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Perspective Article

Opportunities for improving recognition of coastal wetlands in global ecosystem assessment frameworks

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A R T I C L E I N F O

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A B S T R A C T

Vegetated coastal wetlands, including seagrass, saltmarsh and mangroves, are threatened globally, yet the need to avert these losses is poorly recognized in international policy, such as in the Convention on Biological Diversity and the United Nations (UN) Sustainable Development Goals. Identifying the impact of overlooking coastal wetlands in ecosystem assessment frameworks could help prioritize research efforts to fill these gaps. Here, we examine gaps in the recognition of coastal wetlands in globally applicable ecosystem assessments. We address both shortfalls in assessment frameworks when it comes to assessing wetlands, and gaps in data that limit widespread application of assessments. We examine five assessment frameworks that track fisheries, greenhouse gas emissions, ecosystem threats, and ecosystem services. We found that these assessments inform management decisions, but that the functions provided by coastal wetlands are incompletely represented. Most frameworks had sufficient complexity to measure wetland status, but limitations in data meant they were incompletely informed about wetland functions and services. Incomplete representation of coastal wetlands may lead to them being overlooked by research and management. Improving the coverage of coastal wetlands in ecosystem assessments requires improving global scale mapping of wetland trends, developing global-scale indicators of wetland function and synthesis to quantitatively link animal population dynamics to wetland trends. Filling these gaps will help ensure coastal wetland conservation is properly informed to manage them for the outstanding benefits they bring humanity.

1. Introduction

Vegetated coastal wetlands – seagrass, saltmarsh and mangroves – sustain biodiversity, support fisheries production, protect shorelines, mitigate climate change, and improve water quality (Alongi, 2012; Carrasquilla-Henao and Juanes, 2017; Nordlund et al., 2016; Sievers et al., 2019; Silliman et al., 2019). As coastal wetland ecosystems are being lost (Davidson, 2014; Hamilton and Casey, 2016; Waycott et al., 2009) so is their biodiversity, and the goods and services these ecosystems provide (IPBES Secretariat, 2019). Rates of coastal wetland degradation are high in many tropical nations, such as Myanmar, Bangladesh and Indonesia, where mangrove forests are cleared for aquaculture, agriculture and urban development (Hamilton and Casey, 2016; Richards and Friess, 2016). These are nations where the livelihoods of people are most vulnerable to the loss of ecosystem services (Unsworth and Cullen, 2010), and they often benefit from international

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funding for conservation actions. Rates of coastal wetland degradation are also high in some temperate regions, whereas in other areas management has been partially successful at recovering these habitats (de los Santos et al., 2019; Lefcheck et al., 2018). These declines in coastal wetlands are often overlooked in international policy agendas (Cullen-Unsworth and Unsworth, 2018; Duarte et al., 2008; Unsworth et al., 2019).

Conservation priorities for governmental and non-governmental organisations are increasingly driven by generalized ecosystem assessment frameworks, including those recognized by the UN Sustainable Development Goals (SDG) (e.g. Potter Foundation; Global Innovation Fund Hák et al., 2016; Ian Potter Foundation, 2020; Zwane, 2017). SDG Goals 14 (Life below water – Conserve and sustainably use the oceans, seas and marine resources for sustainable development) and 15 (Life on land – Protect, restore and promote sustainable use of terrestrial ecosystems) recognize the benefits of ecosystem services but do not identify coastal wetlands, and no specific targets or indicators have been developed in the Global Indicator Framework (UN, 2017). Similarly, the UN Convention on Biological Diversity's (CBD) Aichi Biodiversity Targets, which are used to prioritise funding, do not explicitly mention coastal wetlands. The challenge of including coastal wetlands in ecosystem assessments stems, in part, from their location at the transition zone between terrestrial and marine ecosystems. Assessment of their status therefore requires considering terrestrial, coastal and marine pressures and processes. There are also large gaps in global scale data for coastal ecosystems, and this is a problem because assessments often rely on global databases to fill gaps in areas lacking local measurements. Relative to other marine habitats, mangrove and saltmarsh ecosystems are monitored by fewer observing programs (Miloslavich et al., 2018), and many of the world's coastal wetlands have never been mapped, or if they have, the data is not widely available (e.g. seagrass, McKenzie et al., 2020).

The gap in recognition of coastal wetlands has been repeatedly documented in earlier studies (Cullen-Unsworth and Unsworth, 2018; Duarte et al., 2008; Unsworth et al., 2019), and the need to enhance their recognition because of their outstanding value to humanity is also well recognized. Commonly cited impediments to recognition of coastal wetlands are that they are not as charismatic as other habitat forming ecosystems and that this has resulted in less public recognition and less scientific interest (Duarte et al., 2008; Unsworth et al., 2018). However, recent strides forward have been made through global scale studies demonstrating the importance of wetland habitats (Adame et al., 2021; Worthington et al., 2020; Zeng et al., 2021). Enhanced recognition seems to be translating into new large-scale conservation initiatives to address wetland declines, such as the Global Mangrove Alliance (Worthington et al., 2020). The path forward for greater recognition of coastal wetlands now requires work to identify the factors that hinder greater recognition of the global scale importance of coastal wetlands.

