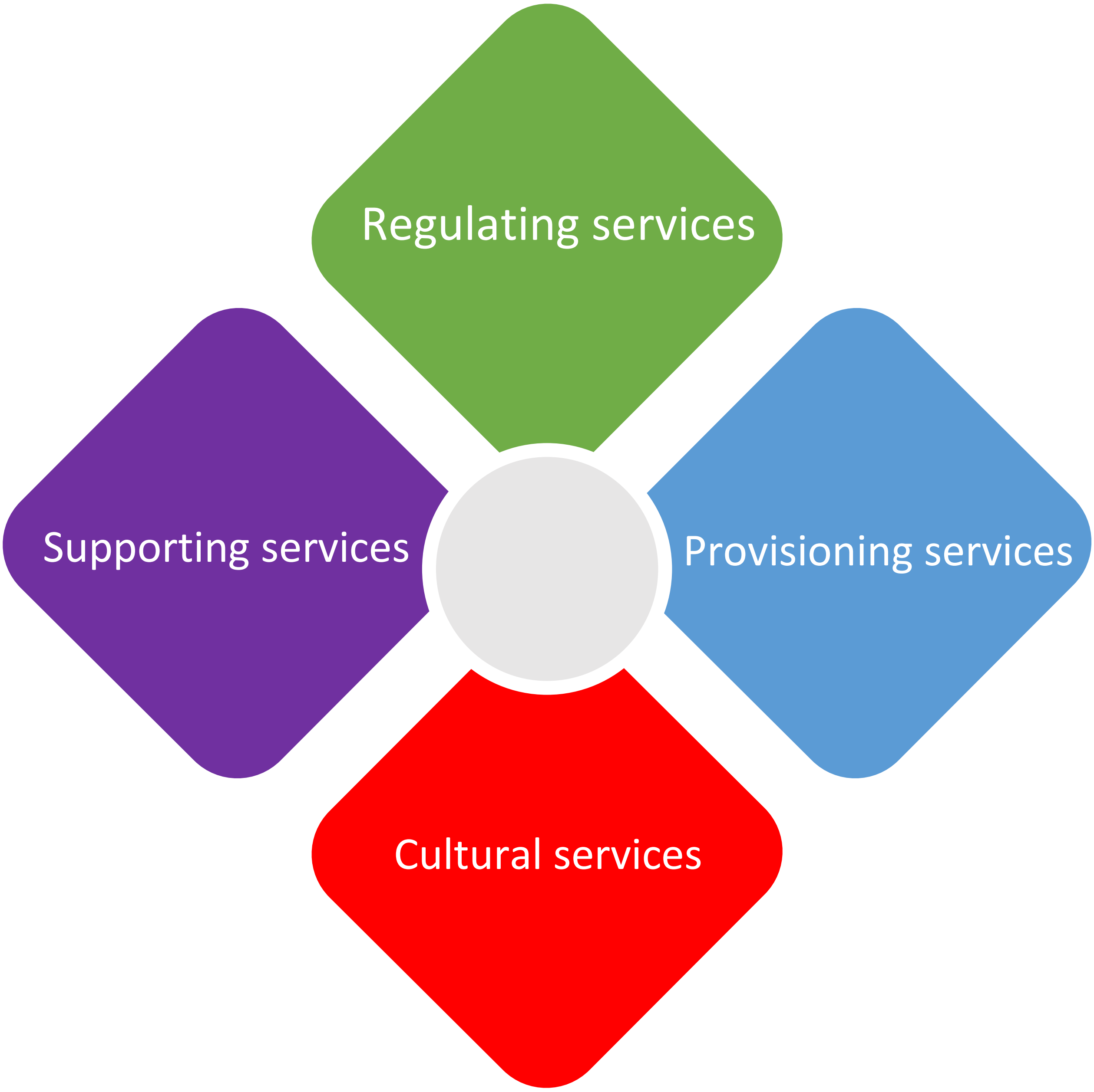
A non-monetary scale for the evaluation of tropical seagrass ecosystem services in the Indo-Pacific through meta-analysis



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Cover figure. Visual display for the scores provided by the scoring system presented here. Each of the four categories from the Millennium ecosystem assessment: regulating, Supporting, Provisioning and Cultural (MA, 2005), are scored out of 10. The central circle presents the overall score out of 40.

Abstract

Ecosystem service valuation is the process by which naturally occurring services, like carbon storage, can be valued, most commonly in the form of financial valuation. However, monetary valuation can undervalue services that do not provide much economic gain but are still biologically or culturally important. Ecosystem services are generally classified into four categories Cultural, Regulation, Supporting and Provisioning services. This study aims to create a non-monetary scoring system for the valuation of ecosystem services. To do this, I will be using seagrass in the Indo-Pacific region as a case study. Non-monetary valuation is a method by which a numeric value can be assigned to these services, independent of financial value, allowing factors such as fisheries productivity, a provisioning service, to be compared with cultural, supporting and regulating benefits independent of money. These non-monetary systems have previously been vague or difficult to apply for decision-makers. Using Indo-Pacific seagrass meadows as a case study, I present a non-monetary data-driven valuation system that eliminates some of the bias found in other monetary systems and provides a clear points based score for the ecosystem. Literature searching returned 31 papers assessing organic carbon storage covering 68 meadows in 12 countries and one territory, and the sampling depth ranged from 3-140cm. Two were found with BRUV (Baited Remote Underwater Video) data, which were combined with multiple unpublished sources provided by Swansea University to cover 357 drops across seven countries and one territory in the Indo-Pacific. Carbon storage is a key ecosystem service in the prevention of climate change, organic carbon was chosen as it shows the accumulation of organic material by the meadow which contributes to carbon sequestering. Additionally, data for inorganic carbon were less available. Cultural services where assessed using the presence of rare and endangered species as a proxy. Fisheries data were selected due to the high level of reliance on fish for protein in this region. This valuation system shows that ecosystem services in the Indo-Pacific vary by environment type, particularly lagoon and reef meadows. Lagoon meadows show significantly higher carbon storage than coastal, deepwater, reef and estuarine. Conversely, reef meadows show significantly higher scores for fish assemblage associated data than coastal or lagoon meadows. This paper develops a consistently reliable method through meta-analysis for the non-monetary valuation of ecosystem services using a percentile scale. This allows decision-makers to consider non-monetary factors in management situations, therefore considering the community opinion and the financial benefits of conserving an ecosystem. However, this is not the focus of this study, which was to create a scoring system that can be used in non-monetary valuation.

Keywords

Ecosystem services, Seagrass, Indo-Pacific, valuation

Glossary

Seagrass- A marine flowering plant typically found in meadows

Ecosystem service- Benefits provided to humanity by an ecosystem.

