friction generated by the physical structure and roughness of vegetation (including pneumatophores, aerial roots, trunks, and stems) (Mazda et al. 2006, Wamsley et al. 2010).

Under normal tidal and weather conditions, mangroves attenuate wave energy and wave height over short distances. *Rhizophora*-fringed mangrove forests can reduce wave energy and wave heights by as much as 71% and 79%, respectively (Brinkman 2006). *Sonneratia* spp. can attenuate as much as 50% of incoming wave energy within a distance of 100 m (Mazda et al. 2006). In addition to forest width, tree density and species are also key factors in determining the rate of attenuation; in mixed-species mangrove forests, low-density mangrove forests composed of *Avicennia* and *Sonneratia* spp. attenuated 83% less wave energy compared to high-density *Rhizophora*-dominated forests under normal conditions (Horstman et al. 2014).

Under extreme tidal or coastal hazard conditions (i.e. tsunamis and large storm surges), the effectiveness of mangroves to protect coastlines remains unclear given the paucity of empirical observations. This is particularly the case for tsunami events. The 2004 Southeast Asian tsunami spurred huge research interest in the role of mangroves in attenuating high-magnitude waves, particularly after early studies correlated lower levels of tsunami damage with larger areas of mangroves in front of coastal communities (e.g. Danielsen et al. 2005, Kathiresan & Rajendran 2005). However, subsequent analyses have tempered conclusions about the role of mangroves in attenuating tsunami waves, due to misinterpretation of potential causal mechanisms and the role of slope and distance from shore, rather than mangrove cover alone (Kerr et al. 2006). While their role in protecting against tsunami events may be limited, there is a general consensus that mangroves can still confer resilience to the coast and offer greater protection to human life and property than unvegetated coasts under storm conditions (Hochard et al. 2019), such as the Odisha tropical cyclone in 1999 (Das & Vincent 2009).

Salt marshes under normal conditions attenuate as much as 85% wave energy compared to 28% for unvegetated tidal flats (Möller et al. 1999, Möller & Spencer 2002, Yang et al. 2008). Trait differences among different species of salt marsh species (e.g. stem height, flexibility, density, leaf characteristics, and stem diameter) directly influence the extent of wave attenuation (Möller 2006, Rupprecht et al. 2017). For instance, an area of *Spartina alterniflora* attenuated wave energy 2.5 times greater than *Scirpus mariqueter*, likely due to its greater height and biomass providing greater resistance (Yang et al. 2008, Ysebaert et al. 2011).

Similar to mangroves, there are few empirical observations of wave attenuation by salt marshes under extreme conditions. Large-scale flume studies suggest that even a thin fringe of saltmarsh vegetation can attenuate storm surge waves by as much as 20% while still remaining resilient to damage caused by waves (Möller et al. 2014). The degree to which saltmarsh vegetation can attenuate extreme waves is species specific (Rupprecht et al. 2017), with implications for the upscaling of results from low species diversity flume studies to more complex field settings. As a long-term coastal buffer, water depth thresholds may limit the utility of salt marshes in building coastal resilience (Möller et al. 2001), especially when compared to much taller and more rigid mangrove trees. As such, a larger area of salt marsh is required to attenuate the equivalent amount of hydrodynamic energy as a mangrove stand (Doughty et al. 2017). Salt marshes are also vulnerable to bank erosion due to normal waves and tidal cycles, which eventually results in the collapse of marsh edges and the long-term deterioration of the salt marsh (Möller 2006, Tonelli et al. 2010, Fagherazzi et al. 2013).

Coastal stabilisation

In tandem with the direct protection of coastlines, mangroves and salt marshes can mitigate coastal erosion and reduce the vulnerability of people and property (Arkema et al. 2013). Under normal conditions, mangroves and salt marshes stabilise sediments through a number of mechanisms. Roots and shoots resist and slow the flow of water promoting the deposition of suspended sediment and inhibiting its resuspension (Furukawa & Wolanski 1996, Christiansen et al. 2000). Sediments are then mixed with organic matter and consolidated within interlocking belowground roots, a process which further binds sediments and slows rates of erosion by preventing sediments from being entrained and lost by near-bed currents (Feagin et al. 2009). Over time, these processes can lead to the vertical and lateral build-up of land through accretion. Vertical accretion in mangroves can be as much as 12 mm yr^{-1} in some locations (Alongi 2008), and the role of vegetation in encouraging sediment deposition means that accretion rates inside the coastal wetland can be several times higher than accretion in neighbouring unvegetated areas (Marani et al. 2007). Thus, coastal ecosystems provide an ecosystem service by reducing the vulnerability of people and property to coastal erosion by consolidating intertidal surfaces through sediment deposition, stabilisation, and accretion.

Consequently, in areas where vertical accretion rates contribute to positive surface elevation changes in minerogenic systems that exceed projected sea-level rise (SLR), coastal wetlands have been suggested as a possible natural mitigation measure to coastal flooding and erosion. Saltmarsh species (such as *Spartina alterniflora*) have been exported globally (from North America to coastlines across South America, Europe, South Africa, and China) over the last two centuries (Ainouche & Gray 2016). This species was chosen because it has many of the characteristics of a wetland pioneer species: it is fast-growing, can grow in the low intertidal zone, has high stem density that encourages sedimentation, and quickly creates a dense root mat that consolidates sediments (Friess et al. 2012). The growth strategy of *S. alterniflora* is so successful that the species is now invasive beyond the locations where it was originally introduced, with expensive control and eradication programmes required for its removal (e.g. Jardine & Sanchirico 2018).