In this article, we ask whether and how coastal wetland ecosystems are included in prominent globally applicable ecosystem assessments. For the purposes of this article we define ecosystem assessments as conceptual or mathematical frameworks for assessing the status of ecosystems relative to reference conditions. Assessments are made with indicators and possibly indices, where indices are multivariate combinations of indicators (sometimes termed 'ecosystem health'), though we acknowledge there are multiple definitions of indicators and indices in the literature. We first elaborate on the role of assessments in guiding continental and global scale conservation actions. We then highlight several case-studies of how assessments have dealt with coastal wetlands. We identify where there are gaps that relate to: (1) shortcomings in assessment processes when applied to coastal wetlands, and (2) insufficient global scale indicators and indices. We also consider whether those gaps apply to all, or just some of the wetland types. We then discuss the impact that the inclusion or exclusion of coastal wetlands has on potential conservation outcomes. Finally, we highlight opportunities for improving coastal wetland representation in assessments and the likely benefits for these threatened ecosystems. Ultimately, this article provides the global conservation science community, non-governmental organisations (NGOs), practitioners and government officials an overview of the importance of recognising coastal wetlands in global assessments and conservation initiatives.

2. How global assessment frameworks guide scientific research and conservation action

We propose that ecosystem assessments exist in a life-cycle that couples scientific development of the assessment with policy decisions and conservation actions (Fig. 1). Starting at the top left of Fig. 1, indicators help to identify knowledge gaps, which motivate science to fill those gaps (e.g. Halpern et al., 2017). This can then lead to both iterative improvement of indicators, but also assessments of ecosystem health (bottom loop in Fig. 1). Assessments guide action to improve the status of ecosystems and if those actions work then assessment outcomes should eventually improve (e.g. McQuatters-Gollop et al., 2019; Orth et al., 2017). As a result of these feedbacks, environmental outcomes are related to assessment processes and data gaps affecting those assessments.

Policy-makers need assessments of the natural world to inform the effect of past decisions and to motivate new actions. Poor assessment outcomes can prompt governments and managers to conduct research that informs improved management actions (Fig. 1) (McQuatters-Gollop et al., 2019). Examples of policy motivating development of assessments are the European Union's Water Framework Directive (WFD) and Marine Strategy Framework Directive (MSFD), which assesses ecosystem integrity relative to reference conditions. The WFD specifies broad guidelines for what should be assessed, but not specific indicators (Birk et al., 2012). Almost 300 indicators were created by 2012, including 134 indicators for coastal and transitional waters (Birk et al., 2012) and indices specific to marine angiosperms (e.g. seagrass; Neto et al., 2013). Both frameworks have been implemented at the European Union level, meaning a framework that is applicable for coastal wetlands at a global scale is still missing. The MSFD applies adaptive management strategies, meaning changes in indices feed back to affect management priorities (EU, 2008). The WFD and MSFD guided the creation of new research programs that will support monitoring and index calibration and are funded by the European Commission (e.g. FP7, H2020, Horizon Europe; the latter to be launched). At the global scale, impetus to report on the UN Convention on Biological Diversity's (CBD) Aichi Biodiversity Targets has motivated science to develop numerous biodiversity indicators that are now used within assessments of ecosystem status and trends (Butchart et al., 2010; McQuatters-Gollop et al., 2019).

Analyses of global scale priorities for funding are informed by assessments of indicators and indices (Waldron et al., 2013). Global indicators and indices are necessary for measuring progress towards internationally agreed conservation goals, for holding nations accountable for their actions and for evaluating the effectiveness of past policies (e.g. Butchart et al., 2010; Rowland et al., 2019). Notably, the Aichi Biodiversity Targets do not name specific indicators that should be used to assess progress on these targets (Leadley et al., 2014). Many of the targets have not been assessed due to lack of monitoring of conservation actions, and those that have been assessed suggest the targets are not being met (IPBES Secretariat, 2019; Mcowen et al., 2016; Tittensor et al., 2014). Evident shortcomings in indices and conservation progress have become instrumental in debates over goals and indicators for the Post-2020 Global Biodiversity Framework and the European Green Deal, including the EU Biodiversity Strategy for 2030 which aims to implement the SDGs.

In contrast to the global frameworks where monitoring gaps hinder assessments, local examples highlight the power of indices as tools for communication. The score card for Chesapeake Bay water quality and submerged aquatic vegetation, for instance, synthesises multiple indicators in a way that is accessible to policy makers and the general