Coring- The process by which carbon storage is estimated by taking a tube shaped sample of sediment.

BRUV-Baited remote underwater video

Umbrella species- Species which when protected, can confer protection to others

Lay Abstract

Biological systems can be assigned financial value through a process known as valuation. Valuation allows their benefits to be compared to other financial systems, which can help with conservation and management. However, these systems can often overlook benefits that do not make a lot of money, such as mental health benefits or gathering of food for consumption not selling. Valuation is difficult as often because these benefits are not sold, and so none or very little money is involved. Despite this, these benefits are still highly important to those who use them. Alternative systems have tried to apply scores to these benefits without the involvement of money. These systems have often been vague or difficult to use. Using Indo-Pacific seagrass as an example, here a new scoring system has been designed independent of money to assign value to these benefits and allow comparison between sites. Seagrass is a marine flowering plant that is found around the coasts worldwide and typically forms meadows. These meadows support a large amount of marine life and are considered to be large carbon sinks storing between 29.1-129.04 Mg/Ha. Carbon sinks are areas where more carbon is taken out of the atmosphere than put in, and therefore they counter carbon emissions. The carbon seagrass absorbs from the atmosphere can then become buried in the sediment below the meadow as plant material or particulates and is then referred to as stored carbon. Seagrass meadows also support large and diverse fish assemblages, which make them important fishery grounds. Similarly, seagrass meadows are visited by megafauna such as sharks, rays and turtles primarily for the purpose of feeding on these fish assemblages or the seagrass itself in the case of the turtles. As a result, seagrass can be a critical habitat in the conservation of these species. The scoring system assigns scores 0-10 to these benefits according to how well the meadow provides them in comparison to other meadows in the same type of area as themselves, i.e. reef meadows are compared with other reef meadows. These scores are assigned in comparison to the data collected in this study. For example, if a reef meadows carbon storage was in the highest 2.5% of all reef meadows, it would score 10 on the score card. Conversely, if the meadow was in the lowest 2.5%, it would score 0. Anything in between would score 1-9. The way these scores combine into a final score is shown in Figure I. Scoring in this way stops extreme values from preventing meaningful comparisons between meadows which are average for their environment, be that reef lagoon or coastal. Allowing decision-makers to consider the relative benefits of services with high financial value against those which do not directly generate money but are key services to local peoples such as the presence of endangered species on a level playing field.

To create a database to use as a case study, studies were gathered into these benefits from across the Indo-Pacific and combined their data to create this scoring system. The data were from 31 papers assessing organic carbon storage covering 68 meadows in 12 countries and 1 territory, the sampling depth ranged from 3-140cm. Two with BRUV (Baited Remote Underwater Video) papers combined were also used combined with multiple unpublished sources provided by Swansea University to cover 357 drops across 7 countries and 1 territory in the Indo-Pacific. The data gathered showed that the benefits provided by these seagrasses varied with their surroundings. Most notably, seagrass in lagoons stored more carbon than other areas on average 129.04Mg/ha as opposed to meadows with means of 69.29Mg/ha, 64.45Mg/ha, 32.65Mg/ha and 29.1Mg/ha for coastal, reef, estuary and deep water meadows, respectively. Seagrasses near reefs had: higher numbers of commercial species with a mean of 9.62 individuals present, almost double that of coastal meadows at 4.68 individuals, roughly ten times that of lagoon meadows which had a mean of 0.97 individuals. A similar

trend was seen in food fish species, larger varieties of fish, higher numbers of predators and more endangered species. Data were limited for the Indo-Pacific. However, the scoring system proposed could be applied with additional data incorporated or applied to data on other ecosystems.