Nutrient regulation

Related to processes that trap and stabilise suspended sediments in coastal wetlands are co-occurring nutrient regulating ecosystem services. Coastal wetlands are highly productive systems with a strong influence on nutrient cycling and regulation in the coastal zone, which translates into two distinct ecosystem services. First, mangrove forests and salt marshes act as a crucial link between terrestrial and marine ecosystems and can account for an integral portion (sometimes >40%) of dissolved nitrogen exported to coastal waters (Valiela 1995). Thus, coastal wetlands provide an ecosystem service by enriching and regulating broader estuarine and coastal food webs that humans rely on through detrital production and nutrient processing (Boesch & Turner 1984, Turner 1993). Second, land-use change and terrestrially derived organic pollution mean that coastal wetlands receive large inputs of nutrients (Tobias et al. 2001). Mangrove forests and salt marshes alleviate these impacts and improve water quality by transforming, recycling, and removing excess nutrients, such as nitrogen and phosphorus, from the water column (Mitsch et al. 2001). Nitrogen is mostly absorbed as nitrates by coastal wetland plants, either from the available pool of nitrates or due to the activity of nitrogen-fixing bacteria (Craft 1997).

Removal rates of nutrients by coastal wetlands and their soils are influenced by temperature, soil moisture, species and age, soil redox, density, hydrology, geomorphology, and other edaphic conditions (Feller et al. 1999, Cott et al. 2018, Bourgeois et al. 2019). Among salt marshes, *Sarcocornia* spp. and *Atriplex* spp. are efficient at nitrogen removal, whereas *Spartina* spp. are best suited for the removal of phosphorus (Sousa et al. 2010). Pioneer vegetation tend to be net importers of nutrients, whereas older vegetation are net exporters (Hughes & Paramor 2004, Lovelock et al. 2010). These factors result in high spatial variability of ecosystem service provision.

Fish nurseries

Transient and resident communities of fishes and invertebrates utilise mangrove forests and salt marshes for food, shelter, and refuge (Nagelkerken et al. 2015, Whitfield 2017). Many of these species are important to commercial fisheries (Lugendo et al. 2007, Nagelkerken 2009). Complex root and stem structures create shelter for juveniles from larger predators, which alongside high food abundance creates an environment that can support high densities of juveniles (Verweij et al. 2006, Nagelkerken et al. 2010). This forms the basis of the nursery ecosystem service, where higher densities of juveniles can be found in coastal wetlands, which contributes to fish and invertebrate catches and associated food security (Nagelkerken 2009).

Evidence is available for this ecosystem service for both mangrove forests and salt marshes. Mangrove forests have long been recognised as a nursery area, especially to tropical reef fish (Mumby et al. 2004, Nagelkerken et al. 2008, Unsworth et al. 2009). Some species of fish may also prefer certain subhabitats over others for their nursery functions, such as the preference of *Avicennia* spp. pneumatophores over *Rhizophora* spp. prop roots for some fish species (Rönnbäck et al. 1999). Salt marshes support blue crab (*Callinectes sapidus*) fisheries, historically one of the largest commercial fisheries in the Gulf of Mexico, United States (Thomas et al. 1990). Similarly, significant higher densities of penaeid shrimp inhabit salt marsh-dominated estuaries than unvegetated habitats (Raoult et al. 2018).

The role of coastal wetlands as a nursery habitat, however, is not without controversy. Connectivity between coastal ecosystems makes it difficult to attribute nursery services to a single ecosystem such as a mangrove forest or a salt marsh, and it is difficult to prove if such a service is permanent or if fish use coastal wetlands as a nursery opportunistically (Whitfield 2017). Correlations between offshore fish catches and coastal wetland extent are not always statistically significant (Loneragan et al. 2005), and where correlations do exist, they may be driven by broader estuary characteristics rather than the coastal wetland itself (Manson et al. 2005). While more research is required to better quantify the nursery function of coastal wetlands, it is clear that mangroves and salt marshes are a key component of the coastal seascape for a wide variety of aquatic species.

Cultural ecosystem services

Recreation and tourism

Recreation and tourism opportunities are some of the most common cultural benefits that people derive from coastal wetlands (Himes-Cornell et al. 2018), which as an industry contributes substantially to local economies. Costanza et al. (1997) estimated the global value of recreational services (which was partially calculated from usage fees) to be US$815 billion yr^{-1}, of which US$574 billion yr^{-1} could be attributed to wetlands. Recreational and tourism opportunities in mangrove forests and salt marshes range from the non-extractive such as walking, photography, bird watching, social gatherings, and ecotourism (Davidson et al. 2017, Queiroz et al. 2017) to the extractive, such as fishing and hunting (Kelleway et al. 2017). Underpinning this service in many instances is the rich biodiversity that coastal wetlands support (Feagin et al. 2010, Uddin et al. 2013). For example, the Sundarbans mangroves are home to over 300 species of flora and 425 species of fauna, some of which are endangered flagship species, such as the Royal Bengal tiger (Biswas et al. 2007). Its biodiversity value saw parts of the Sundarbans recognised as a UNESCO World Heritage Site, which has helped promote tourism opportunities (Salam et al. 2000); the Tiger Reserve alone attracted almost 175,000 visitors and permits in 2015 (Bhattacharyya et al. 2018). The revenue from ecotourism has provided substantial economic benefits to the surrounding area and fostered community management (Khanom et al. 2011, Uddin et al. 2013). Similarly, salt marshes and adjoining mudflats are often visited for their high biodiversity. Salt marshes support large numbers of migratory and resident birds, which has made them popular among tourists and birders (Burger et al. 1995, Klein et al. 1998, Myatt et al. 2003). Feagin et al. (2010) attributed differing recreational values to various zones within the salt marsh, with the salt flat and the high marsh recording high values for birding and hunting, owing to their being prime bird habitat, and the low marsh flagged as high value as the habitat supported recreational fishing activities.