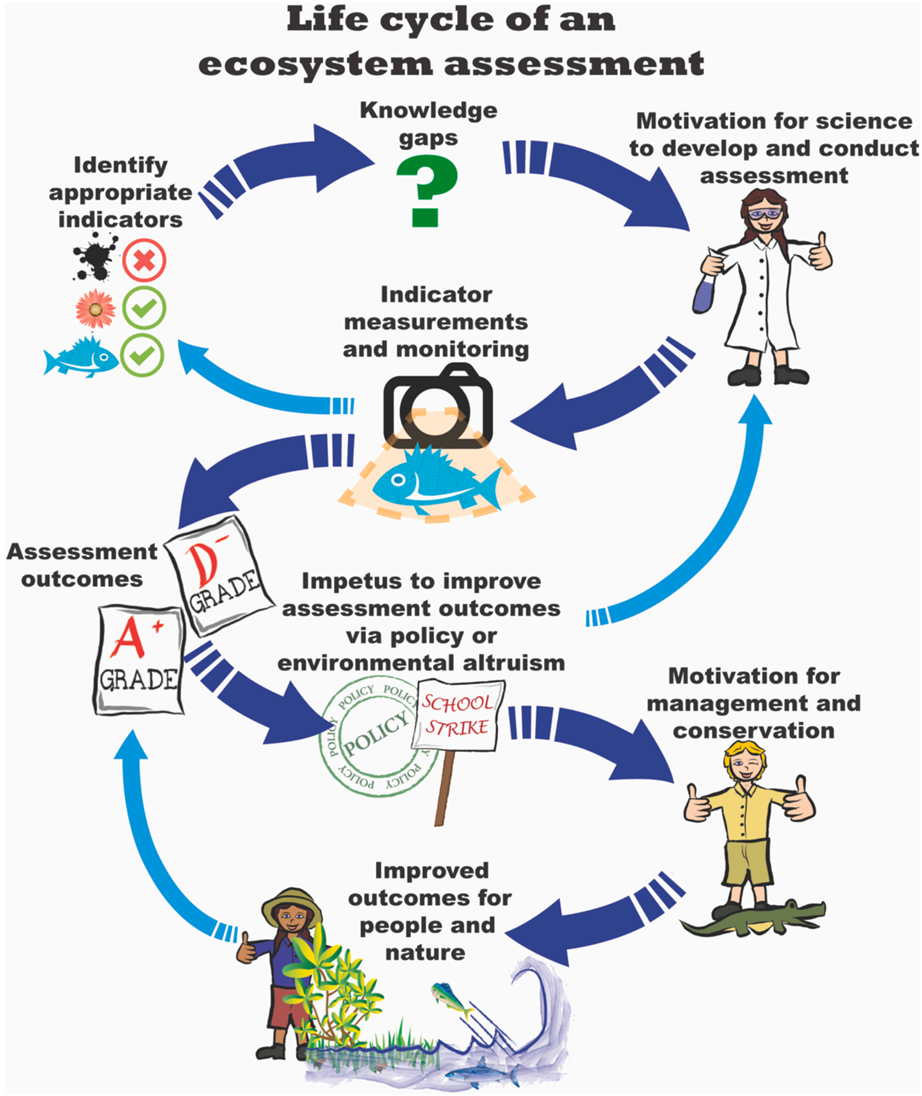

Fig. 1. Idealized life cycle of an ecosystem assessment, emphasizing the connection between ecosystem health assessments and funding priorities. Thick arrows represent major pathways from conception to on-the-ground outcomes and thin arrows represent feedback loops that may optimise the index and overall outcomes. There are many potential starting points on this graph, hence it is titled a 'life-cycle'.

public. As such, the score card has been important for raising public awareness about water quality issues and creating the impetus for

Table 1
The five assessment frameworks and their key strengths and gaps when applied to coastal wetlands.

Assessment framework	Objective	Key strengths	Challenges
Assessments of fish stock status	To inform managing fisheries for productive and long-term sustainability of fish populations and fisheries	- Has clear reference points for management action - Emphasises the role of managing fisheries in improving stock productivity	- Change in coastal nursery habitats can have complex effects on stock status - Limited data to link habitat change to stock status
IUCN Red List of Ecosystems	Assess the risk of ecosystem collapse	The framework is holistic in its consideration of multiple pressures, ecological interactions and foundational habitats	- Preferable to have long-term (>50 years) data on ecosystems. - Not clear how to incorporate cross-ecosystem connections
Nationally determined contributions	Commit to actions that reduce greenhouse gas emissions	Carbon storage in coastal wetlands is accounted for by many nations	The definition of 'forest' used by some nations may exclude many mangrove forests
Ocean Health Index (OHI)	Quantify the health of ocean ecosystem services	Comprehensive consideration of ecosystems and ecosystem services	- Misses some ecological interactions wetlands support (fish nurseries) - Current assessments mostly dependent on habitat area data and lack data on ecosystem structure and function
System of Environmental-Economic Accounts	Track the value of ecosystem services	Systematic framework for reporting on the value of ecosystem services in coastal wetlands	- Same as for OHI - Quantifying the monetary value of natural assets

management to improve water quality (Orth et al., 2017).

Ecosystem types that are not fully represented by global assessments may be undervalued by policy-makers (Nicholson et al., 2019). Incomplete representation of an ecosystem means it may receive less management and scientific attention than its true status warrants. For example, many conservation funds use lists of threatened species to prioritise projects, including the IUCN Save Our Species grants (https://www.saveourspecies.org/), Australia's Threatened Species Recovery Fund (https://www.environment.gov.au/biodiversity/threatened/threatened-species-recovery-fund) and the Mohammed bin Zayed Species Conservation Fund (https://www.speciesconservation.org/). Support for actions to avert declines in coastal wetlands depends, therefore, on how coastal wetlands are represented in ecosystem assessments.

The large gaps in monitoring of coastal wetlands, combined with the challenge of monitoring ecosystems that are highly interdependent, suggests that incorporating them into indices for ecosystem assessments will be challenging. To identify how ecosystem representation in indices affects conservation priorities, we now consider five existing assessments as global-scale case-studies (Table 1). We chose the five case studies to represent assessments that: (1) represent some of the diversity of objectives in conservation science, ranging from biodiversity focused to human-centric and monetary outcomes (Evans, 2021) (2) have evidence of impact on policy. Three of the assessments are specific to particular outcomes (fisheries, ecosystem collapse and decreased greenhouse gas emissions) and two (the Ocean Health Index and the System of Environmental-Economic Accounts) are hybrids in that they address multiple different types of goals. Finally, we picked these five frameworks because all are well established in the international literature and are the subject of significant ongoing scientific work to improve their representation.

3. Case-studies for how representation of coastal wetlands in ecosystem assessments affects conservation actions

3.1. Assessments of fish stock status

Decisions by fishery managers to permit increased catches, or to reduce catch and rebuild stock biomass are informed by indicators of a stock's status. Stock status is also an indicator for the Sustainable Development Goal 14.4 that aims to restore overexploited fish stocks to productive levels (Food and Agriculture Organisation – FAO, 2018). Stock status therefore has a central role in informing advice on global scale fishery policies (FAO, 2018). A fishery's status is determined by classifying stock biomass and exploitation rate against reference points for safe limits (Mace, 2001). Reference points are commonly set relative to the stock biomass and exploitation rate that maximize long-term catches, while also considering economic productivity and ecological sustainability. Globally, 33.1% of fish stocks are estimated to be fished at biologically unsustainable levels, whereas 59.9% are fished at maximal sustainable levels (FAO, 2018).