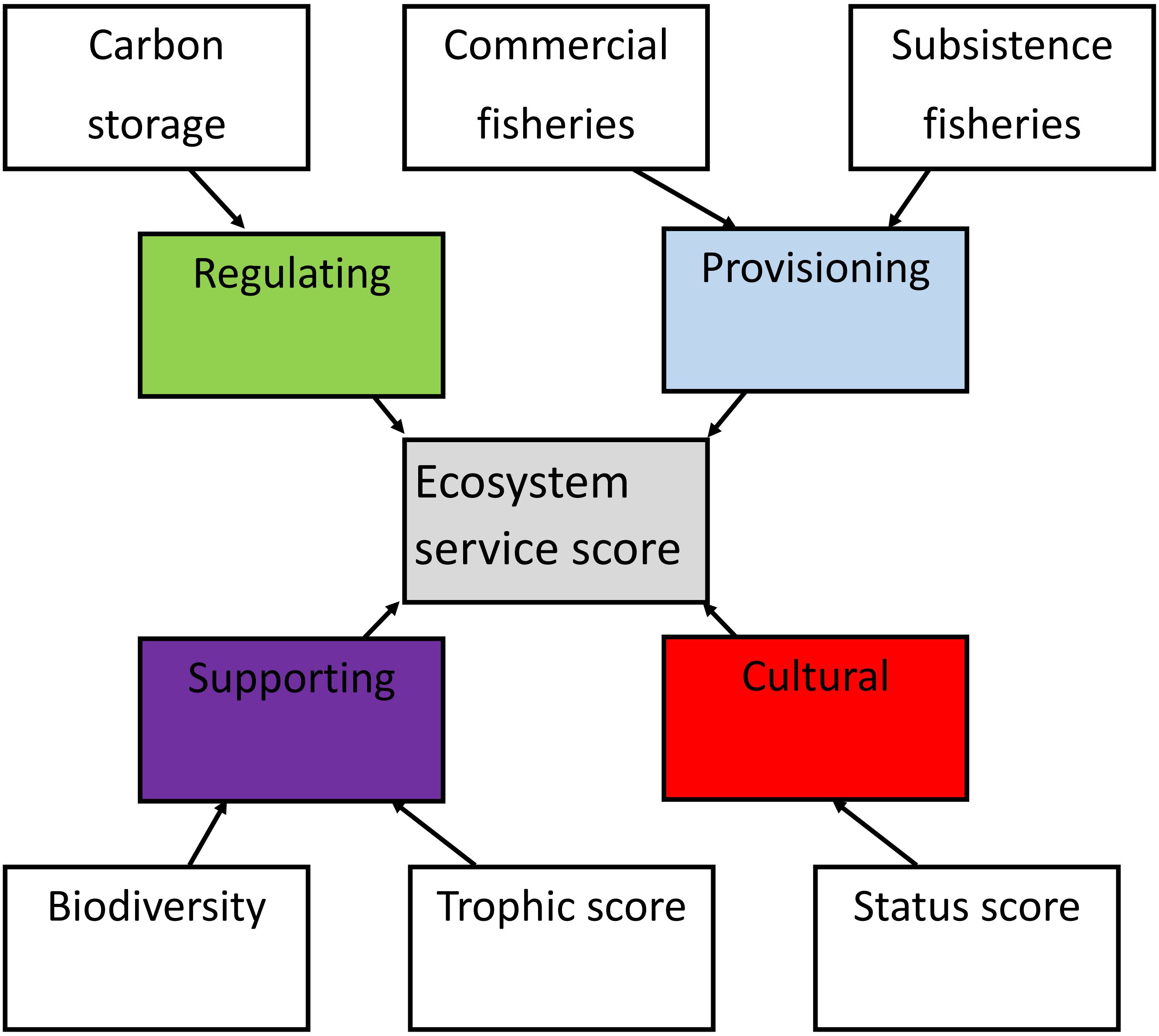

Figure I. Six scoring metrics were included in the scoring system. Each metric was assigned a numerical value between 0 and 10 according to how it compared to other meadows for this metric. I.e. if it had exceptionally high levels of commercial species present, this would score 10, but if there was incredibly little to none, then it would score 0. The scoring parameters then are averaged within their categories to give a score for the category, i.e. subsistence fisheries and commercial fisheries feed into provisioning services. These category scores are then added together to create an overall score for the meadows ecosystem services.

Declarations

This work has not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree.

Signed.... [redacted] ....

Date.......... 24/01/22 ....

This thesis is the result of my own investigations, except where otherwise stated. Other sources are acknowledged by footnotes giving explicit references. A bibliography is appended.

Signed.... [redacted] ....

Date.......... 24/01/22 ....

I hereby give consent for my thesis, if accepted, to be available for photocopying and for inter-library loan, and for the title and summary to be made available to outside organisa[redacted]

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Date.......... 24/01/22 ....

The University's ethical procedures have been followed and, where appropriate, that ethical approval has been granted.

Signed... [redacted] ....

Date.......... 24/01/22 ....

Projects Ethics Assessment Status

Project Title	Status	Approval Number
A non-monetary scale for the evaluation of tropical seagrass ecosystem services in the Indo-Pacific through meta-analysis	Completed	SU-Ethics-Student-260321/2468

Statement of expenditure

Category	cost
Incidental costs/materials	£0
Conferences	£0
Relevant training courses	£0
Software, data storage/sharing	£0
Equipment or consumables	£0
Fieldwork	£0
Other	£0
Total	£0

Statement of Contribution

Contributor Role	Contributors
Conceptualization	NH, RU
Data Curation	NH
Formal Analysis	NH
Funding Acquisition	N/A
Investigation	NH, RU
Methodology	NH
Project Administration	NH
Resources	RU
Software	NH
Supervision	RU, PN
Validation	NH
Visualization	NH
Writing – Original Draft Preparation	NH
Writing – Review & Editing	NH,RU,PN

NH-Nicholas Harman RU-DR Richard Unsworth PN-DR Penny Neyland

Effect of Covid-19

Covid-19 has had a significant effect on my thesis. Principally having to transition from a fieldwork-based project to a desk-based project in March 2020. In addition to this communication with my supervisor became difficult and mental health suffered from lockdowns

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1. Introduction

1.a Ecosystem service valuation

Ecosystem services are defined as the benefits provided to humanity by ecological processes and ecosystem functions (MA, 2005). For example, carbon storage, provided by a forest, or seaweed, which can be harvested from a coastline for food. Many ecosystem services are critical for the welfare and survival of humanity such as the production of oxygen and food. These services can be assigned value typically in the form of money (Barbier 2019, Wilkinson et al. 1999). This valuation can help management decisions on how resources will be used comparing the relative values of the ecosystems (Rizal & Dewanti, 2017).

One school of valuation is the capital approach. This is the idea that all systems can be separated into three different types of capital: Human, manufactured and natural (Barbier 2019). Human capital is the experience and skills of a workforce (Barbier 2019). Manufactured capital, sometimes referred to as reproducible capital, generally refers to things constructed or produced by humans, such as buildings and roads (Barbier 2019). Natural capital is an economic principle based on the idea of ecosystem valuation using money. The value of Natural capital can be either the value of good provided (Barbier 2019, Wolloch 2020). Monetary valuation techniques can be useful and logical when applied to things such as timber, a traded commodity, as the market price of these commodities reflects their rarity (Lienhoop et al., 2015). When valuing processes such as water filtration, monetary valuation is often used to justify the cost of conservation measures (Carpenter et al., 2009).

Early studies into ecosystem valuation initially focused on a particular species or habitat (Nunes & van de Burgh, 2001); most recent studies, however, take the approach of valuing ecosystem services. For example, Wilkinson *et al*. (1999) valued the coastal protection at US\$174$ha^{-1}yr^{-1,}$ and Rizal and Dewanti (2017) valued fish diversity in the Sikakap strait at US\$745,602. This method allows decision-makers to use a commonly comparable unit (money) to inform decisions and to easily compare with costs (Balmford et al., 2002).

Ecosystem services can be broadly categorised into four categories provisioning, regulatory, supporting and cultural (MA, 2005). Provisioning services are services which provide physical resources such as timber for construction or fish as a food source (MA, 2005). Regulatory services regulate the environment and include flood defence, water quality and carbon storage (MA, 2005). Supporting services are processes such as nutrient cycling, which allow other services to be provided (MA, 2005). Cultural services are less tangible benefits and generally refer to religious, leisure and wellbeing services such as the mental health benefits of spending time in nature (MA, 2005).

