Ecosystem Service Valuation of Blue Carbon Habitats: A Review for Saltmarshes and Seagrasses

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1. INTRODUCTION

In the past 150 years, atmospheric CO$_2$ concentrations have risen to levels that have not been experienced in at least 2 million years (Arias et al. 2021). Along with increasing concentrations of other greenhouse gases, this has led to significant increases in global mean temperatures (Dusenge et al. 2019). This human-induced climate change has been observed to cause widespread loss and damage to nature and people (Arias et al. 2021). Adverse socio-economic impacts are broad ranging and include reduced agricultural yields and economic production, increased risk of violent conflict and crime, and higher mortality rates (Carleton and Hsiang 2016). Recognition of the risks posed by climate change is reflected by the Paris Climate Agreement commitment to hold “the increase in the global average temperature to well below 2°C above pre-industrial levels” (UNFCCC 2015). This requires not only reducing emissions from human activities, but also large-scale CO$_2$ removal methods (Smith et al. 2016).

Improved land and ocean stewardship is the most mature and relatively low cost means of CO$_2$ removal, and provides co-benefits of improving habitat quality and biodiversity (Field and Mach 2017). These methods include reforestation, afforestation, and coastal restoration. Marine and coastal restoration is less developed relative to terrestrial restoration (Saunders et al. 2020), yet oceans and coastal marine ecosystems play a significant role in the global carbon cycle as a long-term carbon sink. Vegetated coastal habitats such as seagrass, saltmarsh and mangroves are estimated to account for 50% of carbon storage in marine sediments, despite occupying less than 2% of the ocean (Duarte et al. 2005). However, these ecosystems are being degraded 5-10 times faster than rainforests (Nellemann and Corcoran 2009). This not only compromises their ability to act as carbon sinks but can result in carbon being released back into the atmosphere (Cott et al. 2021).

“Blue carbon” is a term that was introduced by the United Nations Environment Programme (UNEP) and the International Union for Conservation of Nature (IUCN) in 2009 to describe the carbon stored in these ecosystems (Nellemann and Corcoran 2009). The specific definition of “blue carbon ecosystems” refers to mangrove forests, saltmarsh, seagrass meadows and tidal freshwater forests (Blue Carbon Initiative 2015). International policy mechanisms such as the Convention on Biological Diversity (Aichi Target 11 and 15); the UN Sustainable Development
Goals (SDGs 13 and 14); the European Green Deal and the EU Biodiversity Strategy 2030 all increase pressure to appropriately manage these ecosystems.

Blue carbon habitats are not only valuable for their carbon storage capabilities, but also for the range of ecosystem services they provide, including coastal protection and improving coastal water quality. However, restoring these ecosystems can be costly and will entail trade-offs with economic activity. To make effective and appropriate management decisions, these costs must be compared with the benefits provided by these ecosystems. Ecosystem services provided by blue carbon ecosystems (BCEs) should therefore be defined and quantified in order to understand their socio-economic value and to inform policy decisions in relation to carbon emission reduction policies and environmental conservation and restoration efforts. The BlueC project, supported by the Marine Institute and the Environmental Protection Agency of Ireland, specifically aims to advance understanding of two particular BCEs; saltmarsh and seagrass. With that in mind this research carries out a global review of the literature of ecosystem service valuation related to saltmarsh and seagrass habitats.

1.1 Ecosystem Service Valuation

The modern use of the term ecosystem services (ES) began in the 1970s (Gómez-Baggethun et al. 2010), and was originally intended as a metaphor to communicate the benefits that humans receive from nature (Norgaard 2010). What we now refer to as ES have always existed, the term is simply a new way to describe the relationship between humans and nature. The concept gained recognition at the turn of the century with the publication of Costanza et al.’s global valuation of the world’s ecosystems and natural capital (1997), and the Millennium Ecosystem Assessment (MEA) (MEA 2005). Since then, the ES literature has grown exponentially (Fisher et al. 2009). In the ES framework, the ecosystem is conceptualized as a fund or a stock of natural capital from which a range of services are drawn (Bateman et al. 2011; Farley 2012). It is often depicted as a cascade, and the services link the environmental system to the social and economic system (Haines-Young and Potschin 2010) (Figure 1). They are commonly split into four branches, called provisioning, regulating, cultural (or information) and supporting services (de Groot et al. 2002; MEA 2005). Provisioning services are often traded in markets, examples include timber and fish that are harvested from the forests and
seas. Regulating global temperatures through carbon sequestration or providing flood defense by slowing water flows are two examples of regulating services, which refer to the ways in which ecosystems moderate or mediate the environment that benefit humans. Cultural services are outputs that allow us to have a relationship with nature in recreational, spiritual, aesthetic or educational ways. Finally, supporting services are the processes that underpin the other services. Primary production and soil formation are examples. Provisioning, regulating and cultural services are known as final ES, while supporting can also be known as intermediate. As this framework only accounts for the ways in which nature impacts human well-being, it is inherently anthropocentric (Bateman et al. 2011). While nature can and does have an intrinsic value apart from the value associated with the impact on human well-being, impact on well-being is measurable in economic terms and the ES framework only addresses these values.

Figure 1. The Ecosystem Services Cascade, adapted from Potschin and Haines-Young (2016)

A common classification system is necessary to be able to meaningfully discuss and compare ES. The classification developed by The Economics of Ecosystems and Biodiversity (TEEB) is an early example of this (Van der Ploeg et al. 2010). A
more recent system is the Common International Classification of Ecosystem Services (CICES). It was developed as part of work on the System of Environmental-Economic Accounting (SEEA) led by the United Nations Statistical Division (Haines-Young and Potschin 2018). The system is endorsed by the European Commission (Campagne et al. 2015) and will be used to classify ES here. The first version was published in 2013. Version 5.1 is the latest iteration and it was published in 2018 (Haines-Young and Potschin 2018). While earlier works such as the MEA described ES as the benefits people receive from ecosystems (MEA 2005), CICES defines them as the contributions ecosystems make to human well-being, distinct from the goods and benefits that are derived from them (Haines-Young and Potschin 2018). For example, seagrasses purify coastal waters – the contribution – which leads to reduced risk of illness or disease – the benefit. Therefore, in this classification, services are defined in terms of “what ecosystems do” (Haines-Young and Potschin 2018). In CICES and commonly in valuations to avoid double counting, only provisioning, regulating and cultural services are included (Haines-Young and Potschin 2018). As supporting services form the building blocks for all final services, including them in valuations would lead to inflated figures.

Since the publication of the MEA, the focus in the literature has increasingly fallen on the economic valuation of ES – or ecosystem service valuation (ESV) (Fisher et al. 2009; Gómez-Baggethun and Ruiz-Pérez 2011). Valuations of entire ecosystems can be an important tool in natural capital accounting and sustainable development, as maintaining natural capital stocks is essential for sustainable development and ensuring future provision of ES (Guerry et al. 2015). However, as ES are flows, ESV should assess how marginal changes in service provision (flows) impact human well-being. This requires ecological understanding of how external changes will affect ecosystems, and how these changes in stock will vary the level of ES provision (Bateman et al. 2011). Provisioning services are often traded in markets, so assigning a value to these services can be less complex. However, regulating and cultural services provide non-market benefits, and thus require non-market valuation.

ESV and non-market valuation has become a requirement in some policy areas (Hanley et al. 2015). As an example, the EU Water Framework Directive (WFD) allows states to extend the deadline for achieving good ecological status if it is “disproportionately expensive” (European Commission 2000). This is also the case for the European Marine Strategy Framework Directive and the Maritime Spatial
Planning Directive (Milon and Alvarez 2019). Good ecological status is not something that can be bought and sold on the market, so to ascertain whether achieving it is too costly, non-market valuation of the benefits associated with good ecological status is required (Hanley et al. 2015). Although ESV rhetoric is present in policy, the extent to which it has an influencing role rather than an informing one is questionable (Milon and Alvarez 2019). Part of the reason for this may be the observed gap between academic ESV research and policy applicable data and results (Olander et al. 2017). Non-market valuation methods have been in existence since the 1970s (Hanley et al. 2015), and commonly used methods for ESV can be found in Table 1. It has been noted that these methods, while established in academia, can be resource and data intensive and not always understood by stakeholders, which can limit their applicability in the policy-making realm (Olander et al. 2017).

Table 1. Common methods used for ecosystem service valuation

<table>
<thead>
<tr>
<th>Method</th>
<th>Overview</th>
<th>BCE ES that can be valued</th>
</tr>
</thead>
<tbody>
<tr>
<td>Adjusted market</td>
<td>Market prices adjusted for distortions</td>
<td>Contribution of saltmarsh to agriculture through grazing, e.g., saltmarsh-fed lamb (CICES 1.1.3.1 animals reared for nutritional purposes)</td>
</tr>
<tr>
<td>Production function</td>
<td>Estimates of production functions to isolate the impact of the ES as an input</td>
<td>Contribution of seagrass and saltmarsh to commercial fisheries (CICES 1.1.6.1 wild animals used for nutritional purposes)</td>
</tr>
<tr>
<td>Damage cost avoided</td>
<td>Costs avoided by the presence of ecosystems</td>
<td>Role of saltmarsh in coastal protection (CICES 2.2.1.3 hydrological cycle and water flow regulation)</td>
</tr>
<tr>
<td>Replacement cost</td>
<td>Cost of artificial substitutes for environmental goods or services</td>
<td>Contribution of seagrass to bio-remediation of coastal waters (CICES 2.1.1.1 bio-remediation by micro-organisms, algae, plants and animals)</td>
</tr>
<tr>
<td>Revealed preference</td>
<td>Examines expenditure on ecosystem related goods, e.g., travel costs for recreation, hedonic modelling with house prices for aesthetic properties</td>
<td>Birdwatching on saltmarshes (CICES 3.1.1.1 characteristics of living systems that enable activities promoting health, recuperation or enjoyment through active or immersive interactions)</td>
</tr>
<tr>
<td>Stated preference</td>
<td>Contingent valuation or choice modelling, using surveys to</td>
<td>Non-use or use value of seagrass and saltmarsh (CICES 3.1.2.3)</td>
</tr>
</tbody>
</table>
The ES approach is not without its criticisms. The lack of ecological data and understanding of the link between ecosystem functions and services, especially for marine and coastal ecosystems, has been cited as an impediment to producing meaningful valuations (Hanley et al. 2015; Norgaard 2010). Safe ecological thresholds are often unknown, and there can be disagreement between ecologists about what will trigger tipping points such as species extinction (Norgaard 2010). The ability to predict smaller changes is not guaranteed, yet it is often what is required of valuation studies which attempt to value marginal changes. In the face of ecological thresholds and the possible existence of some level of critical natural capital below which human welfare will be drastically reduced, there is a question to be answered about whether such incremental analysis is appropriate (Farley 2012). At the other end of the cascade, some concerns have been raised that well-being in the ES framework does not reflect the more holistic concepts of well-being that have been developed in recent years, as the focus remains on monetary units and number of beneficiaries (Rendón et al. 2019).

A further concern is that ESV is a step that may lead to the commodification of nature by incorporating ecosystem services into the market through monetization, appropriation – which often means privatization – and commercialization (Gómez-Baggethun and Ruiz-Pérez 2011). Practitioners have maintained that ESV is not intended to commodify nature, rather to make transparent the implicit trade-offs that humans already make in terms of ecosystem services (Costanza et al. 2014; de Groot et al. 2012). However, the prevailing socio-political processes and institutions in place since the 1980s and the enthusiasm for market-based solutions will influence how these valuations are used (Gómez-Baggethun et al. 2010; Gómez-Baggethun and Ruiz-Pérez 2011).

Ethical arguments aside, other critiques have focused on the policy relevance of ESV. Firstly, the uncertainty inherent in ESV and the range of possible values and valuation approaches may weaken arguments for conservation policies against developments with certain monetary benefits (D. King 1998). Secondly, even when links between function and service are known, studies are often based on a small site and selection of services, which can make application to broader policy

| estimate willingness to pay (WTP) or willingness to accept (WTA) changes in ES provision | characteristics of living systems that are resonant in terms of culture and heritage |

Adapted from Barbier 1997, Bateman 2011.
impractical (Milon and Alvarez 2019). Publication pressures may incentivize novelty and precision, resulting in papers that may be of little benefit to decision-makers due to their complexity and scale (Olander et al. 2017). The complexities of ESV are too extensive to discuss in their entirety here, for further information on the debates surrounding ESV, see Farley (2012), Gómez-Baggethun et al. (2010) and Norgaard (2010), and for a discussion on the current state of ESV in marine policymaking, see Hanley et al. (2015) and Tinch et al. (2021).