The assessment of fishery status against reference points based on fishing exploitation puts the emphasis on the management of fisheries as the solution to poor stock status. But habitat change also determines stock status indirectly through effects on fish life-histories. Almost 25% of stocks in the key global assessment database (Ransom Myers Legacy database) are associated with threatened coastal wetland habitats (especially seagrasses) (Brown et al., 2019). The global stock database is biased towards temperate regions, with relative paucity in tropical regions. If tropical regions were included then significantly more stocks would also be associated with mangrove habitats (Sheaves, 2017). In any respect, global degradation of wetland habitats may explain why some stocks are not recovering, despite reduced fishing pressure (Brown et al., 2019). The exclusion of fishery-habitat linkages in the assessment of fishery status may mean opportunities are missed to use habitat restoration and protection to rebuild fisheries. For example, management recommendations for improving the status of fisheries, and thus achieving SDG 14, do not acknowledge the widespread contribution that habitat restoration could play in supporting productive fisheries (FAO, 2018). Thus, the global policy agenda for recovering productive fisheries continues to overlook the habitat role of coastal wetlands.

The exclusion of fish-habitat linkages likely stems from both insufficient understanding of fish-habitat linkages and insufficient global data. We lack a sufficiently complete synthesis of how the population dynamics of fishery species respond to habitat loss (Brown et al., 2019). Habitat responses of fish species can be highly species specific and depend on multiple contextual factors, including life-history and habitat arrangement (Nagelkerken et al., 2015). To date only mangroves have a global synthesis of dependent fishery catch (Table 2), we lack truly global assessments of habitat presence and areal trends for saltmarsh and seagrass. Further, across ecosystems, the best global scale data for habitat trends is for area of habitat (Table 2), but seascape connectivity and the intertidal zone of mangrove forests are more important for fish productivity than a forest's overall area (Nagelkerken et al., 2015).

3.2. Assessing ecosystem collapse with the IUCN Red List of Ecosystems

The International Union for the Conservation of Nature (IUCN) recently developed the Red List of Ecosystems (RLE) to assess the risk of ecosystem collapse (Keith et al., 2013). It defines criteria for ecosystem collapse based on changes in the distribution and extent of the ecosystem, and changes in environmental and biotic components important for ecosystem persistence. The RLE posits that "an ecosystem is collapsed when all occurrences lose defining biotic or abiotic features, no longer sustain the characteristic native biota, and have moved outside their natural range of spatial and temporal variability in composition, structure and/or function" (Bland et al., 2017). The framework has been applied to more than 2500 cases of ecosystems across 100 countries, with clear evidence of management and conservation outcomes (Bland et al., 2019). The RLE framework's consideration of ecological interactions, including persistence of plant species as well as their foundational role as habitat for other species, and integration across multiple pressures, means the framework has better coverage of the conservation issues facing wetlands than other reductionist indicators.

Previously, the RLE framework encouraged the assessment of ecosystem types in isolation from each other. But simultaneous assessments of multiple ecosystem types can begin to account for the connections between coastal wetlands (Sievers et al., 2020b), such as the movement of fish between mangrove and seagrass habitats (Olds et al., 2016). More recently, the development of a new global ecosystem typology (Keith et al., 2020) allows RLE assessments to more easily be conducted across higher hierarchical levels than ecosystems delineated by the habitat forming species (e.g. mangrove forests). The typology 'permanently open riverine estuaries and bays', for example, explicitly includes areas of mangroves, saltmarshes and seagrasses, and thus incorporates aspects of connectivity and influences outside of immediate management control. There is a need now to test the hierarchical application of typologies to coastal wetlands to ensure it can capture ecosystem connections and their influence on risk of ecosystem collapse.

Although the RLE assessment methodology is amenable to the requirements of coastal wetlands, accurate assessments need much data. Most criteria that contribute to assessing risk of collapse are based on temporal trends in indicators, with data series of 50 years being preferred, although shorter timeframes can be extrapolated. As for fisheries stock assessments, the substantial lack of data on saltmarsh and seagrass presence and trends prevents globally comprehensive assessments. The advent of several global mangrove datasets has provided important information for region-specific assessments (e.g. Sievers et al., 2020a) and may facilitate assessments at larger scales. As products similar to those for mangroves become available for saltmarsh and seagrass, global assessments can begin to be produced, supplementing local-scale assessments and supporting global conservation initiatives.

Table 2
Examples of global scale datasets that have been used within the assessment frameworks we discuss.