37% of the global population in 2017 could be found in coastal communities (UN, 2017). It is estimated that the global population will increase by 2 billion people by 2050, 77% of which will occur in Sub-Saharan Africa (52%) as well as Central and South Asia (25%)(UN, 2019). Both of these regions have coastal areas within the Indo-pacific. As populations increase, we are ever more reliant on ecosystem services, which were defined by Daily (1997) as "the conditions and processes through which ecosystems sustain and enrich human life", ranging from oxygen and food production to recreational and spiritual benefits (MA, 2005; TEEB, 2010). Ecosystem services are a useful concept for valuation (Gómez-Baggethun and Barton, 2013) as they conveniently partition humanities needs into quantifiable metrics to which values can be assigned. When investigating ecosystem services, sociocultural values are an important aspect to consider (De Groot et al., 2002) as while they do not always provide high financial gain, they can have significant positive impacts on local peoples. The concept of these values has been around since the early 2000s (De Groot et al., 2002), and valuation studies show inconsistency in methodologies (Clifton et al., 2014), as demonstrated by Porto et al. (2020), who found large variations in values found for crop pollination varied between US$195 billion-US$387 billion due to differing methodologies. Many non-monetary systems employ imprecise methods for valuation (Seppelt et al., 2011). Imprecise scoring is also an issue in monetary valuation and can result in large ranges of values varying by orders of magnitude (Clifton et al., 2014). These kinds of valuation techniques can be difficult to implement (Kelemen et al., 2014). For example, Nuttle (2013) used a scoring of 1-5; each number score referred to the state as 'poor' or 'good' or 'very good', which can be interpreted differently by observers with differing experience levels. Alternatively, some systems use percentage boundaries, for example, Kam et al. (2013) used four categories: 0-25, 25-50, 50-75 and 75-100. Kam's method is less variable than Nuttle's system with observers, but including only four categories may miss variation within a biological system. Valuation of ecosystem services is growing as an important method for conservation (TEEB, 2011), allowing meaningful comparison between different sites or systems.

A non-monetary valuation can be used as an alternate aid to decision-making processes (Kelemen et al., 2014). While economic valuation can be useful in decision-making processes, it can overlook or fail to incorporate factors such as ecological or cultural significance. These may not have a high financial value but are valued by the local people (Christie et al., 2012). For example, small scale fishers, which represent 90% of fishers worldwide (FAO, 2010) and up to 95% of which are for human sustenance (The World Bank, 2012). These can be crucial for social benefits like spiritual and religious values (UNEP 1999). Despite cultural services being broadly recognised as important, they are often not incorporated into aids for decision-makers (de Groot et al., 2002, MA 2005). However, the importance of sociocultural values has gained momentum recently (Pascual et al., 2017). Equally, other more quantifiable services can also be difficult to value in a monetary sense, i.e. nutrient cycling (Christie et al., 2012; Farber et al., 2002). Non-monetary valuation is undertaken by assessing humanity's needs and desires of nature and creating ways to compare these services separated from financial benefits (Chan et al., 2012). Non-monetary methods are variable and may include input from local peoples. For example, Unsworth et al. (2014) used interviews to ascertain the prevalence of seagrass fisheries in the diets of local peoples, and Videira et al. (2009) used participatory modelling to create a simulation model to explore differing management options with a group of stakeholders.

Some studies have looked into mixed-valuation methods incorporating monetary and non-monetary valuation into one system (Kronen 2007, Daams et al., 2019). Kronen *et al.*, (2007) used small-scale fisheries in the Indo-Pacific as a case study. They found that the price of fish appeared to vary with cultural values disproportionately to what might be expected. However, the unit of valuation was still monetary but influenced by non-monetary factors while this may be a more accurate valuation system on small scale it would be hard to implement over large scales.

1.b Seagrass and their ecosystem services

Seagrasses are marine flowering plants (angiosperms) (Larkum et al., 2006) and are found on all continents except Antarctica (Pauly and Zeller, 2016). They typically form meadows (Short et al., 2007) and are easily accessible as they are normally found near the coast (De la Torre-Castro et al., 2014; De La Torre-Castro and Rönnbäck, 2004). Seagrass provide many ecosystem services, including carbon storage, acting as nurseries for commercially important groups such as *Mugilidae* (mullet), *Gadus* (cod), *Lutjanidae* (snapper) and *Clupeidae* (herring) (Lilley and Unsworth, 2014; Nordlund et al., 2018) and provide fishing grounds for coastal communities (Nordlund et al., 2018). Seagrass meadows are a large marine sink for carbon (Duarte et al., 2005; Fourqurean et al., 2012) and bury just under 10% of oceanic carbon (Blue carbon) at a rate of 27.4Tg of carbon per year (Duarte et al., 2005). Reducing the impact of carbon emissions (Lal et al., 2015) which are increasing (Chen et al., 2018; Yasmeen and Subhadra, 2019). Seagrass receives relatively little attention compared to other marine systems like coral reefs and mangroves, and is often overlooked in coastal management (Duarte et al., 2008; Grech et al., 2012; Nordlund et al., 2014).

The Indo-Pacific is the largest seagrass bioregion, spanning from the East coast of Africa and the Middle East throughout South East Asia to Australia, and contains the most diverse assemblage of seagrass on the planet (Short et al., 2007), comprising 24 species (Nordlund et al., 2018). Seagrass in the Indo-Pacific is estimated to cover 74,579.39 $km^{2,}$ roughly 44% of the global seagrass per area and equates to roughly 50% of the Indo-Pacific coastline by area (McKenzie et al., 2020). Indo-Pacific seagrass meadows play an important role in providing food security; coastal regions of the Indo-Pacific rely heavily on marine ecosystems for protein (Unsworth and Cullen, 2010). This may be attributed to the easily accessible nature of seagrass meadows, typically located near the shore in shallow waters (De la Torre-Castro et al., 2014; De La Torre-Castro and Rönnbäck, 2004). However, as subsistence fishing is not commercially significant, it can be hard to value, as little to no money is exchanged (Worm et al., 2009). Subsistence fisheries, however, are a priceless resource for the communities that rely on the meadows as their primary source of protein, accounting for 20% of total fisheries catch worldwide (Pauly and Zeller, 2016).

Access to seagrass meadows can be essential to the welfare of coastal communities (Cullen-Unsworth et al., 2014), but since 2004 the average catch weights of these fishers have decreased by 51% (Unsworth et al., 2014). Reduction in catch weights is particularly concerning as there is a high dependence on seafood in the tropics for protein (Donner & Potere, 2007). Furthermore, Waycott et al. (2009) found that 30% of global seagrasses have been lost since the industrial revolution, resulting in a loss of ecosystem services. Implying a need for more effective protection and management informed by research (Unsworth et al., 2015). The level of ecosystem service provision varies between different seagrasses and the environments they are found in, the extent of which is not fully understood (Lavery et al., 2013; Nordlund et al., 2016). It is also thought that meadows consisting of a single species may provide differing levels of services to mixed-species meadows (Duarte, 2000); for example, a mixed-species meadow may potentially support a larger variety of fish than a single species meadow. The geographical location of a meadow can also affect the services provided (Nordlund et al., 2018). For example, fish assemblages in marine habitats vary depending on their surroundings (Nagelkerken, 2009). Usage of the seascape can also vary with lifecycle stages, diurnal, lunar and seasonal cycles (Unsworth et al., 2015). Connecting tropical habitats into an interconnected system, each benefitting from each other (Nagelkerken, 2009), and so they should not be considered self-contained units.