1.2 Blue Carbon Ecosystems

Oceans and coastal ecosystems play a significant role in the global carbon cycle, as marine living organisms capture over half of the carbon that is sequestered globally (Nellemann and Corcoran 2009). “Blue carbon” describes the carbon that is captured and stored by the oceans and, in particular, stored by vegetated coastal habitats (Nellemann and Corcoran 2009). Blue carbon ecosystems (BCEs) such as seagrasses, saltmarshes and mangroves cover less than 0.5% of the seabed, yet may account for up to 71% of all carbon storage in ocean sediments (Nellemann and Corcoran 2009). The capacity of BCEs to act as a carbon sink is however being threatened by degradation (Cott et al. 2021; Nellemann and Corcoran 2009). Research and management strategies are urgently needed to conserve these resources globally to combat both climate change and the biodiversity crisis.

1.2.1 Seagrasses

Seagrasses are marine flowering plants (angiosperms) found in shallow coastal waters on all continents except Antarctica (Short and Green 2003). Modelled estimates of the global extent of seagrass range from 300,000 km² (Duarte et al. 2005) to 1,646,788 km² (Jayathilake and Costello 2018). A 2020 study stated that between 160,387 km² and 212,562 km² of seagrass were mapped globally (McKenzie et al.). Difficulties with mapping and a paucity of data for bioregions in the developing world make estimates of their true extent challenging (McKenzie et al. 2020).

Seagrasses can sequester carbon and store significant amounts in the sediment and soils beneath them (de los Santos et al. 2020; Duarte 2017; Fourqurean et al. 2012). They are also an important nursery habitat, which contributes to commercial
and recreational fisheries (Nordlund et al. 2018). Through dampening wave movement (wave attenuation) and sediment stabilization, they protect the coast (Newell and Koch 2004; Paul and Amos 2011). Seagrasses remove nutrients and pollutants from the water by filtering, cycling, and storing them (de los Santos et al. 2020), which is one of their most valuable services (Costanza et al. 1997). Seagrass is often used in research and especially as a bioindicator (Govers et al. 2014), and there is potential to use seagrass wrack as a source of bioenergy (Balata and Tola 2018). Historically, it has been used as an insulation material, as medicine, and as a fertilizer (de la Torre-Castro and Ronnback 2004; Fernández et al. 2022; Wyllie-Echeverria and Cox 1999). In Europe, seagrass was harvested for use as a fertilizer up until the late 20th century (Santos and Duarte 1991), and there continues to be interest in the use of seagrass as an organic fertilizer around the world (Grassi et al. 2015; Emadodin et al. 2020).

Despite providing valuable services that impact human well-being, seagrasses are still threatened by water quality degradation, coastal development, and eutrophication (Duarte 2002; Dunic et al. 2021). These and other human associated impacts can alter food webs and contribute to further degradation (Fredriksen et al. 2004). As a result, seagrasses are in decline globally. It is estimated that 29% of seagrass coverage has been lost since the first known mapping (de los Santos et al. 2019). Previously it was found that seagrasses were disappearing at a rate of 110 km² per year since 1980 (Waycott et al. 2009), while a more recent study found that declining trends have stabilized and are recovering in some bioregions (Dunic et al. 2021). However, persistent declines of around 1-2% were still found in individual meadows, which raises concerns about the long-term viability of these ecosystems (Dunic et al. 2021). The most recent study measuring seagrass trends in Europe found that decadal loss peaked in the 1970s and has since been declining (de los Santos et al. 2019).

Alarm bells and calls for increased monitoring, modelling and education efforts for seagrass have been sounding since the early 2000s (Duarte 2002; Orth et al. 2006). A Web of Science search for publications that refer to seagrass returns over 14,000 results, 62% of which have been published in since 2010, indicating that research is increasing (Figure 2). However, difficulties remain. Large areas of seagrass remain unmapped, hindering monitoring efforts (McKenzie et al. 2020). Models are improving, yet there is still a lack of knowledge about the dynamics of their most well-known service – blue carbon (Unsworth et al. 2022). There have been efforts in Europe to educate the public about seagrass - see Barracosa et al.
Portugal (2018) - yet public perception of seagrass services is assumed to be low, and has been found to be so even among frequent users of these ecosystems (Barañano et al. 2022; Losciale et al. 2022).

1.2.2 Saltmarshes

Saltmarshes are coastal ecosystems found in the intertidal zone that are regularly flooded with salt or brackish water (Weis et al. 2016). They are found on every continent except for Antarctica, and are more commonly situated in middle to high latitudes (Weis et al. 2016). It has been estimated that they occupy 22,000 – 400,000 km² around the globe (Pendleton et al. 2012), yet a 2017 study puts the total global mapped saltmarsh area at 54,950 km² across 99 countries (McOwen et al.). Saltmarshes are highly productive ecosystems (Deegan et al. 2012; Gedan et al. 2009; Wilson and Koutsagiannopolou 2014), characterized by the presence of salt-tolerant plants, herbs and low shrubs (Weis et al. 2016).

Saltmarshes are a key BCE, acting as significant carbon sinks and providing sequestration services (Ribeiro et al. 2010; Weis et al. 2016). While blue carbon has recently been the focus of saltmarsh research efforts (Burke et al. 2021; Penk and Perrin 2022; McMahon et al. 2023), they also provide many other services. Saltmarsh edges are an important nursery habitat, which contributes to the health of recreational and commercial fisheries (Voltaire et al. 2017; Wilson and Koutsagiannopolou 2014). Saltmarshes protect the coast from hydrological disturbances and erosion (Gedan et al. 2010; S.E. King and Lester 1995; Liu et al. 2019; Weis et al. 2016). The creation and restoration of saltmarshes by inundation of previously dry or reclaimed land - a process known as managed realignment in the UK - is seen as a sustainable coastal defense option (Jones et al. 2011; Luisetti et al. 2011). They also regulate nitrogen levels (Alldred and Baines 2016; S.C.L. Watson et al. 2020), thereby controlling water quality (Breaux et al. 1995; Kazmierczak 2001). They are sites of recreational hunting, fishing, birdwatching and walking (Jones et al. 2011; Lewis 2019; McKinley et al. 2022). Traditionally and to the present day they have been used as grazing sites for ruminants (Curtis 2003; Davidson et al. 2017; McKinley et al. 2022). Historically, saltmarsh plants have been used as a raw material for insulation, fertilizers, and ornamental uses (Curtis 2003; Gedan et al. 2009).
As with many coastal ecosystems, saltmarshes are in decline, at an estimated rate of 1 - 2% per annum (McLeod et al. 2011). Large swathes of saltmarshes have been lost in previous centuries as the land was converted for agricultural or urban use (Gedan et al. 2009). In the southeast of Ireland for example, embankment and agricultural reclamation reduced much of the original saltmarsh area (Curtis and Sheehy Skeffington 1998). Where saltmarshes are used as grazing sites, inefficient grazing regimes can reduce the level of associated services (Curtis and Sheehy Skeffington 1998; Davidson et al. 2017). Land reclamation and dredging, particularly for coastal development, remains a threat (Gedan et al. 2009; Weis et al. 2016). Other human associated threats include eutrophication, hydromodification and sea level rise (Horton et al. 2018; Weis et al. 2016). Research on saltmarshes began earlier than seagrasses. The first publication referencing saltmarshes on the Web of Science is from 1945, and there are over 16,000 studies referencing saltmarsh since then (Figure 2). However, in recent years, the number of references to seagrass has been slightly higher.

2. LITERATURE REVIEW METHODOLOGY

A review of existing literature on seagrass and saltmarsh ESV was conducted following the systematic method described in Pickering and Byrne (2014) and the
literature selection guidelines in Moher et al. (2010). A database of economic valuation studies of saltmarsh and seagrass ES was constructed. Both primary studies and those using benefit transfer methods were included. Meta-analyses on their own were not included, but when they were identified in the search, the literature incorporated into the analyses was extracted and considered for inclusion.

Searches with variations of the terms: “valuation,” “seagrass” and “saltmarsh” were conducted in Science Direct, Ebsco and Google Scholar. This returned 3,567 studies. Abstracts and key words were scanned to narrow the selection, which reduced the total to 294. Global ESV databases have previously been developed, and these were also consulted. The TEEB database was published in 2010 and contains 1,310 value estimates from 267 studies (Van der Ploeg et al. 2010). This contributed to the Ecosystem Services Valuation Database (ESVD), by the Foundation for Sustainable Development, which is being continuously updated. Currently it contains 6,700 value records from over 950 studies (Foundation for Sustainable Development 2021). The Canadian Government maintains the Environmental Valuation Reference Inventory (EVRI), which collates studies of the economic values of environmental assets (Environment and Climate Change Canada 2022). After papers from the systematic review and the existing databases were gathered, duplicates were removed, and the full text was accessed and assessed for inclusion in the final database (Figure 3).

For each study, the full reference, location, socio-economic status of country, study area, geographical coordinates, condition, protected status, type of ecosystem, service valued, original value (including currency and year), method of valuation, and data type and source were added to the database. When the geographical coordinates were not included in the study, they were estimated using the place names with Google Maps. The socio-economic status of the country followed the classification of the UN World Economic Situations and Prospects Report (UN 2022). Study area, condition and protected status were not always included in the original paper. The services were classified according to the CICES class definition.
To enable comparison between values, monetary figures were standardized to 2018 international dollars by adjusting for purchasing power parity (PPP) and inflation, following Ghermandi et al. (2010). PPP indices and GDP deflators from the World Bank were used. The year 2018 was chosen as it was the year with the most studies published that was removed from influence of the distortion caused by COVID-19 and the inflation that has followed. When the currency year was not specified in the paper, it was presumed to be the year prior to publication, or the year of data collection in stated preference studies. If the original values were provided in a foreign currency, they were first converted to the currency of the country, unless they had already been adjusted for PPP. When exchange rates were provided in the paper, these were used to convert the values to the original currency, and where they weren’t provided historical exchange rates from the World Bank were used. Some studies provided global values, or values for entire regions such as Europe. These values were not corrected for PPP as they were not country
specific. GDP deflators at this scale are not available from the World Bank, but real and nominal GDP figures are available. These were used to construct global and regional GDP deflators.

Where possible, values were presented in a per hectare format, to allow for comparison. Some studies calculated marginal values, but many present average values per hectare. This implies a linear relationship between ecosystem extent and service provision, which may not always be the case (Milon and Alvarez 2019). Thus, while the per hectare values are presented for comparison, the original values are also available in the database. Values for stated preference studies could be presented in per hectare format if information was available on the extent of the ecosystem valued and the size of the population in question but this was not always provided in the study. Studies that undertook scenario analysis and provided net present values were standardized to present values in the same way as other figures. In most cases it was not possible to convert these values to an annual per hectare value, as the assumptions and prices applied in the scenarios changed year on year, and information on the original condition (extent especially) of the ecosystem was not always provided.

Standardizing the values to a common year and currency enables comparison across studies, but most of these values are specific to a time and place. While PPP and inflation have been accounted for, changes in the ecosystem extent and condition have not. For example, the per hectare value for nutrient cycling on a Louisiana saltmarsh may have been calculated in 1990. The 2018 international dollar value does not represent the present-day value of that service at that site, as in the intervening years, the value may have increased as service provision becomes scarce, or collapsed entirely if the saltmarsh has been degraded to an extent where it can no longer filter nutrients.