Data type	Wetland ecosystems	Strengths	Caveats	Source
Cumulative human impacts to the world's oceans	All	Globally comprehensive, multivariate, time series allows evaluation of trends.	Poor overlap with coastal wetlands (e.g. 88% of mapped mangrove habitats do not overlap with these layers).	Halpern et al. (2019)
Human footprint	Non-specific	Globally comprehensive, multivariate, time series allows evaluation of trends.	Terrestrial focus, so not directly relevant to sub-tidal wetlands.	Venter et al. (2016)
Habitat extent	Mangrove	Globally comprehensive, multiple sources created with different methods enables comparisons.	Discrepancies between different data sources, see Worthington et al. (2020)	Bunting et al. (2018); Hamilton and Casey (2016); Giri et al. (2011)
	Seagrass	Global observations, combine different data sources.	Many observations are points, many polygons are incorrect McKenzie et al. (2020), large geographic gaps, no time series available to evaluate trends.	Assis et al. (2020); Short (2018)
	Seagrass	Most comprehensive global scale mapping, includes an assessment of data quality.	Not open access, geographic gaps, no time series available to evaluate trends.	McKenzie et al. (2020)
	Saltmarsh	Most complete synthesis of saltmarsh extent to date, high data quality for important saltmarsh regions of Europe, USA and Australia.	No time series available to evaluate trends, many observations are points only, not globally comprehensive, large geographic gaps in many countries.	McOwen et al. (2017)
Trends	Mangrove	Globally comprehensive and possible to compare trends across different methods.	Discrepancies between different data sources.	Multiple e.g. Bunting et al. (2018); Hamilton and Casey (2016);
	Seagrass	Most complete global trends data available.	Not recently updated, geographic bias, linear rates of change miss high inter-annual variability of seagrass meadows.	Waycott et al. (2009)
	Saltmarsh			None yet available
Ecosystem services and functions	Mangrove soil carbon	High-spatial resolution, globally accessible.	Overestimates low soil carbon and underestimates high soil carbon locations.	Sanderman et al. (2018)
	Mangrove above ground biomass	High-spatial resolution, globally accessible.	High uncertainty at small scales (30 m^2) and may mis-represent some mangrove forests on Pacific Islands.	Simard et al. (2019)
	Mangrove tourism	Globally comprehensive.	Only sites that are advertised in specific online searching tools.	Spalding and Parrett (2019)
	Mangrove fisheries catch	Global scale meta-analysis for fish and invertebrate catch (and density).	Geographic gaps would have to be filled through model predictions.	zu Ermgassen et al. (2020)
	Threatened species	Synthesis of expert assessments from 100 s of experts across 100 s of species globally.	Many species associated with coastal wetlands are not recognized as such in assessments (Sievers et al., 2019).	IUCN (2020)
	Coastal protection (mangroves)	Globally comprehensive for larger mangrove patches, model considers local processes affecting protection benefits.	Missing areas of mangroves (e.g. southern Australia), validation of the storm model only in the Philippines.	Menéndez et al. (2020)

3.3. Nationally determined contributions on greenhouse gas emissions

The Paris Climate Agreement includes contributions from each member country to reduce their greenhouse gas emissions (United Nations Framework Convention on Climate Change, 2020). The Nationally Determined Contributions (NDC) can include protection, restoration and improved management of forests as actions for reducing emissions. The definition of 'forest' has important implications for whether conservation of mangroves can be counted towards the NDCs. For the purposes of the now lapsed Kyoto Protocol under the UNFCCC, forest is defined as a minimum area of land of 0.05–1.0 ha with tree crown cover of more than 10–30 percent with the potential to reach a minimum height of 2–5 m at maturity in situ (UNFCCC, 2001). In arid regions or in poor-nutrient soils, mangroves are dominated by short trees <2 m in height, excluding them from National Forest Inventories and Global Forest Assessments (FAO, 2015). Nearby, the same mangrove species could form forests of trees as high as 40 m and would be included (Adame et al., 2018b). In Mexico, many mangrove areas in the country are short-statured and were not included in the National Forest Inventory. The exclusion of short-statured forests contributed to Mexico under-estimating emissions from mangrove deforestation by 31 times in their NDC (Adame et al., 2018a).

Recently, some countries are reporting coastal wetlands in their National Greenhouse Gas Inventories (Lovelock et al., 2018). In 2016, at least 28 countries' NDCs included a reference to coastal wetlands in terms of climate mitigation and adaptation (Herr and Landis, 2016). The inclusion of wetlands can have top-down effects on ground activities for wetland conservation and restoration. For example, in Australia, all coastal blue carbon ecosystems (mangroves, seagrass and tidal marshes) are now recognised through the International Partnership for Blue Carbon and the Ramsar Convention as important to the climate change agenda ("Conservation, restoration and sustainable management of coastal blue carbon ecosystems", 13th COP, 2018). As a result, funding for coastal wetland restoration has increased, for instance, through the Land Restoration Fund for carbon farming (LRF, 2019) and Australia's Climate Solutions Fund to reduce carbon emissions (Department of Agriculture, Water and the Environment 2019). In Africa, work done decades ago to establish improvement in ecological indicators following mangrove restoration (e.g. Kairo et al., 2001) has facilitated the Abidjan and Nairobi Conventions that recognize coastal wetlands and provide financial and practical mechanisms for funding conservation (Feka and Morrison, 2017; Maina et al., 2020).

3.4. Ocean health index

The Ocean Health Index (OHI) is a global index that quantifies ocean health based on the potential benefits derived from ecosystem services that healthy systems provide (Halpern et al., 2012). The OHI provides annual health scores (since 2012) at the national level for all countries with coastlines, allowing trends to be assessed and compared with changes in on-the-ground drivers and pressures. The focus on ecosystem services as the definition of health for OHI puts the management focus on potential changes that may directly benefit biodiversity as well as humans and society. In this respect, the OHI has great potential to trigger valuable legislative changes that may conserve or improve outcomes for coastal environments. For example, OHI has been integrated into environmental plans for multiple jurisdictions as they track progress towards SDG 14 ("Life below water") (Halpern et al., 2017).