Nordlund et al. (2016) stated that seagrass can provide 25 ecosystem services to humanity. Of these services, I have data on food provision (commercial and subsistence species), carbon storage and cultural benefit/tourism. As well as these direct benefits, I have data on diversity and trophic scores, which support the provision of other services.

1.c Scoring metrics

Apex predators such as sharks (Osgood et al., 2020) are commonly used as indicators of ecosystem health, particularly in marine systems. An idea that has come under scrutiny for the implication that conserving top predators conserves the ecosystem as a whole (Simberloff, 1998). Some studies have found a weak relationship between the conservation of top predators and overall diversity (Sergio et al., 2008). However, these studies focused on mammalian carnivores, so they may not hold the same relevance for sharks (Sergio et al., 2008). A recent review of the literature by Lindenmayer and Westgate (2020) was inconclusive as to whether umbrella species as a whole are a useful concept. However, a study in South Africa has shown that some sharks show high potential as umbrella species (Osgood et al., 2020). Trophic scoring considers all the species present and their relative position in the food chain and, therefore, may be a more reliable indicator of ecosystem health than umbrella species. Trophic scoring will be used along with diversity as a metric to show the resilience of a system as this has been linked closely with the interactions between predators and prey in fisheries (Dunn et al., 2017). It was also suggested by Worm and Duffy (2003) that linking food-web interactions and diversity may be a useful tool for conservation which is achieved by the trophic score system.

In addition to developing a scoring system, I will also look to incorporate the cultural values of species to the local communities. Furthermore, I will investigate food security through the presence of commercially important groups: *Carangidae* (Jacks), *Elasmosbrachii* (sharks and the rays)*, Clupeidae* (Herring and Sardines), *Lethrinidae* (Emperors), *Nemipteridae* (Threadfin bream), *Scaridae* (Parrotfish), and *Siganidae* (Rabbitfish) (SeaAroundUs, 2020). As well as the groups that are important for subsistence fisheries: (Gerreidae (Mojarra), Labridae (Wrasses), Lethrinidae (Emperors), Lutjanidae (Snapper), Mullidae (Goatfishes), Nemipteridae (Threadfin bream), *Scaridae* (Parrotfish) and Siganidae (Rabbitfish) (Unsworth et al. 2014, De la Torre-Castro et al. 2014, Exton et al. 2019, De la Torre-Castro and Ronnback 2004).

1.d Aims

This study aims to create a non-monetary scale to compare ecosystem services between seagrass meadows as a means of communicating the value of all services to decision-makers and stakeholders. This scale can be used to help management decisions consider not only the economic benefits of conserving seagrass meadows but also the needs of local communities, many of whom rely on the seagrass meadows for subsistence, or to whom those meadows may hold cultural significance. In this study, the analysis will focus upon four broad areas of ecosystem service provision: Carbon storage, fisheries and cultural importance and will aim to sub-categorise provision with respect to species or environmental differences.

The objective of this study is to create a robust scoring system to compare the ecosystem services provided by natural habitats. Seagrass meadows will be used as a case study to create this system. I hypothesise that estuarine meadows will show higher levels of carbon storage than other meadow types due to higher levels of imported carbon. Alternately reef meadows are likely to show higher diversity than other meadow types.

2. Methods

2.a Data collection

Data were compiled from multiple sources; Two literature searches, analysis of footage from Green Island, SeaAroundUs.org and combining data from specific studies on subsistence fishing. This data are then used to measure, two provisioning services :commercial fisheries, subsistence fisheries, one cultural service: IUCN redlist status score, two supporting services: trophic score, diversity and one regulating service: carbon storage.

2.a.i Systematic review

I conducted two systematic literature reviews according to PRISMA (Moher et al., 2009) between March 2020 and June 2020 using four search engines; Scopus, Web of Science, and Science Direct, which were identified as appropriate principal search engines by Gusenbauer and Haddaway, (2019). BioRxiv was also used to find any relevant preprints. Google scholar was not used as while a useful search engine, it has been found to not be reproducible in its search results (Gusenbauer and Haddaway, 2019) which is an important aspect for a systematic review.

The initial literature review was for seagrass sediment organic carbon storage, hereafter referred to as organic carbon storage. Sedimentary carbon was selected for its stability for long term carbon storage (Mateo et al., 1997). The search used the search terms ("Seagrass" OR "*Thalassia*" OR "Enhalus" OR "*Syringodium*" OR "*Halodule*" OR *Thalassodendron*" OR "*Cymodocea*" OR "*Halophilia*") AND "Carbon" AND ("Indo-Pacific" OR "country name"). A complete list of country names used can be found in Appendix I. The second search was for Baited Remote Underwater Video or BRUV data. It used the terms ("Seagrass" OR "*Thalassia*" OR "*Enhalus*" OR "*Syringodium*" OR "*Halodule*" OR *Thalassodendron*" OR "*Cymodocea*" OR "*Halophilia*") AND ("BRUV" OR "Baited Remote Underwater Video") AND ("Indo-Pacific" OR "country name"). Search results were reviewed until no results were shown except BioRxiv for which results were limited to the period of March 2019-March 2020.

To create my data set for organic carbon storage, I excluded papers such as Angrelina et al. (2019), which did not use coring methodology as these would be difficult to compare with previous reviews such as Fourqurean et al. (2012). Review papers such as Rahmawati et al. (2019) were also excluded as its data collection were not separated from its data collected by review, this was done to avoid including the same data multiple times. Similarly, papers that reported data from previous papers were excluded, for example, Schile et al. (2017). Some papers were excluded as they used units that could not be converted into Mega grams per hectare Mg/ha with the information available, for example, Ashok et al. (2019). Equally, papers like Gillis et al. (2017) which did not state values, instead used percentages and rates for the carbon stored, were excluded.

For the Baited Remote Underwater Video (BRUV) data, a number of papers were excluded as they were not in tropical meadows; for example, White (2011). Papers were also excluded when alternative variants of BRUV were used, for example, the addition of artificial lights (e.g. Fitzpatrick et al., 2013), as this would introduce bias into the data set. Few papers were found using BRUV, but the data were supplemented with BRUV data from unpublished sources provided by Swansea University such as Clayton (2006), Gourlay (2017) and new data from Green Island Great Barrier Reef Australia.

2.a.ii Data extraction

Data were extracted from papers on carbon storage by looking for the raw data. If the paper stated a mean value for organic carbon storage to 1m, then this was used. If data were not present to 1m in depth, then data were taken for the depth stated, i.e. 30cm in depth. These data points were then all converted into Mg/Ha to be consistent and aid in comparison.