3. RESULTS

The final database included 162 studies with 736 associated value observations. Over half of the studies were retrieved through the systematic review searches. Many studies in the existing ESV databases had already been collected through the systematic review search, so fewer studies were added in this way. The most studies were found for seagrasses, which had 80 entries, compared to 58 for saltmarsh and 24 for studies valuing both seagrass and saltmarsh. Of the total studies, 81% were
peer reviewed articles. The grey literature included government and consultancy reports, conference papers, four masters’ theses, and one book. Peer reviewed articles made up 86% of the seagrass studies, 78% of the saltmarsh studies and 84% of the studies that valued seagrass and saltmarsh. The articles were published in 59 journals. *Ecosystem Services, Ocean and Coastal Management* and *Marine Pollution Bulletin* published the most seagrass articles, with six each. For saltmarsh studies, *Ecological Economics* and the *Journal of Environmental Management* published the most articles.

<table>
<thead>
<tr>
<th></th>
<th>Seagrass</th>
<th>Saltmarsh</th>
<th>Seagrass and Saltmarsh</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Systematic Review</td>
<td>51</td>
<td>30</td>
<td>17</td>
<td>98</td>
</tr>
<tr>
<td>ESVD</td>
<td>14</td>
<td>6</td>
<td>0</td>
<td>20</td>
</tr>
<tr>
<td>EVRI</td>
<td>4</td>
<td>5</td>
<td>2</td>
<td>11</td>
</tr>
<tr>
<td>TEEB</td>
<td>4</td>
<td>12</td>
<td>0</td>
<td>16</td>
</tr>
<tr>
<td>Meta-analysis</td>
<td>7</td>
<td>5</td>
<td>5</td>
<td>17</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>80</td>
<td>58</td>
<td>24</td>
<td>162</td>
</tr>
</tbody>
</table>

Figure 4. Number of valuation studies published over the years
The first seagrass valuation in this database is from 1989, although this is an outlier as only three more studies were published until the turn of the century. Most of the seagrass valuations (73%) were published from 2013 onwards. This is in line with the increase in scientific literature on seagrasses in the last decade referred to earlier. In order to value an ES, there needs to be sufficient ecological knowledge to estimate the impacts of a change in function of service provision (Bateman et al. 2011; Norgaard 2010). The first saltmarsh valuation was published in 1974, but only nine more were published up to the year 2000. The number of saltmarsh valuation studies has been steadily rising but has not increased as rapidly as seagrass studies have in the last 10 years. Over 60% of the ESV studies for both seagrass and saltmarsh have been published in the last 10 years.

3.1 Geographical Distribution

Figures 5 and 6 illustrate the location of seagrass and saltmarsh ES values found in this review. Some studies provided values for more than one location, and these are entered in the map separately. Data points that are inland represent studies that estimated values on a state-wide basis. Seagrass valuations are mainly located in Europe, Asia, and Oceania. Australia is the site of 20 studies. The United States follows with 8 valuations, yet Europe and Asia are the regions with the majority of seagrass valuation studies. The earliest Australian seagrass valuation study was published in 1993, but most of the studies are from after 2010. Australia is home to 31% of the world’s known mapped seagrass (McKenzie et al. 2020), which may explain the concentration of valuation studies there. Central and South America are underrepresented in the seagrass valuation literature, with the only study from Colombia (Zarate-Barrera and Maldonado 2015). This may be a result of the studies being restricted to those published in the English language, but as there is a known lack of data on seagrass extent in this area (McKenzie et al. 2020), it is likely to be a further significant research gap. Africa is the site of seven seagrass studies. There is a known link across the continent between seagrass and coastal fisheries (de la Torre-Castro and Ronnback 2004; Musembi et al. 2019; Ochieng and Erftemeijer 2003), and two studies attempt to qualify this. One ambitious study attempted to quantify benefits for the nine large marine ecosystems on the continent (Trégarot et al. 2020).
Saltmarsh valuations are overwhelmingly concentrated in the United States and the United Kingdom, with approximately 70% of the valuations found there. This is likely due to early recognition of the importance of coastal wetlands (Gosselink et al. 1974), and the national “No Net Loss” policy for wetlands in the US (Bauer et al. 2004). This policy requires that any development of wetlands must be offset by compensatory mitigation, i.e., the creation of new wetlands (Bauer et al. 2004). Economic values are often required to justify such developments (D. King 1998). These factors combined may be associated with the predominance of saltmarsh valuation studies in the US. In the UK, policy focus on managed realignment (Tinch and Ledoux 2006) may have stimulated interest in quantifying the benefits generated by saltmarshes. Africa and South America are not represented in the saltmarsh valuation literature. Similar to seagrass, mapping of saltmarsh extent in these regions is patchy and known to be missing large data points (McOwen et al. 2017), which likely contributes to the lack of saltmarsh valuations. It may also be the case that mangroves are more important wetland habitats in these regions as opposed to the North Atlantic region which is home to some of the largest areas of saltmarsh (McOwen et al. 2017).
3.2 Blue Carbon

Regulation of the chemical composition of the atmosphere and oceans (CICES code 2.1.6.1; hereafter this ecosystem service will be referred to by numerical code alone) is the service associated with blue carbon and climate regulation. This is the most commonly studied service for seagrass, with 29 associated studies. A further 14 studies valued climate regulation for seagrass and saltmarsh habitats, and 14 for just saltmarsh. A summary of climate regulation values for saltmarsh and seagrass can be found in the Supplementary Materials, in Tables S1 and S2, respectively.

The most common way to measure the value of this service was to estimate rates of carbon sequestration and storage at a site and combine this with carbon prices. In some studies field samples of carbon were taken (Beaumont et al. 2014; Carnell et al. 2019; Ganguly et al. 2018; Green et al. 2018; Lavery et al. 2013; Luisetti et al. 2019), but in most cases sequestration rates were taken from the literature. The price used to value carbon varied from study to study, with most studies using estimates based on the Social Cost of Carbon (SCC) or market prices. The SCC is the discounted cost of future damages associated with emitting a ton of carbon today (Nordhaus 2017; Tol 2005). Carbon is also traded in markets, notably under the European Union Emissions Trading Scheme (EU ETS), and 12 studies used carbon market prices. Studies based in the UK often used the non-traded
carbon values published by the UK Department of Business, Energy and Industrial Strategy (formerly published by the Department of Energy and Climate Change). These values are based on the cost of abatement and estimates are provided for up to the year 2100 (DECC 2011). Outside of the UK, other studies also based their prices on policy recommendations or national carbon tax rates, for example Campagne et al. (2015); Hynes, Norton, et al. (2013) and Wawo et al. (2014). In the papers found in this review, the lowest price used was US$2.93 tC\(^{-1}\) – based on the assumption that India’s share of the global SCC is 9% - (Ganguly et al. 2018) and the highest was $312 tC\(^{-1}\) - presented as part of a range of values (Luisetti et al. 2013). Some studies presented a range of estimates based on different carbon prices (Luisetti et al. 2013; Luisetti et al. 2019; McHarg et al. 2022).

While sequestration rates are commonly used to value this service, using carbon stock as an indicator is also an option. Measuring carbon stocks can be seen as an overly simplistic proxy for the climate regulation service (Boerema et al. 2017), yet knowing the original value of the stock is important for calculating the loss of future benefits of carbon storage if habitat is destroyed. The value of the stock was presented in the context of these analyses in González-García et al. (2022); dos Santos (2018); Luisetti et al. (2013) and Montero-Hidalgo et al. (2023) yet it was also presented as a value for the climate regulation service on its own in Bañolas et al. (2020); de los Santos et al. (2020); Praisankul and Nabangchang-Srisawalak (2016) and Wawo et al. (2014). This can be misleading, as the value of the stock doesn’t account for the continual benefits brought by sequestration. This was an issue observed in seagrass studies but not as often in saltmarsh studies. Most studies reported annual values for carbon sequestration, which showed variation as well. The range of values reported in the papers is found in Table 3. The value of carbon sequestration and storage depends not only on the price used, but also on site- and species-specific conditions.
Table 3. Range of values for climate regulation

<table>
<thead>
<tr>
<th></th>
<th>Min</th>
<th>Max</th>
<th>Median</th>
<th>Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Carbon storage</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Seagrass</td>
<td>29 ha⁻¹&lt;sup&gt;a&lt;/sup&gt;</td>
<td>91,919 ha⁻¹&lt;sup&gt;b&lt;/sup&gt;</td>
<td>654 ha⁻¹</td>
<td>14,841 ha⁻¹</td>
</tr>
<tr>
<td>Saltmarsh</td>
<td>22 ha⁻¹&lt;sup&gt;c&lt;/sup&gt;</td>
<td>1,246 ha⁻¹&lt;sup&gt;d&lt;/sup&gt;</td>
<td>240 ha⁻¹</td>
<td>503 ha⁻¹</td>
</tr>
<tr>
<td><strong>Carbon sequestration</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Seagrass</td>
<td>8 ha⁻¹·y⁻¹&lt;sup&gt;e&lt;/sup&gt;</td>
<td>1,146 ha⁻¹·y⁻¹&lt;sup&gt;f&lt;/sup&gt;</td>
<td>107 ha⁻¹·y⁻¹</td>
<td>218 ha⁻¹·y⁻¹</td>
</tr>
<tr>
<td>Saltmarsh</td>
<td>3 ha⁻¹·y⁻¹&lt;sup&gt;g&lt;/sup&gt;</td>
<td>1,540 ha⁻¹·y⁻¹&lt;sup&gt;h&lt;/sup&gt;</td>
<td>122 ha⁻¹·y⁻¹</td>
<td>271 ha⁻¹·y⁻¹</td>
</tr>
</tbody>
</table>

Source: <sup>a</sup> Ganguly <i>et al.</i> (2018) <sup>b</sup> González-García <i>et al.</i> (2022) <sup>c</sup> Luisetti <i>et al.</i> (2013) <sup>d</sup> Schmidt <i>et al.</i> (2014) <sup>e</sup> Sangha <i>et al.</i> (2019) <sup>f</sup> Bann and Başak (2013) <sup>g</sup> Dhivya <i>et al.</i> (2023) <sup>h</sup> (S.C.L. Watson <i>et al.</i> 2022). Median and mean values authors own calculations.

3.3. Seagrass

Seagrasses provide as many as 27 ES, provision of which varies according to region and species (Nordlund <i>et al.</i> 2016). A comprehensive study identified these services through expert elicitation and literature review and categorized them according to TEEB’s classification system (Nordlund <i>et al.</i> 2016). Some of these services have no corresponding valuations in the literature. Seagrasses used as food for humans (1.1.5.1) was only identified in the tropical Indo-Pacific bioregion by experts, and
reference to use as food in the literature was historical (Prendergast 2002), or concentrated in small traditional communities (Ochieng and Erftemeijer 2003). This explains the lack of valuations for this service.

Table 4. Studies valuing seagrass ES and location.

<table>
<thead>
<tr>
<th>Services</th>
<th>No. of Studies</th>
<th>Location of primary studies</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Provisioning</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Aquaculture (1.1.4.1)</td>
<td>2</td>
<td>CHN, SWE</td>
</tr>
<tr>
<td>Materials from wild plants (1.1.5.2)</td>
<td>4 (2)</td>
<td>TUS, VNM</td>
</tr>
<tr>
<td>Food from wild animals (1.1.6.1)</td>
<td>33 (5)</td>
<td>AUS, CHN, ESP, FRA, IDN, MLT, MOZ, PHL, PRT, SAU, SWE, THA, TUS, TZA, USA</td>
</tr>
<tr>
<td><strong>Regulating</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Decomposing wastes (2.1.1.1)</td>
<td>5 (3)</td>
<td>FRA</td>
</tr>
<tr>
<td>Filtering wastes (2.1.1.2)</td>
<td>17 (12)</td>
<td>AUS, GBR, POL, USA</td>
</tr>
<tr>
<td>Controlling erosion rates (2.2.1.1)</td>
<td>5 (3)</td>
<td>FRA</td>
</tr>
<tr>
<td>Regulating flows of water (2.2.1.3)</td>
<td>7 (2)</td>
<td>FRA, TUS, USA,</td>
</tr>
<tr>
<td>Providing habitats (2.2.2.3)</td>
<td>21 (6)</td>
<td>AUS, ESP, FRA, JPN, PRT, USA</td>
</tr>
<tr>
<td>Seawater quality (2.2.5.2)</td>
<td>8</td>
<td>AUS, FRA, JPN, POL</td>
</tr>
<tr>
<td>Climate regulation (2.2.6.1)</td>
<td>43 (6)</td>
<td>AUS, CHN, COL, ESP, FRA, GBR, IDN, IND, KEN, KOR, PRT, SWE, THA, TUR, USA</td>
</tr>
<tr>
<td><strong>Cultural</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Active recreation (3.1.1.1)</td>
<td>16 (3)</td>
<td>AUS, FRA, GBR, IDN, JPN, MLT, POL</td>
</tr>
<tr>
<td>Passive recreation (3.1.1.2)</td>
<td>5 (1)</td>
<td>AUS, FRA, IDN, MLT</td>
</tr>
<tr>
<td>Researching nature (3.1.2.1)</td>
<td>5 (2)</td>
<td>FRA, IDN, JPN</td>
</tr>
<tr>
<td>Aesthetic value (3.1.2.4)</td>
<td>1</td>
<td>IDN</td>
</tr>
<tr>
<td>Existence value (3.2.2.1)</td>
<td>13 (2)</td>
<td>AUS, FRA, GRC, KOR, PHL, THA, USA</td>
</tr>
</tbody>
</table>

Number of which benefit transfer studies in brackets

Seagrasses as a source of genetic material (1.2.1.1) was not mentioned by experts and only one paper was found to refer to it (Sinclair et al. 2013). The limited ecological knowledge about the use of seagrasses as a source of genetic material is likely a reason for the lack of valuations of this service. Cultural ES were the least frequently studied services (Figure 7). Recreation was the most frequently valued cultural service for seagrass. No primary valuations were found for the symbolic meaning of seagrass (3.2.1.1) or being significant in terms of culture and heritage.
(3.1.2.3) – these services were only valued as part of a bundle of cultural services in two benefit transfer studies (Cole et al., 2014; Brenner et al., 2010). Table S3 provides an overview of seagrass services and values that have been found for them in the literature.