The OHI assessment process does allow for multiple contributions of coastal wetlands towards its goals, including habitat condition, carbon storage and coastal protection (Halpern et al., 2017). Important gaps for coastal wetlands are in the fisheries and biodiversity goals, because the OHI assessment process does not recognize the role of wetland habitats in supporting animal populations. OHI assessments of coastal wetlands are also affected by data gaps. For example, OHI scores are linked to IUCN Red List of Threatened Species™ assessments (Halpern et al., 2017), so they perpetuate the same data gaps (such as coastal wetland dependence) that afflict IUCN Red List assessments (Table 2). Further, habitat condition indicators are primarily based on trends in habitat area and so may misrepresent, for instance, animal-habitat relationships that depend on structural complexity. These shortcomings in the OHI reflect gaps in global datasets for habitat degradation and gaps in understanding of links between habitat change and animal population dynamics. The precision of the OHI at capturing changes in wetlands could therefore be improved in the future through greater access to indicators for change in wetland ecosystem functions, such as quantifying how the productivity of fisheries species relates to wetland habitat condition. Adding condition would better capture the full contribution of coastal wetlands to ecosystem services.

3.5. System of economic environmental accounts

A System of Environmental-Economic Accounts (SEEA EEA) is used by governments to keep track of the provision of ecosystem services. Thus, the SEEA is a form of ecosystem assessment that seeks to quantify indices of environmental condition and ecosystem service value. At the highest level, they intend to be an environmental complement to economic indices like the Gross Domestic Product, which are used worldwide to track policy outcomes and inform on future policy actions. The UN Statistical Commission has endorsed a standardised framework called the System of Environmental-Economic Accounts – Experimental Ecosystem Accounting (SEEA EEA). The SEEA-EEA standardises environmental accounting across nations and ecosystems. These accounts are intended to become long-term records that inform decision making on environmental issues (Obst and Vardon, 2014). A SEEA-EEA case study in Guatemala, for example, identified agriculture, urban development, and timber harvesting as key drivers of forest loss over a 60-year period and that 95% of the commercial logging was illegal, ultimately resulting in policy reform including the approval of a new Forest Incentive law (Banerjee et al., 2016).

The SEEA EEA assessment framework expands definition of a region's production (i.e. human benefits) to include ecosystem services. SEEA EEA thus includes the diverse ecosystem service values provided by coastal wetlands, including biodiversity, coastal protection, fisheries production, water filtration and climate regulation through carbon storage. However, actual assessments will likely suffer from the same data gaps that inflict the OHI. For example, SEEA is like the OHI in that it proposes to use IUCN Red List of Threatned Species™ as one of its indicators (Bogaart et al., 2019), but has gaps for wetland associated species (Sievers et al., 2019). Further, both the OHI and SEEA are challenged to account for the role of wetlands in supporting fishery catch, for the same reasons that fish stock assessments do not consider this link (Table 2). Since SEEA (like GDP), are used as indices of economic performance, gaps in the processes they represent could bias the importance of coastal wetlands in policy debates.

Another challenge is quantifying the monetary value of ecosystem assets. This is because ecosystem goods and services are generally not transacted on the market and welfare valuation methods are not appropriate for accounting the value of natural assets (Obst and Eigenraam, 2017). SEEA EEA's recommendation is to approximate the value of an ecosystem through the present value of expected future profits (resource rent) or calculating the net present value (NPV) (Edens and Hein, 2013). However, the challenge lies in quantifying a relevant cost share to all the ecosystem assets involved in a producing single ecosystem service or good (Obst and Eigenraam, 2017).

4. Opportunities for improving recognition of coastal wetlands in global ecosystem assessments

Based on our case-studies, we identify opportunities to fill the gap in recognition of coastal wetlands in globally applicable ecosystem assessments. These relate to gaps in the assessment process and gaps in data availability.

4.1. Gaps in the assessment process

The primary gap in the assessment process that we identified was lack of the consideration of animal-habitat linkages. The consideration of animal-habitat linkages would improve fish stock assessments and ecosystem services assessments, which commonly use the IUCN Red List of Threatened Species™ as an indicator. Global scale synthesis of linkages exists (Brown et al., 2019; Sievers et al., 2019), but they do not provide quantitative synthesis of functional relationships between population parameters and wetland indicators. Quantitative synthesis is needed to inform how assessment outcomes should change in response to wetland degradation (Brown et al., 2019; Sievers et al., 2019).

Quantitative indicators linking habitat and animal function would put greater emphasis on the role of coastal wetland conservation in managing fisheries and preventing species extinctions. Making the link between fisheries catch and fish population habitats at a global scale is possible now for mangroves, because of global meta-analysis of mangrove dependent fish catch (Table 2, Fig. 2). Global assessments of fish catch dependence on saltmarsh and seagrass are held back by incomplete maps of seagrass distribution and trends, and lack of global synthesis linking fishery production to these wetlands (Fig. 2). Synthesis of foodweb and survival studies will help fill this gap (Jänes et al., 2020; Lefcheck et al., 2019), particularly when combined with habitat maps to produce global summaries of catch dependence (Fig. 2).

4.2. Gaps in data availability

For all five case-studies we found lack of suitable data was creating gaps in the ability of assessments to represent coastal wetlands. Data gaps are a common issue for assessments, but are not insurmountable. Coastal wetland extent crosses both land and sea, but many indicators are developed for either land or sea, and are not intended to cross the coastal transition, so there is a gap at the coastal interface. The coastal gap, also termed the 'white ribbon' may often stem either from gaps in even the most basic geographic data, which require specialist techniques to fill (e.g. Leon et al., 2013). For example, the global threat maps that support the OHI assess only 'marine' habitats, so they miss much mangrove and saltmarsh habitat (e.g. 88.4% of the mangroves in Bunting et al. (2018) do not overlap with the threat maps in Halpern et al. (2017)). Important threats to coastal wetlands like aquaculture are also not mapped at a global scale, which has contributed to misconceptions about the human causes of wetland loss (Hamilton, 2015). Regional scale indices, such as in Europe (Neto et al., 2013) and the US (Orth et al., 2017) have met the challenge of monitoring the coastal transition. We now need to scale-up these approaches to global scales.