Aesthetic appreciation

The aesthetic quality of a landscape can have a positive effect on human wellbeing and health (Hermes et al. 2018) by fostering mental rejuvenation, triggering positive emotions, and improving moods, whilst nurturing social interaction and advocating physical activity (Chang et al. 2008, Russell et al. 2013). Coastal wetlands have aesthetic appeal, as they have particular features that are unique in evoking a sense of true wilderness (Smardon 1978). This allows mangrove forests and salt marshes to be iconic and perceived as examples of outstanding beauty, bolstered by their relative scarcity in many landscapes.

Mangroves are an integral part of the coastal landscape that uniquely exist at the intersection of land and sea, and within this broad context are viewed as a place for rest and reflection by many (Kaplowitz 2001, Rönnbäck et al. 2007, Queiroz et al. 2017). The mystery and complexity of the extensive vistas of intricate waterways and dense mangrove canopies (Odum et al. 1982) have also been the motive for musical compositions, such as a composition in Australia titled *Mangrove*, by Peter Sculthorpe in 1979 and artwork by Sidney Nolan and Ian Fairweather in 1961, which conjured imagery of this 'alien environment' (Cumming 2008).

Salt marshes are similarly highly valued for their natural beauty (Wiegert & Pomeroy 1981, Casagrande 1997a) and have an enduring history of influencing landscape painting, literature (such as *The Snow Goose* by Paul Gallico), and poetry (such as *The Marshes of Glynn* by Sidney Lanier). These sources provide romanticised accounts of the vast expanses of wilderness and natural beauty of coastal wetlands or the wildfowl associated with the landscape (Jones et al. 2011, Seabrook 2012).

Spiritual value and sense of place

Many groups attach spiritual or religious value to coastal wetlands. With many local communities having lived alongside neighbouring coastal wetland ecosystems for generations, the traditional rights, practices, and knowledge gained from their plural interactions are invariably intertwined in the culture of these communities (Diegues 2002, Walters et al. 2008). In some instances, spiritual values are attributed to specific coastal wetlands, resulting in these sites being considered holy or sacred (Verschuuren 2006).

The spiritual value of mangrove forests materialises from peoples' contact with nature and is enhanced by specific components of the system, such as the spiritual significance of water and heightened sense of wilderness that people may experience in this unfamiliar habitat (Queiroz et al. 2017). In Brunei, cultural and spiritual beliefs are inextricable from the maintained practice of traditional lifestyles and customs (Islam & Yahya 2017). Similarly, in the Sundarbans, spiritual festivals such as Rush Mela (Uddin et al. 2013) and celebrations of other deities (Jalais 2014) still continue within the mangroves by local Hindu communities. Additionally, the use of mangrove roots in totemic carvings seen to be of spiritual value is widespread in cultures spanning Indonesia to northern Australia, a practice that continues today (Kelleway et al. 2017).

Both mangrove forests and salt marshes can provide spiritual value and a sense of place for communities that have traditionally been displaced or marginalised. Local accounts for salt marshes of the Gullah Geechee community who have lived on the Sea Islands from North Carolina to Florida since the 1600s describe how the area holds particular importance for the descendants of slaves, as a 'sacred place' where their history, heritage, and culture were founded in the salt waters and marshes and, as such, held physical, emotional, and spiritual roots of their present day existence (Seabrook 2012). Similarly, in coastal Louisiana, Cajuns, Native Americans, and escaped slaves utilised the extensive coastal wetlands as their home (Gramling & Hagelman 2005). These productive habitats provided both abundant resources and protection to marginalised communities and continue to contribute to their modern culture and sense of place.

The value of coastal wetland ecosystem services

Methods of ecosystem service valuation

The promotion of ecosystem services in recent decades was often triggered by the realisation that these crucial benefits are underestimated in decision-making (Hein et al. 2006). As such, ecosystem service valuation has become a prominent field in both ecological economics and environmental science (Atkinson et al. 2012). The complex and numerous ecosystem services provided by coastal wetlands require ecosystem service valuation to take a multifaceted approach using myriad market and non-market approaches (Birol et al. 2006, Table 2).

Despite our increasing knowledge of coastal wetland ecosystem services, values attributed to these services are not well represented in the literature (Himes-Cornell et al. 2018). In general, it is understood that both mangroves and salt marshes are undervalued economically (Brander et al. 2012). Coastal wetlands can be difficult to value because they are ‘public goods’, and society cannot be excluded from receiving that service, nor can the use of a benefit by one beneficiary alter how it is provided to another (Brander et al. 2012). Complications arise when attempting to value such services, as their underlying ecological functions vary spatially and temporally and may also have a degree of connectivity which should be considered during any valuation exercise (Barbier et al. 2011).