### 3.3.1. Seagrass Provisioning Services

Thirty-six studies valued the provisioning services of seagrass, with the vast majority valuing their contribution to commercial fishery activity. Two studies valued the contribution of seagrass to animals reared by in-situ aquaculture (1.1.4.1) (S.G. Cole and Moksnes 2016; Han et al. 2008). One primary study used market prices to estimate the net benefit of the use of seagrass plants for direct use and processing (1.1.5.2), in the Tam Giang–Cau Hai lagoon complex in Vietnam (Tuan et al. 2009). Another looked at the value lost to fertilizer production due to the decline in seagrass beds (El Zrelli et al. 2023).

The contribution of seagrasses to commercial fisheries – wild animals used for nutritional purposes (1.1.6.1) had 33 associated studies. Seagrasses contribute to commercial fisheries through providing juvenile and adult habitat, as a food source, and through their regulating services (Nordlund et al. 2018). When studies provided values for the contribution of seagrass to commercial fisheries without naming explicitly which service they aimed to value, the CICES class was assumed to be 1.1.6.1. This can be an attractive service to study due to the availability of market data.

Most studies use market prices to value this service. The prices are adjusted to isolate the contribution of seagrass, but not adjusted for any market distortions. A common method is to estimate fish biomass in a seagrass area and combine this with market prices (Carnell et al. 2019; Erzini et al. 2022; Tuya et al. 2014; Unsworth et al. 2010). These values are often based on specific commercially targeted fish species. Average values using this method ranged from $20 ha⁻¹ y⁻¹ (Carnell et al. 2019) to $1,415 ha⁻¹ y⁻¹ (Tuya et al. 2014). Another method involves developing a seagrass residency index to weigh the importance of seagrass to the abundance of fish species. This index was applied to landed values for each species to give a monetary value for the seagrass contribution. This ranged from $14 ha⁻¹ y⁻¹ in the Mediterranean (Jackson et al. 2015) to $143 ha⁻¹ y⁻¹ in Southern Australia (McArthur and Boland 2006). Three Australian studies used stable isotope analysis,
which traces the nutritional value of seagrass in fish diets to assign part of the market value to seagrass. According to this method, the gross value of product associated with seagrass in Wallis Lake, New South Wales was $43 ha\(^{-1}\) y\(^{-1}\) (Raoult et al. 2022), and in the Richmond River estuary, $70,753 of annual estuary caught commercial catch value could be attributed seagrass (Jánes, Macreadie, et al. 2021). The average value of commercial fish catch attributable to seagrass across Australia was $ 9 ha\(^{-1}\) y\(^{-1}\) (Jánes et al. 2020). Some studies attributed a percentage of catch value to seagrass based on expert opinion or the literature (Han et al. 2008; Hughes 2015; Interwies and Görlitz 2013). Other studies, often in developing countries, used surveys of local fishers to make this link (Chitará-Nhandimo et al. 2022; S.G. Cole and Moksnes 2016; de la Torre-Castro et al. 2014; Dirhamsyah 2007; Praisankul and Nabangchang-Srisawalak 2016; Samonte-Tan et al. 2007).

Despite established methods of using production function models to value habitat-fishery linkages (Barbier 2007), only two studies used this method. Anderson calculated the long run consumer and producer surplus in the Virginia hard-shell blue crab fishery associated with an increase in seagrass habitat at $1,044 ha\(^{-1}\) and $776 ha\(^{-1}\) respectively (1989). Hoagland et al. calculated the marginal productivity of the coral reef-seagrass ecosystem in the fisheries of the Red Sea on the coast of the Kingdom of Saudi Arabia to be $24 - $329 ha\(^{-1}\) y\(^{-1}\) under different fishing regimes (2013).

### 3.3.2 Seagrass Regulating Services

Regulating services were addressed by 70 studies, the majority of which value climate regulation. Seagrasses remove waste from water and cycle nutrients, thus contributing to the quality of coastal waters. These services are described as bio-remediation (2.1.1.1), filtration/sequestration/accumulation of nutrients by plants (2.1.1.2) and regulating the chemical condition of salt waters (2.2.5.2) in the CICES classification. These services have similar impacts on human well-being, and determining which service is being valued can be difficult if it is not made clear in the study. A review of CICES 4.3 classes and the indicators used to measure them indicated that these services were often studied using the same indicators and the practical difference between them was small (Czúcz et al. 2018). When studies referred to general wastewater treatment, it was assumed they were valuing bio-remediation, and when they referred to a particular nutrient removal, it was assumed...
they were valuing filtration, following Costanza et al.\(^1\) (1997). The WTP for improvements to water quality was examined in some studies (Börger and Piwowarczyk 2016; Duijndam et al. 2020), which can be achieved through bio-remediation or filtration – as it was not explicit which of these services was being valued, the service was assumed to be regulating the chemical condition of salt waters. Only one study reported a negative benefit associated with this service, S.C.L. Watson et al. (2020) found that there was a net loss of phosphorous from seagrass, meaning the economic value of phosphorus removal is negative. These services were often valued using cost methods, see Table 5 for examples.

One study used a choice experiment to value the reduction in algae caused by the filtration service of seagrasses in the Gulf of Gdansk, Poland (Börger and Piwowarczyk 2016). The WTP to reduce the presence of algae ranged between $36 and $39 per household, and the WTP for improved visibility due to seagrass was $8. Other choice experiments investigated WTP for improved water quality in the Caribbean (Duijndam et al. 2020), and WTP for seagrass as an indicator of “clean, clear, sunlit waters” in Tasmania (Kragt and Bennett 2011). The oft cited study by Costanza et al. (1997) estimated the value of nutrient cycling by seagrass to be $19,002 ha\(^{-1}\) y\(^{-1}\) (1994 USD). This value and the underlying assumption that one third of the service is provided by coastal systems is used to estimate the value of this service in other locations through benefit transfer (Batker et al. 2010; Campagne et al. 2015; A.O. Cole et al. 2014; Han et al. 2008).

Further regulating services provided by seagrass include erosion protection (2.2.1.1) and the regulation of the hydrological cycle and water flow (2.2.1.3). Valuations for these services were made using replacement cost methods but were not as common. The value of erosion protection ranged from $120 ha\(^{-1}\) y\(^{-1}\) in Turkey (Bann and Başak 2013) to $4,028 ha\(^{-1}\) in 2005 in Guangxi Province, China (Han et al. 2008). The regulation of the hydrological cycle and water flow was valued in Martinique at $15,464 ha\(^{-1}\) y\(^{-1}\) (Failler et al. 2015). Chen, Swallow and Yue (2020) found household WTP for “living shorelines” in East Virginia (a combination of seagrass and saltmarsh restoration to protect the coast) ranged from $35 – $4,430.

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\(^1\) In Costanza et al.’s global valuation of ES and Natural Capital, nutrient cycling was defined as the storage, internal cycling, processing, and acquisition of nutrients such as nitrogen and phosphorous, while waste treatment referred to the recovery and removal or breakdown of excess foreign nutrients (1997). These definitions correspond to CICES 2.1.1.2 and CICES 2.1.1.1 respectively.
Maintaining nursery habitats and populations (2.2.2.3) was the most frequently studied regulating service after climate regulation. As mentioned previously, the nursery habitat contributes to the importance of commercial fisheries. Adjusted market prices were used in much the same way as for valuing the provisioning service of seagrasses, where the modelled biomass was combined with prices. However, these values focused specifically on fishery enhancement and the population and growth of juveniles in the habitat. This was studied in Gran Canaria, Spain (Tuya et al. 2014), Australia (Blandon and zu Ermgassen 2014; R. Watson et al. 1993), Portugal (Erzini et al. 2022), and Japan (Kamimura et al. 2011). The lowest value found using this method was $140 ha\(^{-1}\) yr\(^{-1}\) (Erzini et al. 2022), and the highest value was $22,788 ha\(^{-1}\) yr\(^{-1}\) (Blandon and zu Ermgassen 2014).

Table 5. Filtration and bio-remediation service of seagrass valued with cost methods

<table>
<thead>
<tr>
<th>Method</th>
<th>Value (2018 Int$)</th>
<th>Description</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avoided damage costs</td>
<td>25 million</td>
<td>Annual avoided damages globally through reduced cases of gastroenteritis</td>
<td>(Ascioti et al. 2022)</td>
</tr>
<tr>
<td>Replacement costs</td>
<td>1,501 ha(^{-1}) yr(^{-1})</td>
<td>Wastewater treatment provided by seagrasses in Martinique</td>
<td>(Failler et al. 2015)</td>
</tr>
<tr>
<td>Replacement costs</td>
<td>8.5 million</td>
<td>Increase in denitrification from seagrass between 1982 and 2010 in Tampa Bay, Florida</td>
<td>(Russell and Greening 2015)</td>
</tr>
<tr>
<td>Replacement cost</td>
<td>87,507 ha(^{-1}) yr(^{-1})</td>
<td>Value of nitrogen removal on southeast coast of the UK</td>
<td>(S.C.L. Watson et al. 2020)</td>
</tr>
<tr>
<td>Replacement cost*</td>
<td>-19,979 ha(^{-1}) yr(^{-1})</td>
<td>Value of phosphorous removal on the southeast coast of the UK</td>
<td>(S.C.L. Watson et al. 2020)</td>
</tr>
<tr>
<td>Replacement cost</td>
<td>480 – 1,173 ha(^{-1}) yr(^{-1})</td>
<td>Nitrogen removal by seagrass along Adelaide coast</td>
<td>(Gaylard et al. 2023)</td>
</tr>
</tbody>
</table>

* This value is negative as study found that seagrasses were a source of phosphorous rather than a sink.
This service was also valued through stated preference methods (Antinori 2021; Duijndam et al. 2020; Rogers 2013; Pascoe et al. 2019; Tsuge and Washida 2003; Uehara et al. 2018). A contingent valuation study in Japan elicited WTP for restoration of 10 ha of *Zostera marina* based on its importance as a nursery habitat, and the aggregated WTP for the country was $1bn (Tsuge and Washida 2003). Uehara et al. ran the same survey again in 2015 and found that the mean WTP for this project had increased by 17.4% (2018). A choice experiment in Australia estimated WTP for a 5% increase in seagrass area in a proposed marine park, given the importance of its nursery service, as between $34 - $42 per household per year (Rogers 2013).