Recent progress in synthesizing global data layers for mangroves (Worthington et al., 2020) provides hope that similar rapid advances could be made for seagrass and saltmarsh. Mangrove scientists have applied advances in analysis of remotely sensing datasets and analysis techniques to produce globally comprehensive maps of mangrove area, rates of loss, forest condition and ecosystem services (Table 2).

Global datasets for seagrass and saltmarsh lag behind those available for mangroves (Table 2). There are global databases for the extent of seagrass (McKenzie et al., 2020; Short, 2017) and saltmarsh (McOwen

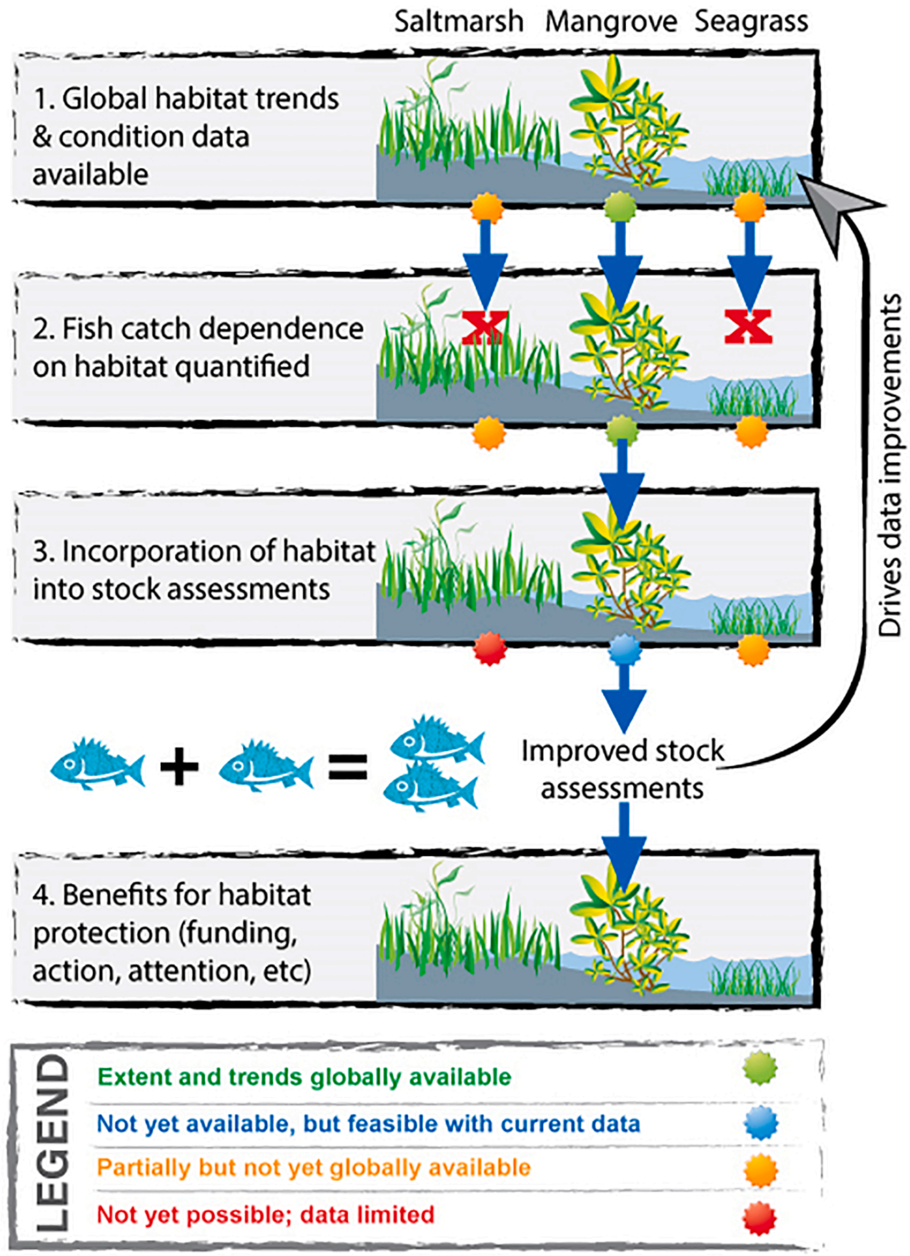

Fig. 2. Example of knowledge frontiers for linking coastal wetland ecosystem extent and condition indicators for analysis of fishery productivity fish stock assessments, given current global datasets and syntheses for saltmarsh, mangrove and seagrass habitats.

et al., 2017), but these maps have large geographic gaps. Data on habitat trends for saltmarsh and seagrass are biased towards a few well monitored regions, like the USA (Halpern et al., 2012) and Europe (de los Santos et al., 2019). The gap in saltmarsh and intertidal seagrass trends would be feasible to fill by applying new remote sensing techniques for analysing intertidal areas (e.g. Murray et al., 2019). The gap in subtidal seagrass trends may be impossible to fill completely, but past global scale syntheses could be updated (Waycott et al., 2009). A nascent global seagrass working group can hopefully fill these data gaps for seagrass (Duffy and Weatherdon, 2020), and we propose a similar effort is needed for saltmarsh habitats. These global syntheses then need to be communicated and distributed to decision-making and funders. For instance, the Global Mangrove Alliance, which brings together researchers, global conservation agencies, policy-makers and practitioners, has supported networks and data portals that are facilitating the use of big data for mangroves conservation (Worthington et al., 2020).