Coastal wetland ecosystem services are further undervalued in decision-making because many are ‘non-market’ goods and therefore difficult to quantify in purely monetary terms. An analysis of coastal ecosystem service valuation studies shows that market value analysis of provisioning services was much more common than the valuation of regulating or intangible cultural services (Himes-Cornell et al. 2018). Cultural ecosystem services such as inspiration for art, culture, and design are particularly underrepresented and undervalued (Himes-Cornell et al. 2018).

Table 2 Economic valuation methods for coastal wetland ecosystem services

Ecosystem service	Economic valuation methods											
	MA	PFA	NFI	R/SC	COI	TCM	HP	CV	CE	DAC	PGL	DEC
Provisioning services												
Construction materials	X											
Fuel	X											
Products (other)	X											
Food from coastal wetland organisms	X											
Ornaments and aquaria	X											
Fodder	X											
Pharmaceuticals and natural compounds	X											
Regulating services												
Global climate regulation		X		X						X	X	X
Microclimate regulation		X		X						X	X	X
Coastal protection	X	X		X						X	X	X
Coastal stabilisation	X	X		X						X	X	X
Bioremediation of pollutants				X	X					X		
Nutrient regulation				X	X							
Fish nurseries	X	X	X					X	X			
Disease and pest regulation				X	X			X	X			
Cultural services												
Recreation and tourism						X	X	X	X			
Aesthetic appreciation and artistic inspiration								X	X			
Scientific and educational knowledge								X	X			
Spiritual and cultural heritage and sense of place								X	X			

Source: Adapted from Birol et al. 2006. *Science of the Total Environment* **365**, 105–122.

Abbreviations: (PFA), production function analysis; (NFI), net factor income; (R/SC), replacement/substitution cost; (MA), market analysis; (COI), cost of illness; (TCM), travel cost method; (HP), hedonic pricing; (CV), contingent valuation method; (CE), choice experiment method; (DAC), damage avoidance costs; (PGL), productivity gains and losses; (DEC), defensive expenditure costs.

Estimating the global value of coastal wetlands

Despite the limitations outlined previously, several studies have attempted to aggregate values for coastal ecosystem services at national to global scales (Brouwer et al. 1999, Woodward & Wui 2001, Brander et al. 2006, 2012). Most large-scale ecosystem service valuations are conducted using benefit transfer, assuming a constant unit of ecosystem service value per hectare of each type of ecosystem, which is then multiplied by the area of each ecosystem type to produce aggregated totals (Batker et al. 2008). This approach is useful when trying to aggregate values on a national or international scale using scarce data; however, it assumes that an ecosystem provides services uniformly across its range. For coastal wetlands, ecosystem status and service provision vary significantly across space due to population density (Rao et al. 2015) and climatic (e.g. Ouyang et al. 2017, Simard et al. 2019) and geomorphological variation (Twilley et al. 2018).

The first notable study to estimate global coastal wetland ecosystem service value was conducted by Costanza et al. (1997), which valued mangrove forests and tidal marshes at US$9,990 ha^{-1} yr^{-1} (US$1995, converted to US$13,786 ha^{-1} yr^{-1} in US$2007; Costanza et al. 2014). More recently, values were aggregated again by De Groot et al. (2012), who estimated the global value of ecosystem services provided by coastal wetlands as high as US$193,843 ha^{-1} yr^{-1} (US$2007). The substantial increase in ecosystem service value estimated by the latter study does not necessarily indicate an increase in the value of ecosystem services over time but is instead more likely a reflection of an increase in research effort coupled with more robust analysis techniques (Costanza et al. 2014).

Ecosystem services and coastal wetland policy

Another way to understand the value of coastal wetland ecosystem services is to see how they have been used and valued by policy makers. Several international policy initiatives have incorporated the ecosystem services provided by mangrove forests and salt marshes. For example, the Ramsar Convention on Wetlands (2018) describes how the ecosystem services of coastal wetlands can contribute substantially to all of the United Nations' Sustainable Development Goals (SDGs). The SDGs are a set of 17 priorities established to help countries improve sustainable economic development, ensure social safeguards, and encourage environmental protection. Many coastal wetland ecosystem services contribute to livelihoods, which can help countries move towards achieving SDG 1 (*End Poverty*). The provisioning services of coastal wetlands also contribute to the achievement of SDG 2 (*End Hunger*). The carbon sequestration potential of coastal wetlands makes them suitable for achieving SDG 13 (*Climate Action*), while coastal wetlands also contribute to fisheries and healthy oceans (SDG 14, *Life Below Water*).

Blue carbon is being increasingly discussed in the context of global climate change policies, such as Article 5 of the Paris Agreement of the United Nations Framework Convention on Climate Change. While mangrove forests and salt marshes may not substantially impact the global carbon cycle, their contributions to carbon sequestration may be important at the national scale for countries with long coastlines and lower carbon emissions (Taillardat et al. 2018).