### 3.3.3 Seagrass Cultural Services

Twenty-nine studies addressed the cultural value of seagrass ecosystems, some of which provided values for a bundle of cultural ES (A.O. Cole et al. 2014; Pascoe et al. 2019). The contribution of seagrass to recreational fishing (characteristics of living systems that enable activities promoting health, recuperation or enjoyment through active or immersive interactions (3.1.1.1) was the most frequently studied cultural service for seagrass. Port Phillip Bay in Melbourne, Australia has been the site for multiple valuations of this service. Jänes, Carnell, et al. (2021) examine the impact of seagrass on a particular species and combine fish abundance models with recreational fishing trip values from the Victorian Fisheries Authority (VFA) dataset. The recreational fishing value of King George Whiting supported by seagrass is estimated to be $583 - $4,439 ha\(^{-1}\) y\(^{-1}\). Huang et al. (2020) also use the recreational fishing dataset from the VFA to estimate WTP for changes to fishing site and find that a 10% increase in seagrass in Port Phillip Bay is associated with increased annual recreational benefits of $560,405. The Mapping Ocean Wealth Australia report used travel cost and choice modelling on an existing dataset, and valued recreational fishing related to seagrass at $738 ha\(^{-1}\) y\(^{-1}\) in the same location (Carnell et al. 2019). The value of recreational fishing was also studied through a combination of travel cost values and fish and angler distribution surveys in Florida (Fulford et al. 2016), through a survey of the local population in East Bintan, Indonesia (Wahyudin et al. 2018), and through a residency index approach in Europe (Jackson et al. 2015). A report investigating the value of potential Marine Protected Areas for divers and anglers in the UK found WTP to visit Blakeny seagrass site was between $10 and $92 (Kenter et al. 2013).
Although tourism is not explicitly included as a cultural service in CICES, it was included in the TEEB. Notwithstanding the debate around whether tourism is a cultural ES – see Pueyo-Ros (2018) – where tourism was valued in a paper it has been included in the review under the assumption that it relates to characteristics of a living system that enable activities promoting health, recuperation or enjoyment through passive or observational interactions, e.g., sightseeing (3.1.1.2). Dirhamsyah (2007) estimate the value of seagrass for foreign and local tourism at $5,160 ha\(^{-1}\) y\(^{-1}\). They attribute all foreign tourist expenditure to the presence of seagrass and use a benefit transfer from a travel cost study at a similar site to estimate local tourism value. This is a problematic form of valuation, as it is highly unlikely that all tourist visits to the site are due to the presence of seagrass. The value of seagrass for tourism was also calculated in Thailand (Praisankul and Nabangchang-Srisawalak 2016) and Martinique (Failler et al. 2015).

Seagrass enables scientific investigation (3.1.2.1) and is often used as a bioindicator. Three studies use research funding for seagrass related projects as a proxy for this value (Campagne et al. 2015; Dirhamsyah 2007). In the CICES terminology the simple descriptor of the existence value (3.2.2.1) of an ecosystem is that it is something we think should be conserved. This service can only be valued through stated preference methods. Stated preference surveys that estimated WTP for seagrass restoration (Praisankul and Nabangchang-Srisawalak 2016; Rolfe and Windle 2010), for preservation (Bundal et al. 2018; Pascoe et al. 2019), for maintaining good environmental condition of a specific species (Halkos and Galani 2016), and for protecting endangered species (Wallmo and Lew 2016; Petcharat and Lee 2020) were all presumed to value the existence of seagrass.

3.4. Saltmarsh

Nineteen services were valued for saltmarsh, with the majority of valuations for regulating services (see Figure 8 and Table 6). More saltmarsh studies than seagrass studies valued cultural ES, and less valued provisioning. Some studies valued services for “coastal wetlands” with no further description of the ecosystems. When the description of the sites in the text and comparison with the known distribution of saltmarshes confirmed the likelihood of the ecosystem being a saltmarsh, they were included in this review. A range of values from primary studies found in this
review for saltmarsh ES that can be compared for a like for like basis can be found in Table S4.

Figure 8. Valuation studies for saltmarsh ES according to CICES section

<table>
<thead>
<tr>
<th>Services</th>
<th>No. of Studies</th>
<th>Primary study locations</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Provisioning services</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Food grown by humans (1.1.1.1)</td>
<td>1</td>
<td>LKA</td>
</tr>
<tr>
<td>Reared animals (1.1.3.1)</td>
<td>4 (1)</td>
<td>GBR, NLD, USA</td>
</tr>
<tr>
<td>Aquaculture (1.1.4.1)</td>
<td>1</td>
<td>USA</td>
</tr>
<tr>
<td>Materials from wild plants (1.1.5.2)</td>
<td>5 (3)</td>
<td>CHN</td>
</tr>
<tr>
<td>Food from wild animals (1.1.6.1)</td>
<td>24 (4)</td>
<td>AUS, CHN, GBR, KOR, LKA</td>
</tr>
<tr>
<td><strong>Regulating services</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Decomposing wastes (2.1.1.1)</td>
<td>9 (4)</td>
<td>KOR, LKA, USA</td>
</tr>
<tr>
<td>Filtering wastes (2.1.1.2)</td>
<td>15 (7)</td>
<td>CHN, GBR, USA</td>
</tr>
<tr>
<td>Controlling erosion rates (2.2.1.1)</td>
<td>3 (3)</td>
<td></td>
</tr>
<tr>
<td>Regulating flows of water (2.2.1.3)</td>
<td>31 (12)</td>
<td>CHN, GBR, LKA, NLD, USA</td>
</tr>
<tr>
<td>Pollination (2.2.2.1)</td>
<td>1</td>
<td>NLD</td>
</tr>
<tr>
<td>Providing habitats (2.2.2.3)</td>
<td>18 (6)</td>
<td>AUS, CHN, GBR, USA</td>
</tr>
<tr>
<td>Disease control (2.2.3.2)</td>
<td>1</td>
<td>USA</td>
</tr>
</tbody>
</table>
Climate regulation (2.2.6.1) 27 (4) AUS, CHN, GBR, IRL, KOR, LKA, NLD, PRT, USA

Cultural services

Active recreation (3.1.1.1) 31 (14) AUS, CHN, GBR, ESP, LKA, NLD, USA
Passive recreation (3.1.1.2) 6 (2) AUS, USA
Researching nature (3.1.2.1) 2 (1) GBR
Cultural and heritage value (3.1.2.3) 2 (2)
Aesthetic value (3.1.2.4) 4 USA
Existence value (3.2.2.1) 7 CHN, FRA, ITA, USA

Number of which benefit transfer studies in brackets

3.4.1. Saltmarsh Provisioning Services

Cultivated terrestrial plants grown for nutritional purposes (1.1.1.1) and animals reared for nutritional purposes (1.1.3.1) were not often valued. This is not surprising given the somewhat limited use of saltmarsh for agriculture production. However, there are some examples such as the Muthurajawela Wetlands in Sri Lanka, where the gross returns from saltmarsh-based agriculture (coconuts, bananas and vegetables) were attributed entirely to the marsh area, giving a value of $731 ha\(^{-1}\) y\(^{-1}\) (Emerton and Kekulandala 2003). Though not a direct valuation of saltmarsh agriculture, a choice experiment near Jiangsu-Yancheng Coastal Wetlands in China investigated what subsidy would be necessary for farmers to enroll in a more sustainable form of saltmarsh agriculture by using less pesticides, and found that the minimum WTA per farmer ranged from $22 to $591 ha\(^{-1}\) y\(^{-1}\), depending on the program characteristics (Bennett et al. 2018). Saltmarsh has also traditionally been used in agriculture as grazing pasture for ruminants, in France for example, saltmarsh-grazed lamb is a premium product (Gedan et al. 2009; Jones et al. 2011). In the Scheide Estuary in the Netherlands, saltmarsh contributes to added value of this product by $949 ha\(^{-1}\) (Boerema et al. 2016). A 1995 UK study estimated the gross margins for livestock grazing on saltmarsh in the UK at $199 - $541 ha\(^{-1}\) (Posford Duvivier 1996) and another study estimated the grazing value at $39 ha\(^{-1}\) (S.E. King and Lester 1995). The potential value of saltmarsh-based aquaculture (1.1.4.1) in the US was valued by Gosselink at $6,857 ha\(^{-1}\) y\(^{-1}\) (1974). The use of saltmarsh plant fibers for direct use or processing (1.1.5.2) was valued using adjusted market prices in Lianyugang, China at $90 ha\(^{-1}\) y\(^{-1}\), where saltmarsh plants are collected for use as a fodder. (Sun et al. 2017).
Much like seagrasses, multiple saltmarsh ES contribute to commercial fisheries by providing food and nursery habitat for commercial species, and through regulating services (Jänes, Macreadie, et al. 2021). When the service was not explicitly mentioned, it was assumed the contribution of saltmarshes to the production of wild animals for food was being valued. Twenty-three studies valued CICEs 1.1.6.1 – wild animals used for nutritional purposes - three of which used benefit transfer.

Market prices were a common way to value this service. In some cases, they were not adjusted at all. For example, Gosselink et al. (1974), assigned the entire value of coastal fisheries to saltmarsh in Georgia, Louisiana and Florida – based on the assumption that the existence of the fishery depended on the existence of the habitat. Similarly, the value of Muthurajawela Wetlands is expressed as the entire value of the local fishery and 10% of the downstream catch (Emerton and Kekulandala 2003). Stable isotope analysis was also used to trace the contribution of saltmarsh plants to the diet of commercial fish, and then adjust prices accordingly. In Australia, the gross value of production of commercial fish catch attributable to saltmarsh ranged from $5 ha\(^{-1}\) y\(^{-1}\) in Victoria (Jänes et al. 2020) to $422 ha\(^{-1}\) in Wallis Lake, New South Wales (Raoult et al. 2022). A residency index was used by McCormick et al. (2021) to assign a value to saltmarsh based on commercial landings. They found that 22.2% - 24.7% of UK landings value and 45.6% - 49% of European landing values in 2015 could be attributed to UK saltmarsh, giving values of $147 ha\(^{-1}\) and $817 ha\(^{-1}\) respectively. Modelling the biomass of fish found in seagrass areas and combining with catch values was a method used by Carnell et al. (2019) and Minello et al. (2012). Only two studies used production function methods to estimate this service; (Costanza et al. 1989; Stevenson 2001). The marginal productivity of one hectare of saltmarsh in a coastal fishery in Louisiana was estimated at $135 (Costanza et al. 1989).

Stated preference methods were also used to value the contribution of saltmarsh to commercial fisheries. Household WTP to improve fisheries productivity through conserving Louisiana’s coastal wetlands was $212 (Petrolia et al. 2014). Similarly, household WTP to increase fish catch through saltmarsh restoration in Alabama and Louisiana ranged from $42 - $102 (Interis and Petrolia 2016)
3.4.2. Saltmarsh Regulating Services

The most studied regulating service was hydrological cycle and water flow regulation, including flood control and coastal protection (2.2.1.3), making up 30% of the valuations of saltmarsh regulating services. As with seagrass studies, distinguishing between valuations of bio-remediation (2.1.1.1) and filtration (2.1.1.2) was difficult, and was done in the same way as detailed above. Both services were valued through replacement cost methods and benefit transfer. The high capital costs associated with building and maintaining treatment plants that would provide the service currently offered by saltmarsh means high values are found for this service. Replacement cost estimates for bio-remediation ranged from $104 ha\(^{-1}\) y\(^{-1}\) for installing domestic sewage treatment in Sri Lanka (Emerton and Kekulandala 2003) to $154,139 ha\(^{-1}\) y\(^{-1}\) for complete tertiary treatment of artificial nutrients in the Mid-Atlantic estuaries of the US (Gosselink et al. 1974). For filtration, the replacement cost estimates ranged from $823 ha\(^{-1}\) y\(^{-1}\) in Florida, USA (Russell and Greening 2015) to $181,210 ha\(^{-1}\) y\(^{-1}\) on the southeast UK coast (S.C.L. Watson et al. 2020), both figures representing the value for nitrogen removal.