A key gap for all wetlands is that many indicators are derived from habitat extent data, yet extent is only weakly correlated to ecosystem functions and ecosystem services (Lee et al., 2019). Indicators of ecosystem function and services could be more accurate if they were informed by datasets of ecological condition (Lee et al., 2019), such as important indicators of eutrophication like plant nitrogen content (Wan et al., 2020). The relative ease of monitoring mangroves has meant there are now global scale indicators for tree biomass, height and soil carbon, and these indicators will improve carbon stock estimates (Worthington et al., 2020). The analogous indicator in seagrass, density, has been mapped at large scales (Europe) (de los Santos et al., 2019), but not yet globally. Further work is needed to develop global scale condition indicators and indices for saltmarsh and seagrass. Ideally, these could be created in conjunction with animal habitat models, to ensure that mapping captures the conditions that are relevant to animal population dynamics, such as seascape connectivity (e.g. Nagelkerken et al., 2015).

Many indicators overlook the role that connected coastal wetland habitats play in supporting biodiversity and ecosystem services (e.g.

Sievers et al. 2019). Connectivity is an important driver of ecological functions and services, and is thus an important indicator to consider in ecosystem assessments. Given the highly connected nature of coastal wetlands (Olds et al., 2016), future health assessments need ways to incorporate connectivity. Incorporating connectivity will be challenging because there is no agreement on globally-relevant metrics to quantify connectivity. Syntheses of coastal animal movement patterns could help inform new indicators of connectivity, as past syntheses have done for informing on marine reserve design on coral reefs (Green et al., 2015). Global maps of habitat fragmentation partly address this problem (e.g. Bryan-Brown et al., 2020), but the impact of fragmentation on biodiversity and ecosystem services likely varies region to region (Boström et al., 2006). Further work is needed to map fragmentation or connectivity of saltmarsh and seagrass habitats, and to link changes in connectivity to ecological outcomes.

We propose that in addition to filling specific data gaps, we also need to recreate existing global data layers with different techniques and source data. Most global indices reuse the same data layers, such as data on global mangrove trends (Hamilton and Casey, 2016) and a synthesis of seagrass trends (Waycott et al., 2009). If more diverse data-sources were available, global indices would have different strengths and weaknesses and therefore be complementary (e.g. Table 2).

Continuous funding for long-term ocean data observations is needed to support high quality data inputs. Government funding is often compromised during financial crises so it is important that scientists continue to fight for the continuation of these programs (Edwards et al., 2010). Therefore, opportunities for raising funding are necessary. However, continuous funding alone will not solve everything. We need to have institutions and adequate management to make the funding effective into appropriate conservation actions, also aligned with international policies and targets.

All of the above opportunities would be supported by comprehensive mapping and identification of existing coastal wetlands indicators and continuing to push for open access data (McKenzie et al., 2020; Worthington et al., 2020). A synthesis of existing indicators for threats, functions, services, extent and policy protection would aid in utilizing these indicators, or a mix of them, to fill gaps in existing indices and ecosystem assessments. Sufficient data are now available to initiate such a synthesis. For example, global data on habitat distribution (Bunting et al., 2018), pressures (Halpern et al., 2019), and governance and policy effectiveness (info.worldbank.org/governance/wgi/) are all freely available. These indicators could also be combined into an index tailored to suit coastal wetlands, which would ultimately facilitate communication among all types of stakeholder, and allow direct comparisons among management objectives in these habitats (McQuatters-Gollop et al., 2019).

5. Conclusions

We found the recognition of coastal wetlands in global scale indicators and ecosystem assessments is patchy. While we only considered five case-studies for assessments, the case-studies represent a broad range of assessment types, from biodiversity to economically focused. Further, many of these assessments rely on the same underlying data, so it is likely that other global assessments face similar gaps. Future priorities for research are to expand global habitat trends data for seagrass and saltmarsh, develop indicators of ecological condition and quantify habitat-animal linkages. Filling these gaps will help track the success of policy changes, keep nations accountable for declines in coastal wetland status and motivate conservation action for these ecosystems.

CRediT authorship contribution statement

Christopher J. Brown: Conceptualization, Writing - original draft, Writing - review & editing, Supervision, Funding acquisition. **Maria F. Adame:** Writing - original draft, Writing - review & editing. **Christina A. Buelow:** Writing - original draft, Writing - review & editing. **Marieke A. Frassl:** Writing - original draft, Writing - review & editing. **Shing Yip Lee:** Writing - review & editing, Funding acquisition. **Brendan Mackey:** Writing - review & editing, Funding acquisition. **Eva C. McClure:** Writing - original draft, Writing - review & editing. **Ryan M. Pearson:** Writing - original draft, Writing - review & editing, Visualization. **Anusha Rajkaran:** Writing - review & editing, Funding acquisition. **Thomas S. Rayner:** Writing - original draft, Writing - review & editing. **Michael Sievers:** Writing - original draft, Writing - review & editing. **Chantal A. Saint Ange:** Writing - original draft, Writing - review & editing. **Ana I. Sousa:** Writing - review & editing, Funding acquisition. **Vivitskaia J.D. Tulloch:** Writing - original draft, Writing - review & editing. **Mischa P. Turschwell:** Writing - original draft, Writing - review & editing. **Rod M. Connolly:** Conceptualization, Writing - original draft, Writing - review & editing, Supervision, Funding acquisition.

Declaration of Competing Interest

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

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