The coastal protection services of coastal wetlands contribute to the aims of the Sendai Framework for Disaster Risk Reduction, a recent initiative by the United Nations Office for Disaster Risk Reduction to increase interdisciplinary collaboration and opportunities for risk reduction against hazards (Aitsi-Selmi et al. 2015). The use of the natural environment to reduce hazard risk through 'ecological engineering', 'building with nature', 'ecosystem-based adaptation', or 'grey-green infrastructure' (Morris et al. 2018, 2019) are attempts to achieve the aims of the Sendai Framework by promoting ecological disaster risk reduction (eco-DRR) through the use of ecosystem services that sustainably regulate hazards (Faivre et al. 2018). Wetlands can be incorporated into broader integrated coastal management planning to reduce risk to coastal hazards (Wanger et al. 2020).

Ecosystem disservices of coastal wetlands

Ecosystem disservices are largely understudied in most ecosystems, and this is especially the case for coastal wetlands. As such, categorisations and frameworks are less developed compared to the larger field of ecosystem services, and we still lack an operationable and locally adaptable classification of ecosystem disservices (Blanco et al. 2019). This review utilises and adapts one of the most recent ecosystem disservice frameworks (Vaz et al. 2017) to allow broad comparability with existing ecosystem service frameworks. Ecosystem disservices have been divided into five categories: *health ecosystem disservices* include the direct or indirect negative impacts of biota and their existence on human physical and/or mental health and wellbeing; *material ecosystem disservices* are those that cause a nuisance or physical damage to built infrastructure; *security and safety ecosystem disservices* are those that directly or indirectly disrupt physical, personal, national, or financial safety and security; *cultural and aesthetic ecosystem disservices* represent the direct or indirect impacts of an ecosystem that contribute to cultural and spiritual disconnection with the environment; and *leisure and recreation ecosystem disservices* are those that reduce the demand for recreational opportunities. Examples of these categories are given in Table 3. There is substantial overlap between ecosystem disservice categories, and the fuzzy and perceived nature of many ecosystem disservices means that they may span several categories at once (Vaz et al. 2017).

Health ecosystem disservices

Specific components of mangrove forests and salt marshes have the potential to cause physical or mental harm to people, whether through injury, illness, or distress. Such components may include plants (e.g. thorns), animals (insects, aggressive interactions with macrofauna such as crocodiles and monkeys), or diseases that may be present in these environments.

It was long considered that coastal wetlands were a source of diseases such as malaria, though the exact mechanism by which disease was transferred has changed. Disease was originally associated with their odour; indeed, the etymology of the word 'malaria' involves the Italian phrase for 'bad air' (Hempelmann & Krafts 2013). For nearly 2000 years, it was assumed that diseases were transmitted from mangrove forests and salt marshes through their odour of decaying organic matter, or 'miasma'. For example, colonial explorers in the Zambezi Delta in East Africa considered miasmatic air emanating from mangroves to carry the 'death-germ' (Rankin 1890). Colonial explorers in Central America considered mangroves to be 'generating unhealthy miasmata' (Fitzroy 1853), which may have limited efforts to construct a canal or railroad across the isthmus. The miasma theory was supported by respected scientists at the time such as Alexander von Humboldt, resulting in the wide acceptance of the theory (Browne 1944). It was only with the advent of modern medicine and germ theory that diseases were understood to be bacterial or viral in origin and transferred by vectors such as mosquitoes (particularly *Aedes vigilax, A. camptorhynchus, A. albopictus, Verrallina butleri*, and *Culex sitiens*) instead of bad air (Hempelmann & Krafts 2013). Thus, while diseases are still a disservice caused by coastal wetlands, the perceived mechanism by which this disservice operates has changed.

Mosquitoes are common in coastal wetlands, supported by components of the ecosystem such as vegetation, standing water, microtopographic variation, and moist substrate (Dwyer et al. 2016, Rowbottom et al. 2017). Mosquitoes and associated vector-borne diseases remain a common public health concern today and can have substantial impacts on wellbeing and economic productivity. Several integrated mosquito control strategies are employed to reduce this ecosystem disservice in urban and peri-urban areas. This includes the application of chemical larvicides, reduction of ecosystem components (such as standing water) that encourage larval growth, and creating buffer areas between coastal wetlands and human settlements (Dwyer et al. 2016).

Table 3 Summary of direct or indirect, perceived, and/or actual ecosystem disservices provided by coastal wetlands

Real or perceived ecosystem disservices	Description	Examples of disservice
Health ecosystem disservices: affecting human physical or mental health		
Injury, irritation, or reaction	Negative interaction with plants, animals, or abiotic components caused by exposure, consumption, and/or contact	Death and/or injuries sustained by wildlife or habitat encounters; examples in mangroves include: crocodile attacks, snake bites, cuts by sharp or uneven vegetation (Friess 2016) Biting insects are often a serious nuisance and source of irritation in and near salt marshes (Scott & Littig 1962)
Illness, infection, viral, or disease	Physical or emotional discomfort, pain, or incompetence caused by exposure, consumption, transmission, or interaction with flora, fauna, bacteria, viruses, or zoonotic infections or diseases	Coastal wetlands provide habitat for vectors of malaria (Dunn 2010) Salt marshes and surrounding estuaries can harbour vibrio bacteria, especially in shellfish populations, that may result in illness if consumed (Colwell et al. 1981)
Material ecosystem disservices: damaging the human built environment		
Physical damage	Physical damage to built infrastructure caused by ecosystem components	Saltmarsh wrack deposition can lead to clean-up costs (beaches, lawns, docks) and unsightly and less available beaches (Gisselman 2014) and may result in allowing unwanted species to invade an area
Nuisance	Irritants to built infrastructure	Animal excrement or leaf litter from coastal wetland organisms can stain pavements and jetties (Vaz et al. 2017)
Security and safety ecosystem disservices: disrupting physical, personal, national, and/or financial stabilisation		
Fear of habitat, pests, plants, and/or animals	Feeling of unsettlement when provoked by nature	In early accounts, mangrove forests were described as 'dark, gloomy, and dangerous'. These actual and/or socially constructed negative perceptions were caused by fear of unknown habitats, wildlife, and indigenous peoples (Friess 2016) Salt marshes were associated with danger and higher crime rates (Casagrande 1996)