Flood control and coastal protection, (2.2.1.3) was valued through replacement cost methods, stated preference methods, historical damage cost avoided, and benefit transfer. Table 7 shows the values calculated using cost methods. Stated preference methods were also used to value this service. As mentioned, Chen, Swallow and Yue et al., investigated WTP for nature based solutions for coastal protection involving seagrass and saltmarsh (2020). In the UK, aggregate household consumer surplus from flood prevention by intertidal and saltmarsh areas in SSSIs was estimated at $22.2m y\(^{-1}\) (Christie and Rayment 2012). Also in the UK, the aggregate WTP for managed realignment, a nature based coastal defense strategy, ranged from $6,158 ha\(^{-1}\) y\(^{-1}\) to $137,948 ha\(^{-1}\) y\(^{-1}\) under different policy scenarios (Luisetti et al. 2011).

A post Hurricane Katrina study in the US compared preferences for coastal restoration as a means to improve storm protection in New Orleans and nationwide. The average individual WTP for coastal restoration was $120, although it was not always statistically significant for the New Orleans sample, and lower income households had no significant WTP for this service (Landry et al. 2011). A contingent valuation study in the UK found the mean household WTP to prevent a reduction in saltmarsh to preserve their flood protection service ranged from $67 to $622 y\(^{-1}\) (Mangi et al. 2011). In Louisiana, a contingent valuation survey estimated...
household WTP/WTA to prevent/accept coastal wetlands loss at $954 and $5,087 respectively (Petrolia and Kim 2011). In Ireland, the coastal protection service of saltmarsh was valued using benefit transfer at $3,060 ha$^{-1}$ y$^{-1}$ (Norton et al. 2018).

One study valued pollination (2.2.2.1), estimating that Dutch saltmarshes contributed $207$ ha$^{-1}$ in 2015 to increased crop yield through pollinating services, using what they describe as a damage cost avoided approach, modelling the relationship between pollinator dependent crops and habitat suitability for crops (Horlings et al. 2020).

While many studies valued the contribution of saltmarsh to commercial fisheries, only two studies used fishery market data to explicitly value the service of maintaining nursery populations and habitats (2.2.2.3). An early study used the production function method to estimate the productivity of marsh in the blue crab fishery of Virginia at $2$ ha$^{-1}$ (Lynne et al. 1981). In the UK, the importance of saltmarsh as a nursery ground for bass was valued at $13$ ha$^{-1}$ (Luisetti et al. 2011). This service was also valued through choice experiments that elicited WTP to improve habitat conditions for saltmarsh fauna. Household WTP to improve saltmarsh habitat for protected bird species was estimated to be between $3$ and $5$ (Luisetti et al. 2011). A restoration option that improved 5 acres of marsh habitat for fish nurseries in Rhode Island, US was valued at $26$ per household (Bauer et al. 2004).

### 3.4.3. Saltmarsh Cultural Services

Cultural services made up a larger proportion of valuations for saltmarsh than for seagrass, with 36 studies valuing these services. Over half of these studies were related to recreational values, or characteristics of living systems that enable activities promoting health, recuperation, or enjoyment through active/immersive or passive/observational activities (3.1.1.1 and 3.1.1.2). Stated and revealed preference methods were used to value these services. Table 8 shows a sample of values for this service.
<table>
<thead>
<tr>
<th>Method</th>
<th>Value (2018 Int$)</th>
<th>Description</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avoided damage cost</td>
<td>259,117 yr⁻¹</td>
<td>Annual value of flood protection provided by managed realignment</td>
<td>(MacDonald et al. 2020)</td>
</tr>
<tr>
<td>Avoided damage cost</td>
<td>29,252 ha⁻¹</td>
<td>Median value of avoided damages per storm in China between 1989-2016</td>
<td>(Liu et al. 2019)</td>
</tr>
<tr>
<td>Avoided damage cost</td>
<td>206,762 ha⁻¹</td>
<td>Mean value of avoided damaged per storm in China between 1989-2016</td>
<td>(Liu et al. 2019)</td>
</tr>
<tr>
<td>Avoided damage cost</td>
<td>2,429 yr⁻¹</td>
<td>Average avoided property damages due to the presence of saltmarsh per house</td>
<td>(Narayan et al. 2017).</td>
</tr>
<tr>
<td>Avoided damage cost</td>
<td>9,872 ha⁻¹ yr⁻¹</td>
<td>Avoided flood damage cost through presence of saltmarsh in Scheide Estuary, the Netherlands</td>
<td>(Boerema et al. 2016)</td>
</tr>
<tr>
<td>Avoided damage cost</td>
<td>10,231-40,333 ha⁻¹</td>
<td>Present value of increased storm damages associated with loss of saltmarsh</td>
<td>(Costanza et al. 1989)</td>
</tr>
<tr>
<td>Replacement cost</td>
<td>1,257 ha⁻¹</td>
<td>Replacement cost of disturbance regulation of saltmarsh in Lianyungang, China</td>
<td>(Sun et al. 2017).</td>
</tr>
<tr>
<td>Replacement cost</td>
<td>15.2m - 210.2m</td>
<td>Capital savings on coastal defense by Tetney Marshes and St. Peter’s flats, UK</td>
<td>(Mangi et al. 2011).</td>
</tr>
<tr>
<td>Replacement cost</td>
<td>11,714 ha⁻¹ yr⁻¹</td>
<td>Replacing and maintaining the flood control measures supplied by Muthurajawela marsh</td>
<td>(Emerton and Kekulandala 2003)</td>
</tr>
<tr>
<td>Replacement cost</td>
<td>772,004 ha⁻¹</td>
<td>Capital savings on sea defense provided by saltmarsh, UK</td>
<td>(S.E. King and Lester 1995).</td>
</tr>
<tr>
<td></td>
<td>15,440 ha⁻¹</td>
<td>Annual maintenance cost savings</td>
<td></td>
</tr>
</tbody>
</table>
### Table 8. Value of saltmarsh recreation service

<table>
<thead>
<tr>
<th>Method</th>
<th>Value (2018 Int$)</th>
<th>Description</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Travel cost and contingent valuation</td>
<td>246 ha⁻¹ - 967 ha⁻¹</td>
<td>Recreational fishing in Louisiana</td>
<td>(Costanza et al. 1989)</td>
</tr>
<tr>
<td>Travel cost and contingent behavior model</td>
<td>168 – 190 per visit</td>
<td>Value per trip of visit to La Pleta saltmarsh, Catalonia</td>
<td>(Pueyo-Ros et al. 2018)</td>
</tr>
<tr>
<td>Travel cost and on-site expenditures</td>
<td>127 ha⁻¹</td>
<td>Recreation at Muthurajawela wetlands</td>
<td>(Emerton and Kekulandala 2003)</td>
</tr>
<tr>
<td>Travel cost</td>
<td>73 per visit</td>
<td>Tourist visit to Nansha Wetland, China</td>
<td>(Xu and He 2022)</td>
</tr>
<tr>
<td>Travel cost</td>
<td>317 ha⁻¹ yr⁻¹</td>
<td>Hunting and fishing in Saginaw Bay saltmarsh, Michigan</td>
<td>(Whitehead et al. 2009)</td>
</tr>
<tr>
<td>Fishery production and WTP for recreational fishing days</td>
<td>5,059 - 33,369 ha⁻¹</td>
<td>Capitalized value of saltmarsh in East and West Coast of Florida for recreational fishing</td>
<td>(Bell 1997)</td>
</tr>
<tr>
<td>Contingent valuation</td>
<td>52.9 million</td>
<td>Consumer surplus of recreation on Louisiana saltmarsh</td>
<td>(Bergstrom et al. 1990)</td>
</tr>
<tr>
<td>Choice modelling</td>
<td>8,705 - 194,797 yr⁻¹</td>
<td>Birdwatching on saltmarsh in Australia</td>
<td>(Carnell et al. 2019)</td>
</tr>
<tr>
<td>Choice modelling and contingent valuation</td>
<td>8 – 79 per visit</td>
<td>Mean individual WTP among divers and anglers for visit to Blakeny saltmarsh site, UK</td>
<td>(Kenter et al. 2013)</td>
</tr>
</tbody>
</table>
Characteristics that enable scientific investigation or the creation of traditional ecological knowledge (3.1.2.1) were also valued. The aggregate household consumer surplus for research and education opportunities provided by intertidal mudflats and saltmarsh in SSSIs in the UK was $15.24m y\(^{-1}\), according to an approach that combined choice modelling with weighting scores from experts (Christie and Rayment 2012). This study also estimated values for “sense of experience” and “charismatic species,” which were taken to refer to characteristics that have an existence value (3.2.2.1). Hedonic modelling and choice modelling was used in Connecticut, US to value the aesthetic service of saltmarsh (3.1.2.4) and found increased house prices of $17,958 - $317,595 due to proximity to restored saltmarsh (Earnhart 2001). The only other study to value aesthetics was Yoskowitz et al. (2017), using benefit transfer from similar ecosystems.

The existence value of saltmarsh had seven associated studies. Stated preference surveys that investigated WTP for protection or preservation of saltmarsh habitats, or for charismatic or endangered species who use these habitats, were assumed to be calculating the existence value. A choice experiment estimated the marginal value of a hectare of saltmarsh on Deben Estuary ranged from $34 ha\(^{-1}\) to $45 ha\(^{-1}\), depending on the exact project aims (Grilli et al. 2022). In Brest roadstead, France, a contingent valuation survey found WTP to protect saltmarshes against invasive species was $17-$30 per household (Voltaire et al. 2017). The mean WTP per tourist for protection of Nansha Wetlands, China, was valued with choice modelling at $27 y\(^{-1}\) (Xu and He 2022). The non-use value for Saginaw Bay saltmarsh was $93 ha\(^{-1}\) y\(^{-1}\), found through contingent valuation (Whitehead et al. 2009). Other studies valued cultural services of saltmarshes through benefit transfer, e.g. Brenner et al. (2010), sometimes as a bundle of cultural services, e.g. A.O. Cole et al. (2014). The full range of values for seagrass and saltmarsh services from the literature, which can be compared on a like for like basis, can be found in the Supplementary Materials section, in Tables S3 and S4, respectively.

4. DISCUSSION

4.1 Ecosystem condition

The literature shows a broad range of values for saltmarsh and seagrass ES, based on a variety of methods (See Table 9). The level of services an ecosystem can provide is dependent on the condition of the ecosystem itself. A saltmarsh
threatened by anthropogenic impacts such as eutrophication will not be able to provide the same level of ES as a saltmarsh in good condition in a protected area. S.C.L. Watson et al. (2022) show that the value of regulating services provided by BCEs depends on the condition of the habitat. However, the condition and protected status of the ecosystem in question is only reported in 28% of studies. Just over half of the studies that valued seagrass ES reported the species in question, which may be an issue where service provision varies with species (Lavery et al. 2013). The reporting of both of these metrics should be improved where possible, not only for the quality of the original assessment, but also to inform any benefit transfer studies developed using these values.

4.2. Methods

4.2.1. Direct Market Valuation

Market prices were used to value the contribution of blue carbon ecosystems to the provision of wild food, the maintenance of nursery habitats and populations, and in some cases the climate regulation service. These prices were adjusted in most cases to value the services related to commercial fisheries, while for climate regulation the market value of carbon was not adjusted. Using market data carries the implicit assumption that the market is competitive and the prices therefore are reflective of marginal costs (Pascual et al. 2010). Whether the carbon market or the commercial fish market are functioning well is debatable. The negative externalities of commercial fishing have been well-documented. About 90% of the world’s fisheries were subject to overfishing in 2011 (World Bank 2017), a practice that is contributing to the global extinction crisis (Dulvy et al. 2021). It was estimated that in 2012 the industry was forgoing billions of dollars in benefits compared to a sustainable management scenario (World Bank 2017). The industry is also heavily subsidized, to the tune of $35.4 billion in 2018, despite evidence that subsidies that artificially inflate profits result in overcapacity and overfishing (Sumaila et al. 2019). In this context, it is questionable whether landed prices reflect the true value of the fish caught, and to use these prices in ESV may give rise to distorted values. Rather than adjusting market prices through the methods mentioned here, isolating the impact of the ecosystem on fishery production through a production function approach may be more of an appropriate economic method. However, this method
has only been used by six studies in this review, raising the question of whether there is sufficient data available to inform this approach. (Pascual et al. 2010).