Bruv data were extracted from papers, new videos and unpublished sources by looking for the complete raw data set, i.e. the count data showing MaxN at the camera drop level. Taking the data at the smallest sampling unit allowed for more effective comparison between environment types as more variation is included.

The SeaAroundUs was used to determine which species of fish were important commercially in the countries studied. Data were taken at the EEZ (Exclusive economic zone) level. The scientific names for each group caught in each EEZ were recorded. These were then compared across EEZs. The groups which were caught in multiple EEZs were then compared to the BRUV data only groups present in the BRUV data were then used for analysis. The commercial families were compared as one unit as the abundances were typically low for individual families.

Similarly to the SeaAroundUs data, published studies where collated on the catches of subsistance fisheries (Unsworth et al. (2014), De la Torre-castro et al. (2014), Exton et al. (2019) and De la Torre-castro and Ronnback (2004)). The fish groups caught in these studies were compared in the same way as the SeaAroundUs data to determine which groups were important to subsistence fisheries in the Indo-Pacific. As with the SeaAroundUs data, only groups present in the BRUV data were used in the analysis.

While this does not give a complete picture on the subsistence fisheries of the Indo-Pacific and more data would be advantageous, it allows a general picture to be built which can be used to assess this service.

2.b Carbon data processing

Papers that clearly assessed multiple meadows (e.g. Chen et al., 2017; Rahayu et al., 2019) were put into the data set as individual meadows rather than one collective. If papers clearly stated the meadow's environment type, for example, Panyawai et al. (2019), then they were classified as stated in the paper. If this was not the case, then the environment type was estimated from locations given in the paper and google earth. If no location was given (for example, LaRoche et al. (2019), then the environment type was left blank.

The organic carbon storage was then extrapolated to estimate the carbon stored in the top 1 metre of sediment in accordance with Fourqurean et al. (2012) as this is most likely to be released if the seagrass is lost (Fourqurean et al., 2012). Fourqurean's methodology was chosen as this is how the majority of papers found extrapolated their carbon storage. Also, Fourqurean et al., (2012) was the last major review of carbon storage in Indo-Pacific seagrass meadows and so following the same methodology allows the data to be compared directly.

2.c Fish assemblage data processing

To obtain data on which fish species utilise seagrass meadows across the Indo-Pacific region BRUV data were compiled from around the region (unpublished sources and new data source) and supplemented with a meta-analysis for any further BRUV data from the region. Each BRUV drop was taken as an individual sample for this study.

2.c.i BRUV footage processing

New footage from 23 camera drops from seagrass meadows around Green Island Great Barrier Reef Australia was analysed manually. To do this, the footage was first trimmed into 30 minute videos. Each video was then watched the species present identified visually. The maximum number of individuals MaxN of each species was recorded for each frame (Andradi-Brown et al., 2016). For each video, the highest count for each species was then recorded as the MaxN for that video. These values were then used in conjunction with MaxNs recorded by other studies to calculate scores for each metric.

Footage collected from seagrass Green Island Great Barrier Reef Australia was trimmed into 30-minute videos. The videos were then analysed by hand to give a MaxN value for each species for comparison with the collated data from published and unpublished sources. MaxN is the maximum number of individuals of one species in one frame during the recording (Andradi-Brown et al., 2016).

2.c.ii Trophic scoring

Trophic scores were calculated for each BRUV sample. To calculate trophic scores, I used trophic level combined with species richness to give each assemblage a numerical value. Trophic scoring involves allocating each species a score between 1 and 5 according to their position in the food chain (Gourlay, 2017). The values were taken from FishBase (2020) and are based on the diet of a particular species; 5 being an apex predator and 1 being a primary producer, these scores do not have to be integer values, so decimal values are assigned if a species fits into multiple levels. The average trophic level for the whole sample is calculated and multiplied by the number of species present in the sample (species richness) to give the sample a trophic score. These scores can then be compared between samples or environments to reflect the usage by individuals and species at different levels of the food chain.

The trophic score was then calculated with this equation from Gourlay (2017).

$$Trophic\ score = \left(\frac{\sum TL \times MaxN}{\sum MaxN}\right) \times n$$

Where TL is the trophic level, MaxN is the maximum number of individuals observed at any one time in the sample, n being the number of species present in the sample. For example, if a species had a trophic level of 2.8 and 30 of individuals were seen in the drop sample, and there were 2 different species seen in the sample, the score would be calculated as follows. The second species has a trophic score of 4 and had 3 individuals present. $\sum TL \times MaxN = ((2.8x30)+(4x3)) = 96$

$\sum MaxN = 30+3 = 33$

N=2

So the final equation would be trophic score=(96/33)x2=5.82

2.c.iii IUCN redlist status scoring

IUCN redlist status scoring used the MaxN values from the BRUV data from published and unpublished sources. IUCN redlist status scoring follows a similar principle to other systems, such as, the Community Conservation Index (Chadd and Extence, 2004), System Aqua (Willen, 2009), Dragonfly Biotic Index (Simaika and Samways, 2009), Lake Assessment for Conservation system (Duker and Palmer, 2009) and System for Evaluating Rivers for Conservation (Boon, 2000, 1997; Boon et al., 2002) by combining the IUCN Redlist status with other parameters such as species richness, susceptibility to disturbances or range of species. It does, however, differ from these systems, which are largely focused on freshwater. IUCN redlist status scoring is based on BRUV sampling and is more appropriate for systems with more visibility in the water column, which are likely to be marine; this could be applied to freshwater systems if conditions allowed. Each species sample was allocated its status from the IUCN redlist. Each IUCN redlist status was then valued numerically, with “Least concern" scoring 1, "Vulnerable scoring" 2, "Near threatened" scoring 3 and "Endangered" scoring 4. These values were then applied to the following equation based on the equation shown above from Gourlay (2017) to give an IUCN redlist status score for each site.

$$Status\ score = \left(\frac{\sum S \times MaxN}{\sum MaxN}\right) \times n$$

Where S is the IUCN redlist status, MaxN is the maximum number of individuals observed at any one frame in the sample, and n is the number of species present in the sample. For example, if a species was considered vulnerable and 30 individuals were seen in the drop and there were 2 different species seen in the sample, then the score would be calculated as follows. The second species is rated least concern and had 3 individuals present. ∑S×MaxN=((2x30)+(1x3))=63

∑MaxN=30+3=33

N=2

So the final equation would be Status score=(63/33)x2=3.82

2.c.iv Subsistence fisheries

The most abundantly caught groups by small-scale fisheries in studies by Unsworth et al. (2014), De la Torre-castro et al. (2014), Exton et al. (2019) and De la Torre-castro and Ronnback (2004) were collated to ascertain which fish families are important to subsistence fisheries. The numbers of fish in these families found in the BRUV data were then extracted to give a comparable unit to compare the potential for each meadow to be used as subsistence fisheries. The families that were significant in multiple studies (Gerreidae (Mojarra), Labridae (Wrasses), Lethrinidae (Emperors), Lutjanidae (Snapper), Mullidae (Goatfishes), Nemipteridae (Threadfin bream), *Scaridae* (Parrotfish) and Siganidae (Rabbitfish) were then compared to the BRUV data. These families were compared as one unit of subsistence fish families rather than individuals as the abundances were typically low for individual families.