(*Continued*)

Table 3 (*Continued*) Summary of direct or indirect, perceived, and/or actual ecosystem disservices provided by coastal wetlands

Real or perceived ecosystem disservices	Description	Examples of disservice
Risk of loss or harm	Financial deficits acquired by way of physical or environmental functions, processes, and/or attributes	Diseases associated with mangroves can lead to decreased financial wellbeing due to healthcare costs and inability to work (Knight et al. 2017) High density grazing of Arctic-breeding geese in coastal salt marshes disturbs and exposes the soils to erosion and flooding and can release carbon emissions (Buij et al. 2017)
Cultural and aesthetic ecosystem disservices: impacting mental, cultural, and spiritual interactions with nature		
Sounds, smells, or sights of biota, landscapes, or seascapes	Reduction, disruption, or inhibition of cultural and spiritual interactions with nature	Historical accounts often described mangroves as 'dark', 'gloomy', 'fetid', and 'dismal' (Friess 2016). Mangroves often form dense vegetation, and today landowners may feel that this blocks their view of the landscape/seascape nearby (De Luca 2015) Both mangrove forests and salt marshes are commonly described as having foul odours. Derived from the anoxic conditions of coastal wetlands, sulphurous compounds result in a 'rotten-egg' smell and are often seen as unpleasant and decrease lifestyle amenity (Gurran 2008, Harty 2009, Friess 2016)
Unpopular species	Ecosystem components that are a religious, traditional, or cultural taboo	Some coastal wetland animal species are associated with evil spirits (Jalais 2014)
Leisure and recreation ecosystem disservices: inhibiting physical interactions with nature		
Sounds, smells, or sights of biota, landscapes, or seascapes	Reduction, disruption, or inhibition of recreational interactions with nature	See *Cultural and aesthetic ecosystem disservices*
Reduction in recreational access	Ecosystem components that reduce recreational opportunities due to perceived or actual nuisance or injury	Presence of mosquitoes may reduce demand for recreation

Source: Adapted from Vaz et al. 2017. *Ecosystem Services* **23**, 94–107

Security and safety ecosystems disservices

Ecosystem disservices related to security and safety are those that have the potential to disrupt a person's physical, personal, or financial stability. These can be actual or perceived situations of hardship and can range from feelings of uneasiness to remorse. Since this disservice is largely cultural and perceived, the magnitude of this disservice differs between individuals.

Due to their dynamic position in the margin between the terrestrial and marine zones, coastal wetlands have long been viewed with suspicion, particularly by those who were not familiar with these ecosystems. For example, British explorers in West Africa during the mid-1800s described mangroves as impenetrable and dark due to the density of foliage and roots, all of which contributed to the coastal landscape feeling sombre (Bacon 1842). Many salt marshes have been viewed as places of crime, dangerous due to pollution, or, similar to mangroves, 'gloomy' in nature (Casagrande 1996). One such study showed almost half of respondents perceived their nearby salt marsh to be a dangerous place due to crime (Casagrande 1997b).

Safety and security disservices extend beyond perceived notions of insecurity to threats to physical safety. Historically, mangroves were avoided by explorers because they were considered to be home to 'dangerous' indigenous populations who would attack ships (e.g., see accounts by Smith & Dalrymple 1860). This ecosystem disservice was particularly apparent during the exploration of the Australian coast by British explorers, where aboriginal groups would use mangroves to retreat or remain from view (e.g. Birtles 1997).

Cultural and aesthetic ecosystem disservices

Similar to cultural ecosystem services, cultural, material, and aesthetic disservices are difficult to categorise and measure because they are influenced by sociodemographics, experiences and knowledge, and personal or spiritual beliefs. Different individuals may or may not find particular landscapes aesthetically pleasing. Such views and perceptions vary greatly among individuals, so the type and level of ecosystem disservice experienced differs from person to person (Lyytimäki et al. 2008). While there is strong evidence to suggest that the stark nature of coastal wetlands inspires substantial levels of aesthetic ecosystem services in many people what is considered 'aesthetically appealing' differs between individuals. This means that some stakeholders consider such coastal wetlands to produce aesthetic disservices due to their bleakness. Historical colonial expeditions often described novel mangrove forests as 'dark', 'gloomy', 'fetid', and 'dismal' (Friess 2016), and explorers noted that mangrove forests had 'few attractions to the lover of the picturesque' (Rankin 1890) because of the primeval look caused by their dense root systems. Similarly, salt marshes were considered 'bleak, appalling, boundless, treeless landscapes' (Zwart 2003).

Aesthetic disservices have also provided literary inspiration, in a similar manner to aesthetic ecosystem services. Charles Dickens drew inspiration from the Thames marshes in the United Kingdom for the bleak and solitary wilderness as a backdrop for an angry sky in the opening chapters of the novel *Great Expectations* (Hynes 1963). Such an example highlights how the aesthetic disservice provided by coastal wetlands can act as a broader negative metaphor.