The replacement cost method was commonly used to value the bio-remediation, filtration and coastal defense services. This method equates the cost of replacing a service with the benefit of that service. This may be misleading as the true benefit is the demand for the benefits these services bring (i.e. clean water) minus the cost of providing it (Barbier 2007), so using just cost figures can lead to unrealistically high values. In some cases, this value may be appropriate to inform policy decisions. If the choice is between constructing a treatment plant or restoring the saltmarsh that provides this service, comparing the costs of construction and restoration is suitable (Barbier 2007). For bio-remediation and filtration services, the impact on human well-being may be difficult to quantify and the paucity of data may in some cases require replacement cost methods as the best available option. The overuse of replacement costs to estimate the value of coastal protection by BCEs was cited in Himes-Cornell et al. (2018), and it has been advised that more reliable estimates can be generated using a damage function approach (Barbier 2016).

Damages are those to property, infrastructure, industry, and in some studies loss of life (Liu et al. 2019; Narayan et al. 2017). Therefore, the avoided damage costs depend on the baseline wealth of the affected communities. This raises ethical issues. For example, the storm protection benefit of mangroves is associated with higher economic benefits in high income countries, but a greater number of beneficiaries, i.e. human lives protected, in lower-income countries (Menéndez et al. 2020). Using this method to value the flood control and coastal protection service of saltmarshes and seagrasses runs the risk of providing greater justification for preservation and protection of these ecosystems that are located in wealthier areas (zu Ermgassen et al. 2021), a particularly insidious risk when it is considered that the global poor in general are more dependent on ecosystems and their services than the global rich (Ghermandi et al. 2013).

Methods using direct market valuation present an exchange value, which is a financial, rather than an economic value and does not take welfare into account (Tinch et al. 2021). While exchange values are necessary for current natural capital frameworks, welfare values are necessary to inform public policy (Tinch et al. 2021). This and the lack of critical discussion about the quality of the original market data are the main issues with direct market valuation observed in these
studies. The availability of market data does not always mean that a method based on market values is the most appropriate for the context in which the valuation is taking place.

4.2.2. Stated Preference

Stated preference methods are used in 49 studies found in this review, with choice experiments being the dominant method. Stated preference has been the source of much controversy – see Hausman (2012); Kling et al. (2012) – yet they remain the best available option for non-use values. Studies used choice modelling and contingent valuation to investigate conservation and recreation values for saltmarsh habitats, which appear to be activities familiar to the respondents. However, a particularly salient issue for stated preference surveys relating to seagrass ecosystems is that public perception of the role of seagrasses is low (Barañano et al. 2022; Losciale et al. 2022), and WTP has been found to correlate with public awareness (Dewsbury et al. 2016). When this is the case, sufficient information needs to be provided to respondents to enable them to form preferences, and it is questionable whether these preferences are reliable when they have low prior knowledge of the ecosystem (Bateman et al. 2008). This may be an issue for some studies found in this review. A choice experiment in South Korea that aimed to investigate public perceptions of blue carbon found that only 5% of the respondents had heard of blue carbon, and only 5.7% of respondents had heard of seagrass previously (Kim et al. 2022). A contingent valuation survey to investigate WTP for conservation in the Philippines found that almost 70% of respondents had no idea what seagrass is (Bundal et al. 2018). Even when respondents are familiar with the site in question, it is unlikely that many will be aware of seagrasses and their role there. The low prior perception of seagrasses may not be an issue once experiments are designed with this in mind (Bateman et al. 2008), but steps taken to address this should be well-documented in studies.

A further issue with using stated preference is that ecosystems found in more affluent areas may be valued higher due to inherent inequalities in ability to pay, and a review of coastal ESV did find that GDP per capita was significantly correlated with the value of the ecosystem (Rao et al. 2015) An example of this is in a choice experiment that addresses coastal restoration in the US, focusing on a nationwide sample and a New Orleans sample (Landry et al. 2011). While 52% of US respondents think coastal restoration is very important and 86% of New Orleans do, a latent class analysis finds that lower income households do not have
statistically significant WTP for coastal restoration while higher income households do (Landry et al. 2011). This is in the context of the aftermath of hurricane Katrina which severely impacted low-income New Orleans households. Similarly to the damage cost avoided method, valuations using choice modelling may provide a stronger rationale for conserving ecosystems in wealthier areas.

Stated preference methods can be resource and time intensive, and the range of coastal and marine services and ecosystems that require valuation are vast. Two studies developed methods to extrapolate values from one choice experiment to a wider context. In an attempt to value multiple coastal ecosystems and avoid the possibility of respondent fatigue and use of heuristics (Johnston et al. 2017), Pascoe et al. used a multi-criteria method called Analytic Hierarchy Process (2019). This method asked respondents to first complete a series of pairwise comparisons between coastal ecosystems, the results of which were used to develop weights for each ecosystem, which were then applied to the results of the choice experiment (Pascoe et al. 2019). Christie and Rayment (2012) also developed a weighting matrix with the help of experts that links ecosystems services and habitats, which was applied to the results of a choice experiment that elicited the public preferences for ES. While initially having a narrow focus, these studies both extrapolate their results to a broader context in ways that may prove useful in addressing concerns about the limitations of current ESV research in the policy context (Olander et al. 2017).

4.2.3. Emergy Analysis

Nine studies attempt to quantify the natural capital value of seagrass or saltmarsh ecosystems through calculating the total energy value of these ecosystems. This approach is based on the work of H.T. Odum, with the idea being that a resource can be valued and compared with another resource on the basis of the solar energy that went into producing it (Odum 1988; Zuidema et al. 2011). The method is known as “emergy analysis” (Odum 1988). It has been described as an eco-centric or donor side estimation of value (Hau and Bakshi 2004), in comparison to ESV which is embedded in an anthropocentric viewpoint. The Emergy Money Ratio, which is the ratio of GDP to total emergy supporting that country’s GDP (in joules) is used to convert values to monetary form (Vassallo et al. 2013).
Costanza et al. (1989) and Gosselink et al. (1974) reference Odum’s work in their valuations of saltmarsh based on the primary productivity of saltmarsh plants, but they do not explicitly name it emergy analysis. They calculate the primary productivity of saltmarsh plants and convert it to dollar values using an emergy conversion factor. They estimate values for coastal Louisiana marsh at $44,624 ha\(^{-1}\) y\(^{-1}\) (Gosselink et al. 1974) and $3,334 ha\(^{-1}\) y\(^{-1}\) (Costanza et al. 1989). More recently, five studies for Italian seagrass ecosystems have used the emergy analysis approach. Among them, Vassallo (2013) estimated annual values of approximately $2.7 million ha\(^{-1}\) y\(^{-1}\) for Posidonia Oceanica seagrass in the Marine Protected Area Isola di Bergeggi in Italy. Environmental flows from the same seagrass species were valued at $2,617 ha\(^{-1}\) y\(^{-1}\), while the standing natural capital was estimated at $9,544 ha\(^{-1}\) in the Islands of Ventotene and S. Stefano in Italy (Franzese et al. 2017). The average value across six Italian sites for the same seagrass species was also calculated at $149,845 ha\(^{-1}\) y\(^{-1}\) (Rigo et al., 2021). The range of values calculated across the same or similar ecosystems using this approach is a cause for concern. The discrepancy between values limits their reliability and applicability in natural capital accounting frameworks and ecosystem management decision-making. Some worry that using figures based on this method to justify ecosystem management decisions could be worse than using no values at all (D. King 1998).

4.2.4. Benefit Transfer

Thirty-two studies were found to use benefit transfer in at least some part of their valuation strategy. Benefit transfer uses information from studies in one location and transfers it to another. This can be useful when there is a lack of data about the initial study site and the budget to conduct primary investigations is limited. For example, large scale assessments of coastal ES were conducted for Mexico and Africa where previous research was limited (Camacho-Valdez et al. 2013; Trégarot et al. 2020). Benefit transfer was used in 43% of the grey literature and only in 12% of the peer-reviewed articles. This is understandable as these grey literature reports often deal with a specific policy site or case where the focus is on producing usable values, rather than publishing related journal articles. An issue noticed by Himes-Cornell et al. (2018) is that benefit transfer is often used without critical discussion as to the validity of the original source, something which is observed in some studies here. Using benefit transfer with stated preference values may be problematic as they can be very context dependent. It has been shown that that
benefit transfers across habitat within a given location perform better than transfers across location for the same habitat (Interis and Petrolia 2016). Some studies used the per hectare value generated by Costanza et al.’s synthesis and applied this to the study area (Blackwell 2006; Han et al. 2008; Zuidema et al. 2011). Often ES provision can vary depending on the region and site specific conditions (Lavery et al. 2013; Nordlund et al. 2016), and global values may not always be appropriate for local application (Himes-Cornell et al. 2018).

On the other hand, benefit transfer is likely to remain an important valuation tool for some time to come, at least due to budget constraints. Studies attempting to refine the approach are necessary. This inclusion of GIS has been important step to refine value estimates (Hynes, Norton, et al. 2013). O’Higgins et al. (2010) and Camacho-Valdez et al. (2013) apply a spatially explicit approach to the north west US and the southern coast of Sinaloa, Mexico respectively. A culturally specific method was tested in Galway Bay to value seagrass and saltmarsh (Hynes, Norton, et al. 2013). In other cases, where the aim of the study was to estimate changes in ES provision due to external factors such as sea-level rise, using benefit transfer to estimate the baseline level of studies was essential to be able to carry out the next phase of work (Feagin et al. 2010).

4.2.5. Climate Regulation Methods

Interest in blue carbon as a tool to mitigate climate change and ensure a transition to a green/blue economy has piqued in recent years (Nellemann and Corcoran 2009), and valuing climate regulation in BCEs is an emerging research field. Most papers studying climate regulation found in this review were published since the year 2010. Thus, the values and methods presented in this review are still characterized by a great deal of uncertainty. For seagrass in particular, the dynamics of blue carbon are not yet fully understood (Unsworth et al. 2022). Sequestration and storage abilities appear to vary depending on water conditions and plant characteristics, so using sequestration estimates from a different species may lead to under or over-estimation of values (Lavery et al. 2013). For both ecosystems, how the mechanisms of sequestration and storage will change in the face of sea level rise, ocean acidification and other climate change related stressors is unclear, and only a few valuation studies have explicitly addressed this (Carr et al. 2018; Feagin et al. 2010; Garrard and Beaumont 2014). As mentioned previously, the true extent of neither saltmarsh nor seagrass ecosystems is confirmed, and state-wide, global, or regional estimates of this service hinge on this uncertainty. Perhaps most
significant for the studies found in this review, there is no consensus over which carbon price to use, as noted in Himes-Cornell et al. (2018). Unless the valuation is for a policy context that calls for a specific price, where possible a range of estimates for this service should be used to reflect this uncertainty. Finally, valuing the climate regulation service should not be at the expense of understanding other services – especially given the inherent uncertainty in these values. Climate regulation is a valuable service and blue carbon may be an important funding mechanism to conserve these habitats, but BCEs are more than just carbon sinks.

4.3. Ecosystem Services

As shown in Table 6, nineteen saltmarsh ES were valued in the literature. The most frequently studied services are characteristics that enable active recreation, flood control and coastal protection, and climate regulation. Eleven services have less than four studies associated with them. For seagrass, 15 services have been studied, and non-consumptive cultural services are the least frequently studied. The variety of ES studied is not uniform across global regions, however the most frequently studied services in each region for both ecosystems are climate regulation, food from wild animals, and active recreation. Asia is the only region where the most frequently studied service for saltmarsh is coastal protection. The intense focus on climate regulation is understandable given the calls for increased research on climate related impacts and the growing interest in the role of blue carbon habitats in climate change mitigation (Duarte 2017; Nellemann and Corcoran 2009). The number of valuations focusing on the contribution of these habitats to commercial fisheries is facilitated by the availability of market data. Habitat-fishery linkages seem to be relatively well understood, and this understanding likely contributes to the number of valuations on active recreation, which commonly focuses on the role of seagrass and saltmarsh in maintaining recreational fisheries.