2.c.v Commercial fisheries

Data were compiled from the Sea Around Us Project (SeaAroundUs, 2020) on the most commonly caught groups by tonnage in each country for which BRUV data were found either from published or unpublished sources. The data were taken at the EEZ level. The most commonly caught species for each country was then recorded. The number of EEZs in which the group were recorded in was then used to determine their relative importance as commercial groups for the region.

The occurrence of fish groups in the SeaAroundUs project was in descending order *Scombridae* (n=16), *Carangidae* (n=9), *Clupidae* (n=8), *Engraulidae* (n=5), *Elasmobranchii* (n=3), *Leiognathidae* (n=3), *Peneidae* (n=3), *Lethrinidae* (n=2), *Nemopteridae* (n=2), *Scianidae* (n=2), *Scombridae* (n=2), *Siganidae* (n=2), *Anguilliformes* (n=1), *Ariidae* (n=1), *Berycidae* (n=1), *Carcharhinidae* (n=1), *Cepolidae* (n=1), *Haemulidae* (n=1), *Labridae* (n=1), *Latidae* (n=1), *Lutjanidae* (n=1), *Muglidae* (n=1), *Scaridae* (n=1), *Sparidae* (n=1), *Synodontidae* (n=1) and *Triglidae* (n=1). Groups that were not represented in the BRUV data were then discounted for this study. The remaining groups were then adjusted to family level if not already presented in this manner except *Elasmobranchii* (sharks and the rays) was kept at subclass level due to the low number this group is typically found in. The resulting groups were (*Carangidae* (Jacks), *Elasmosbrachii* (sharks and the rays)*, Clupeidae* (Herring and Sardines), *Lethrinidae* (Emperors), Nemipteridae (Threadfin bream), *Scaridae* (Parrotfish), and *Siganidae* (Rabbitfish)). The numbers of fish in these families found in the BRUV sampling where then extracted to give a comparable unit to compare the potential for each meadow to be used as commercial fisheries.

2.c.vi Diversity

For each BRUV drop, fish were classified to species level where possible if not then to genus level. Shannon's diversity Index was calculated for each BRUV drop. This value was then used to compare the relative diversity of each site as a proxy for the relative resilience of the fish assemblage present.

2.d Categorising meadows

Meadows were categorised using google earth and their location from GPS if provided or from the area stated in the study. If the location was unclear, then the environment type was not assigned. Estuarine meadows were defined as meadows within an estuary. Lagoon meadows as meadows surrounded on three sides with land, but there is still a link to the ocean. Reef meadows were meadows that were clearly close to reefs. Coastal meadows were meadows near the coast that were not near reefs, estuaries, or inside lagoons. Deep water meadows were typically further offshore and stated as such within the paper.

2.e Forming scales

To create a statistically robust scale that can be modified over time as more data becomes available, I opted for a scale based on percentiles within each scoring metric. A similar approach was adopted by Kam et al. (2013), who used four categories of 0-25, 25-50 etc., rather than the ten present in my system. A ten part scale was opted for in this study to highlight more variation than might be seen in a four part scale. Using percentiles should allow the consistent comparing of seagrasses on a large scale and lead to a greater understanding of a high value or low-value meadow with time while similarly increasing the capture of variation by using a larger number of categories. This straightforward model allows any parameter with data available to be turned into a scale comparable to any other in a non-monetary fashion. This approach was opted for, as it is far less subjective than scales used in some other scorecards. For example, as previously mentioned, Nuttle (2013) used a scoring of 1-5, but each number score referred to the state as “poor’ or ‘good’ or ‘very good’ which can be interpreted differently by observers with differing experience levels. Alternatively, my system attributes specific ranges for each score similar to the Eco-Heart index (Sakai et al., 2018). In this way, there should be no difference in the scoring of sights between individuals.

To create my scorecards each data set's 95% confidence interval was taken to avoid skewing by outliers. Scores were then assigned to ranges within the 95% as shown in table 1. The outliers are not excluded, but instead, they would be assigned a score 0 or 10 depending on which end of the range they fall. The remaining 95% was then separated into nine categories resulting in a scoring system of 0-10 for each ecosystem service.

Table 1. An example of a scorecard for an ecosystem service showing the percentiles used.

Percentage range used	Ecosystem service score
>2.5%	0
2.5-13.06%	1
13.06-23.61%	2
23.61-34.17%	3
34.17-44.72%	4
44.72-55.27%	5
55.27-65.83%	6
65.83-76.39%	7
76.39-86.94%	8
86.94-97.5%	9
< 97.5%	10

These scores were then used to give overall ecosystem scores for a meadow. To achieve this, the scores were averaged for each type of ecosystem service. For our metrics alone, these would add together to give a score from 0-60 for each meadow. However, this would limit the scoring systems ability to include other metrics in the future. To allow further data to be incorporated into the scoring system, I present the scores in the categories used in the Millennium ecosystem assessment (2005). Provisioning services included commercial and subsistence fisheries. Supporting services included diversity and trophic score. Regulating services carbon storage and cultural services IUCN redlist status score. Categorising in this way gives a maximum score of 40. As more data are gathered on different ecosystem services, they can be incorporated into the ecosystem score. The scoring system is separated by environment type: coastal, lagoon, reef, estuarine and deep water to compare similar meadows to each other.

2. f Statistics

Statistical analysis was carried out in R (R Core Team, 2020).

A Shapiro-Wilk test was conducted on each data set to confirm whether they were normally distributed or not.

To compare the average organic carbon stored in the top 1 m of sediment between environments, a Kruskal-Wallis rank-sum test was performed. Which was followed by a Dunn test to determine *post-hoc* differences. For the BRUV data, PERMANOVA tests were performed using the package vegan (Oksanen et al. 2020), comparing subsistence fish, commercial fish, IUCN redlist status score, trophic score and diversity between environments and countries. PERMANOVAs used 999 permutations and Bray methodology. Pairwise PERMANOVAs were then performed using the package pairwiseadonis (Arbizu 2017); this showed where the differences lay in the data. A significance level of P = 0.05 was used for these tests, and the mean±SD is presented.