Consequences of ecosystem disservices

Historical coastal wetland loss

Ecosystem disservices can influence the action of stakeholders to a greater degree than ecosystem services (Blanco et al. 2019), and coastal wetlands are a great example of this. Historically, coastal wetlands have seen high rates of loss due to anthropogenic influences. Coastal wetlands were often perceived as wastelands with little economic value and the source of ecosystem disservices and thus were converted to land for agriculture, aquaculture, and industry. It is believed that up to 87% of the world's freshwater and coastal wetlands have been lost since 1700, with 35% of all coastal wetlands lost since 1970 (Ramsar Convention on Wetlands 2018).

The conversion of coastal wetlands has been practiced in North America, Europe, Africa, China, and elsewhere for centuries to millennia (Bertness et al. 2004, Davidson 2014, Knight et al. 2017). In Europe, urbanised coastlines now account for >50% of coastlines in the Mediterranean Sea, and 15,000 km^2 of coastal wetlands, tidal flats, and other coastal features have been converted in the Wadden Sea alone (Airoldi & Beck 2007). In China, at least 5,352 km^2 of coastal wetlands have been lost since 1978, and remaining coastal wetlands have been subject to pollution, degradation, and overexploitation (Meng et al. 2017). In North America, coastal wetland loss has been dramatic in both urban environments (e.g. Boston: >75% loss of coastal wetlands, Bromberg & Bertness 2005), as well as on the regional scale (e.g. northern Gulf of Mexico: 0.86% loss per year from 1955–1978, Baumann & Turner 1990). Loss and degradation due to agriculture has been common both via direct conversion or through use of coastal wetlands for livestock grazing (Gedan et al. 2009).

In comparison to salt marshes, large-scale mangrove forest loss occurred relatively recently, with coarse estimates suggesting that ~35% of the world's mangrove forests were potentially lost between 1980 and 2000 alone (Valiela et al. 2001). Approximately 1 million hectares of mangroves in Indonesia have been lost since 1800 (Ilman et al. 2016), 12%–25% of all of Thailand's mangroves were lost to shrimp ponds from 1961–1993 (Dierberg & Kiattisimkul 1996), and there was 12% total mangrove loss in Southern and Southeastern Asia from 1975 to 2005 (Giri et al. 2008). The majority of this mangrove loss has resulted from agriculture, aquaculture, and urbanisation (Giri et al. 2008). Encouragingly, rates of mangrove loss have reduced globally since the turn of the 21st century and are now only 0.3%–0.6% per year, though some countries such as Myanmar and Malaysia still experience rates of deforestation that are substantially above the global average (Hamilton & Casey 2016).

While coastal wetland loss has been significant through direct conversion for economic gain, other reductions in wetland area have occurred due to explicit attempts to reduce their ecosystem disservices, such as for the control of mosquito populations (Knight et al. 2017). Thought to be a haven for disease-carrying mosquitoes coastal wetlands have often been subject to intensive ditching and efforts, particularly in North America, Australia, and Europe (Dale & Hulsman 1990). For example, draining efforts lead to the digging of dikes and drainage ditches in 95% of coastal wetlands in the northeast United States as part of efforts to reduce mosquito populations (Buchbaum 2001).

Negative public perceptions of coastal wetlands

With an increasing knowledge of the ecosystem services coastal wetlands provide to communities, it would be expected that public perceptions of these ecosystems would now be different from the historical perceptions that drove coastal wetland loss. While this may be largely true, a negative perception of coastal wetlands still remains with many people today because of the long history of ecosystem disservices discourse associated with these ecosystems. The now common American political phrase 'drain the swamp' has its origin in ecosystem disservices, where the odour and mosquitoes associated with freshwater and coastal wetlands are used as a metaphor for lobbyists and bureaucrats. This phrase has a long history, and authors have argued that draining the swamp is associated with a masculine, colonial mindset of taming the wilderness and conquering nature and its disservices (Giblett 1996).

Lingering negative perceptions of coastal wetlands may be due in part to their poor advertising. A survey of major international media outlets by Duarte et al. (2008) showed that 73% of all newspaper articles on coastal ecosystems focused on coral reefs. Salt marshes and mangrove forests accounted for only 6.5% and 20% of newspaper articles, respectively. The media is a key channel to communicate the importance of coastal wetlands and challenge perceptions of disservices. Since Duarte et al.'s study, coastal wetlands have continued to be in the news, though still not to the degree of other coastal ecosystems. A rapid search of the Google News platform (a news aggregator and search engine) in June 2018 showed that of $\sim$434,900 articles written about coastal ecosystems, 85% of articles focused on coral reefs, and only 8.7% and 6.5% of articles were written about salt marshes and mangrove forests, respectively. Poor representation in the media may affect the communication of ecosystem services, providing a challenge to tackling common misconceptions of coastal wetlands linked to ecosystem disservices.