Agriculture can benefit from saltmarsh as grazing land as an input into the livestock production function (McKinley et al. 2022), and in some cases this is exploited with a price premium, such as saltmarsh-fed lamb in France (Gedan et al. 2009). The value of saltmarsh to agriculture and the impact improperly managed grazing can have on ecosystem functioning are likely to be important areas of research in terms of agricultural and land-use policy, especially with the growing emphasis on conserving BCEs (Muenzel and Martino 2018). However, the value of
saltmarsh to agriculture through providing grazing land has only addressed by three studies, (Boerema et al. 2016; Posford Duvivier 1996; S.E. King and Lester 1995), two of which may now be quite dated. Understanding the value of this service will be important in policy-making decisions around land use, so that the benefits and costs of different uses of saltmarsh can be fully compared.

While North America and Europe are the sites of over 60% of the saltmarsh valuations relating to active recreation, non-consumptive cultural services remain poorly quantified in both regions, despite a strong history of non-market valuation. Cultural services are perhaps the most complex ES to value, yet there seems to be some avenues that have not been fully exploited in terms of saltmarsh ESV. Birdwatching is an established passive recreational activity and has been valued on saltmarshes in Australia (Carnell et al. 2019), yet not in any other locations. Saltmarshes are important sites for migratory wildfowl; in Ireland for example, the North Bull Island saltmarsh in Co. Dublin was the first designated bird sanctuary in the country (Lewis 2019). As an activity it should prove amenable to established methods of valuation such as travel cost and choice modelling and could be an avenue for future research.

The importance of saltmarsh in terms of culture, heritage and existence value is something that has also been poorly addressed by valuations to date. This may be a hangover related to the traditional dominant view of saltmarshes (and wetlands in general) as wastelands (Barbier et al. 1997; Gedan et al. 2009), yet effort should be made to understand this value. A Welsh study found that saltmarsh owners were aware of the economic and ecological benefits related to saltmarshes, but also expressed a deep personal connection to the saltmarshes, with one interviewee describing saltmarsh as a “magical sort of piece [of land]”, illustrating the possible cultural, heritage and spiritual values this land may have for them (McKinley et al. 2022). These values are frequently disregarded in the decision-making process, yet an increased effort should be made to understand them as they may have a significant impact on human well-being.

The link between non-consumptive cultural activities and seagrass is also poorly quantified. Dewsbury et al. (2016) suggested that as activities such as scuba-diving, snorkeling, and boating are popular in areas with healthy coral reefs, the relationship between seagrass beds and coral reefs should be analyzed to estimate the contribution of seagrass to of these activities. Although WTP among divers to visit a seagrass site was estimated by Kenter et al. (2013), this was not the main
focus of their study. Linking non-consumptive recreational activities and seagrasses is an area where more research is needed. There are other links to cultural services that are poorly understood. The historical use of seagrass fibers as a raw material - documented by Curtis (2003), and Wyllie-Echeverria and Cox (1999) - indicates it may have significance for heritage in areas where this practice used to be the norm. Spiritual and cultural values are likely to be very important for indigenous communities, but quantifying their economic value is extremely difficult (Sangha et al. 2019). While Sangha et al. (2019) estimate a general cultural value for marine and coastal ecosystems, it cannot be linked specifically to either of our ecosystems of interest. Similarly, Trégarot et al. (2020a) estimate a heritage value based on the relationship between the indigenous community of the National Park of Banc d’Arguin, Mauritania and their local fishery, which depends on the ecosystems of the park. Again, linking this value to one particular ecosystem is problematic, as the fishery is maintained by several interlinking services and ecosystems. The inherent complexity of valuing cultural ES for seagrass should not mean that it is disregarded. As public awareness of the role of seagrass grows it will be interesting to see how WTP for conservation of these ecosystems’ changes in future research.

Future avenues for research could build on the non-market valuations for aquatic and coastal ecosystems. It has been shown that the people are willing to pay to improve water quality (Doherty et al. 2014), and recreationalists have preferences for improving the quality status of bathing water (Hynes, Tinch, et al. 2013). The relationship between the seagrass and saltmarsh and water quality, through their filtration and bio-remediation services, indicates non-market valuation relating to water quality may be a means to value these services. Another pathway to valuation relates to the fact that extreme weather events on our coasts are now a regular occurrence, and it’s predicted that by 2030 approximately 11% of the global population will be living in low-elevation coastal zones² and at risk of flooding (Neumann et al. 2015). The coastal defense and flood protection services of saltmarsh and seagrass, given the high concentration of coastal activity and the increased climate pressures, may be significant but this service remains undervalued for seagrasses. Finally, coastal communities have been found to have a respect for the marine environment and an outlook that champions the

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² Low-elevation coastal zones (LCEZ) are commonly defined as areas along the coast that below 10 m of elevation and are contiguous and hydrologically connected (Neumann et al., 2015).
conservation of marine ecosystems (Farrell et al. 2017), and future valuation of the associated cultural services should account for these perspectives.

5. CONCLUSION

ESV of saltmarsh and seagrass ecosystems began in the 1970s and 80s respectively. In total, 162 studies were identified in this review that value these ecosystems. While saltmarsh valuation studies have been increasing steadily, the last 10 years has seen a comparatively large increase in the number of valuation studies for seagrass, possibly driven by the recent recognition of their importance as a blue carbon habitat. Research is spread unevenly across the globe, with hardly any valuation studies found in South America. Saltmarsh valuation studies have been concentrated in the US and the UK, and Australia is a hotspot for seagrass studies. Studies valuing climate regulation – the blue carbon aspect – have grown in recent years with 41 seagrass studies and 26 saltmarsh studies valuing this service.

The value of carbon sequestration in seagrasses ranges from $8 ha^{-1} y^{-1}$ in the Northern Territories of Australia, (Sangha et al., 2019) to $1,146 ha^{-1} y^{-1}$ in the Datça-Bozburun peninsulas in Turkey (Bann and Başak 2013). The range for saltmarsh is similar, from $3 ha^{-1} y^{-1}$ in Bhitharkanika, India (Dhivya et al. 2023), to $1,540 ha^{-1}$ in the Southeast Coast of the UK (S.C.L. Watson et al. 2022). These large ranges stem from differences in sequestration rates and the carbon price used. The reliability of these estimates depends also on whether field samples or literature reviews have been used to inform sequestration and storage figures.

Apart from climate regulation, the most frequently studied services for seagrass were the contribution to commercial fisheries and maintaining nursery habitats, services which are clearly linked. Direct market valuation of commercial fisheries using market prices for fish was the main method used to value this service. The prices were adjusted to reflect the contribution of seagrass to the fishery through different methods. Values calculated using market prices for this service ranged from $14 ha^{-1} y^{-1}$ (Jackson et al. 2015) to $742 ha^{-1} y^{-1}$ (Unsworth et al. 2010). Maintaining nursery habitats was valued using market prices as well, but the prices were adjusted to reflect the importance of the nursery habitat specifically. This service appeared to be more valuable, as estimates ranged from $140 ha^{-1} y^{-1}$ (Erzini et al. 2022) to $22,788 ha^{-1} y^{-1}$ (Blandon and zu Ermgassen 2014). While it is understood that the link between seagrass and fisheries is significant on a global
level, ecological data on the abundance and distribution of fish in seagrass meadows is still lacking in some regions to be able to clearly quantify the value of this service (Nordlund et al. 2018).

The most frequently studied services for saltmarsh were active recreation and coastal protection. Coastal protection was valued using cost methods and stated preference methods. The values of coastal protection estimated using replacement costs ranged from $1,257 ha\(^{-1}\) (Sun et al. 2017) to $772,004 ha\(^{-1}\) (S.E. King and Lester 1995). Annual per hectare values estimated using damage cost avoided methods were generally lower – avoided flood damages through the presence of saltmarshes in the Scheide estuary were estimated at $9,872 ha\(^{-1}\) y\(^{-1}\) (Boerema et al. 2016). Updating these figures with primary data would be useful, as it was estimated using benefit transfer methods originally.

Active recreation was estimated using travel cost methods and contingent valuation, and the activities valued included recreational fishing, birdwatching, and tourist visits to saltmarsh sites. These valuation methods have been used extensively to value recreation at terrestrial sites, but similar to many other marine ecosystems they have seen only limited use for recreation taking place in and around saltmarshes and especially seagrasses. Given the importance of saltmarshes to birdwatchers, and the existence of strong birdwatching organizations in many countries, surveying these organization members could be a means to value these blue carbon sites. In general, non-consumptive cultural services are not often valued in the literature, especially for seagrass (Dewsbury et al. 2016).

Despite the traditional perception of wetlands as wastelands (Barbier et al. 1997), there is a growing appreciation and an associated body of literature on the value of saltmarshes and seagrasses. Valuation efforts could be enhanced further by documenting the ongoing changes in the condition of the blue ecosystem area being valued and by expanding the range of cultural services being studied. Global efforts to develop a coherent system of natural capital accounting as part of the UN SEEA framework, with associated ecosystem extent, condition and service accounts, should assist in this regard. Improving the extent and condition of these ecosystems will have economic costs, and it will be important from a policy perspective to have reliable estimates on the ecosystem service benefits from any restoration/conservation to compare to those costs. Information on these benefit values is also important in terms of the development of practical solutions for the financing of blue carbon ecosystem projects.
Table 9: Saltmarsh and seagrass services identified and valued in the literature, by method

<table>
<thead>
<tr>
<th>Services</th>
<th>CICES Code</th>
<th>Saltmarsh</th>
<th>Seagrass</th>
<th>MP</th>
<th>DC</th>
<th>RC</th>
<th>TC</th>
<th>PF</th>
<th>HM</th>
<th>CV</th>
<th>CM</th>
<th>BT</th>
<th>RA</th>
<th>C</th>
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<tbody>
<tr>
<td>Food grown by humans</td>
<td>1.1.1.1</td>
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<td>1</td>
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<tr>
<td>Reared animals</td>
<td>1.1.3.1</td>
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<td>✓</td>
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<tr>
<td>Materials from wild plants</td>
<td>1.1.5.2</td>
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<td>✓</td>
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<td></td>
<td></td>
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<td></td>
<td></td>
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<td>1</td>
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<tr>
<td>Food from wild animals</td>
<td>1.1.6.1</td>
<td>✓ ✓ ✓ ✓</td>
<td></td>
<td>10</td>
<td>21</td>
<td>1</td>
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<tr>
<td>Decomposing wastes</td>
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<td>✓ ✓</td>
<td></td>
<td>1</td>
<td>1</td>
<td>3</td>
<td>1</td>
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<tr>
<td>Filtering wastes</td>
<td>2.1.1.2</td>
<td>✓ ✓</td>
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<td>1</td>
<td>5</td>
<td>4</td>
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<tr>
<td>Controlling erosion rates</td>
<td>2.2.1.1</td>
<td>✓</td>
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<td>2</td>
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<tr>
<td>Regulating flows of water</td>
<td>2.2.1.3</td>
<td>✓ ✓ ✓ ✓</td>
<td></td>
<td>5</td>
<td>1</td>
<td>4</td>
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<tr>
<td>Pollution</td>
<td>2.2.2.1</td>
<td>✓</td>
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<tr>
<td>Providing habitats</td>
<td>2.2.2.3</td>
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<td></td>
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<td>6</td>
<td>1</td>
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<tr>
<td>Disease control</td>
<td>2.2.3.2</td>
<td>✓</td>
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<tr>
<td>Seawater quality</td>
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<tr>
<td>Climate regulation</td>
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<tr>
<td>Active recreation</td>
<td>3.1.1.1</td>
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<td>2</td>
<td>1</td>
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<td>5</td>
<td>2</td>
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<td>Passive recreation</td>
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<td>Researching nature</td>
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<td>4</td>
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<tr>
<td>Cultural and heritage value</td>
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<td>Aesthetic value</td>
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<td>✓</td>
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<tr>
<td>Existence value</td>
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<td>3</td>
</tr>
</tbody>
</table>

Note: ✓ indicates less than 5 primary valuations found in the literature, ✓ ✓ indicates 5-10 primary valuations found in the literature, ✓ ✓ ✓ indicated over 10 valuations found in the literature. MP=adjusted market prices, DC=damage cost avoided, RC=replacement costs, TC=travel costs, PF=production function, HM=hedonic modelling, CV=contingent valuation, CM=choice modelling, BT=benefit transfer, RA=rapid ecosystem assessment, C=carbon calculation methods. Dark grey indicates method used for saltmarsh, light grey indicates method used for seagrass. Numbers refer to numbers of studies.
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