Managing ecosystem services and disservices holistically

Ecosystem services are a key approach to support environmental conservation, so highlighting and quantifying ecosystem disservices has been considered by some to hinder conservation efforts (Lele et al. 2013). Ignoring ecosystem disservices in environmental management, however, may be counterproductive, since ecosystem disservices strongly influence stakeholders' decisions (Blanco et al. 2019) and increase the likelihood of (often unanticipated) negative interactions between ecosystems and people. For example, if a disservice such as odour from a mangrove forest is not defined and characterised by managers, then it is harder to plan for its mitigation or management. Disregarding ecosystem disservices can cause local stakeholders not to buy into management decisions such as coastal wetland restoration (Handel 2016).

Instead of ignoring ecosystem disservices entirely, management may be more successful if disservices are integrated into a more holistic framework of ecosystem management and stakeholders are educated to understand ecosystem disservices, why they occur, and how they can be managed. Knight et al. (2017) propose a conceptual framework for integrating coastal wetland ecosystem services and disservices for better decision-making. Based on 30 years of experience of salt marsh management in southeastern Australia, it allows managers to enhance ecosystem service provision while mitigating potential disservices.

Incorporating disservices into a holistic framework of environmental management also allows managers to understand the tradeoffs caused by their decision-making. In order to make a reasoned and informed decision regarding any potential tradeoffs, managers should ensure that they have enough information to do so. To realistically consider all consequences of management decisions made in trade-off scenarios, it is pertinent to not only consider the valuation of ecosystem services but also of ecosystem disservices.

Future research directions

The field of coastal wetland ecosystem services has attracted huge recent research interest, and, as a result, our knowledge in this area is relatively advanced. However, significant knowledge gaps still remain, particularly around the quantification of intangible cultural ecosystem services and the integration of ecosystem disservices into ecosystem services frameworks:

a. *Cultural ecosystem services*. In general, little is known about cultural ecosystem services compared to other ecosystem service categories, and this is even more the case when considering coastal wetlands (Queiroz et al. 2017). A consideration of cultural ecosystem services is essential because they are a clear link between ecosystems and people and so may be some of the most important to consider during coastal management. A strengthened research focus on cultural ecosystem services will give us a more holistic view of the contribution of coastal wetlands and provide more evidence for their conservation, especially in urban settings where coastal wetland-human interactions are greatest.

b. *Value of ecosystem services.* Scientific knowledge of the full range of ecosystem services that coastal wetlands provide has become increasingly advanced, and for most services, we have clear methods with which to quantify them. A range of methods are also now available to value coastal wetland ecosystems services, but the majority of regional and global syntheses of coastal wetland value still rely on a small number of data points, make various assumptions about data quality and transfer, and assume that the value of coastal wetland ecosystem services is uniform across space. More valuation studies are needed in different coastal settings across the globe to better represent the huge spatial variation inherent in coastal wetland ecosystem service provision and value.
c. *Ecosystem disservices.* Ecosystem disservices have only been conceptualised as an academic research area relatively recently compared to ecosystem services. This is particularly the case for coastal wetlands, where research has been dominated by the ecosystem services paradigm. Ecosystem services have been an established framing for coastal wetlands research for decades, and this review has highlighted that our knowledge of several coastal wetland ecosystem services could be considered to now be quite strong. Ecosystem disserviceshave not received the same amount of attention generally, and this is especially the case for coastal wetlands. The recent introduction of generic ecosystem disservices frameworks (e.g. Vaz et al. 2017) may begin to stimulate ecosystem disservices research in coastal wetlands in the same way that the Millennium Ecosystem Assessment (MEA 2005) did for ecosystem services.
d. *Interactions between ecosystem services and disservices.* Future research needs to close the conceptual and management gaps between ecosystem services and disservices and better integrate them for more holistic coastal management and decision-making. We support recent calls by Blanco et al. (2019) to do this, and there are several steps that can allow this to happen. First, we must acknowledge that coastal wetland disservices can exist in a given management location, and the various disservices that could affect management should be identified. Identified disservices must then be quantified through a variety of techniques. To utilise this information, existing ecosystem services frameworks need to be adapted so that they are more holistic and allow appropriate weighting between services and disservices.

Conclusions

Coastal wetlands have long been considered negatively in history and popular culture, focusing on the perceived and actual ecosystem disservices that they may cause or the economic returns that can be derived by converting these apparent coastal wastelands that have no explicit value themselves. However, coastal communities have also long utilised coastal wetlands for their tangible and intangible ecosystem services, and stakeholders and policy makers are now clearly valuing them for the benefits they provide to coastal societies. The range of ecosystem services provided by coastal wetlands covers broad categories from provisioning to cultural services and can have very high monetary and non-monetary values for coastal communities. As such, coastal wetlands are strongly promoted on the international policy stage for their roles in protecting against natural hazards, sequestering and storing our carbon emissions, and providing goods and materials to support the livelihoods of nearby communities.

As coastal populations continue to increase and human-environment interactions become more common in the coastal zone, there is a need for a more balanced view of coastal ecosystems. This balanced view should take into account the services that coastal wetlands provide alongside the disservices that they cause. Ecosystem disservices have strongly influenced salt marsh and mangrove forest management historically, leading to a view that coastal wetlands have limited value, and incentivising their subsequent large-scale draining. However, it is now important that, in a new age of ecosystem services research, focus doesnot swing too far in the other direction. Ultimately, a binary 'services versus disservices' discourse does few favours for coastal wetland management. Instead, holistic frameworks should embrace and manage the complexity inherent in myriad positive and negative interactions between coastal wetlands and people, in order to find management interventions that encompass the true value of these important coastal systems.

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