PERMANOVA was used as it was designed with ecological data in mind, including count data for abundance for large numbers of species (Anderson, 2017). Which accounts for a large proportion of my data. While it would be possible to use ANOVA to analyse my data, a PERMANOVA will result in p values for univariate analysis without the need for the data to be normally distributed as they are calculated by permutations (Anderson and Millar, 2004). My data also contained many zero values, which a PERMANOVA test is insensitive to.

3. Results

3.a Systematic review

For organic carbon storage, 2041 articles were reviewed. Of these, 1957 were excluded on title and abstract by the criteria outlined earlier or through being duplicates. This left 84 remaining studies. Of these 84 papers, 63% of the remaining papers were then excluded for: location (18%), not reporting mean Carbon storage (27%), inaccessibility (7%), methodology (7%) or re-reporting of data (4%), i.e. revaluations or reviews. The exclusion process left 31 papers, 5 of which were removed for using units that were incompatible with the other studies leaving 26 suitable studies for inclusion. These papers covered 68 meadows in 12 countries and 1 territory, the sampling depths ranged from3-140cm. The flow chart for this process can be found in Appendix II. The data set was formed by taking the papers at the meadow level where possible, rather than as a whole study resulting in 68 meadows in total. The carbon storage data spanned 12 countries across the Indo-Pacific region: Australia (n=7), China (n=6), Guam (n=1), India (n=2), Indonesia (n=8), Kenya (n=4), Malaysia (n=4), Saudi (n=12), Singapore (n=1), Tanzania (n=1), Thailand (n=4), UAE (n=18) (Figure 1). Core sampling depth ranged between 3 cm-140 cm, all of which were included due to the limited data available across the region. This was done to increase understanding of the general trends in the area. A similar issue was found by Fourqurean et al. (2012), where only 41 of the 219 sites they found had been sampled to 1 m. The number of carbon samples for each environment type can be seen in table 2.

Searching for BRUV papers over the region resulted in 72 papers, 63 of which were excluded at the title and abstract stage. Of the nine remaining papers, seven were excluded as they were not conducted on tropical meadows. The remaining two were supplemented with data previously collected during unpublished studies and theses. Data collected spanned 357 drops over 7 countries and one territory: Australia (n=23), Cambodia (n=6), Chagos (n=5), Indonesia (n=84), Mozambique (n=22), Myanmar (n=51), Philippines (n=117), and Zanzibar (n=55) (Figure 1). The BRUV data were then used to assess commercial fisheries, subsistence fisheries, diversity, trophic score, IUCN redlist status score the relative number of samples for each parameter by environment type can be seen in table 2.

Table 2. Number of samples for each environment type and parameter.

Data type	No of Reef meadows	No of Lagoon meadows	No of Coastal meadows	No of Deep water meadows	No of Estuarine meadows	Undefined meadows
Carbon storage	23	10	23	5	11	3
Commercial fisheries	133	34	186	4	0	0
Subsistence particular fisheries	133	34	186	4	0	0
Diversity	133	34	186	4	0	0
IUCN redlist status score	133	34	186	4	0	0
Trophic score	133	34	186	4	0	0

The distribution of studies included in my review can be seen in Figure 1. Data were present for organic carbon storage and fisheries in only two approximate locations, Tanzania and the Great Barrier Reef in Australia, highlighting the relative lack of quantification of seagrass ecosystem services in the Indo-Pacific region.

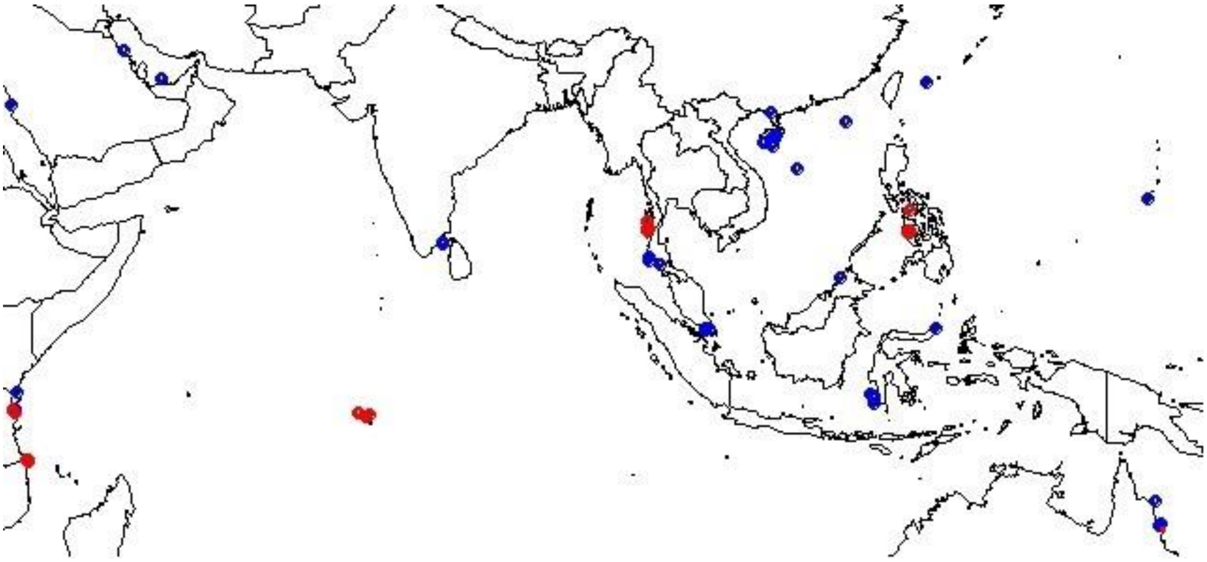

Figure 1. A map of samples across the region found during the literature review as well as new data and unpublished sources, BRUV (Red) and organic carbon storage (Blue). Sampling overlapped only in Tanzania and the Great Barrier Reef in Australia.

3.b Data processing

The distribution of six data sets used to assess ecosystem services (organic carbon storage, subsistence fisheries, commercial fisheries, diversity, IUCN redlist status score, trophic score) can be seen in Figure 2. All of the six data sets were non-normally distributed: Organic carbon storage ($W=0.88$, $P=8.2e^{-09}$), subsistence fisheries ($W=0.58$, $P=2.2e^{-16}$), commercial fisheries ($W=0.56$, $P=2.2e^{-16}$), diversity ($W=0.93$, $P=1.54e^{-11}$), IUCN redlist status score ($W=0.76$, $P=2.2e^{-16}$) trophic score ($W=0.79$, $P=2.2e^{-16}$)

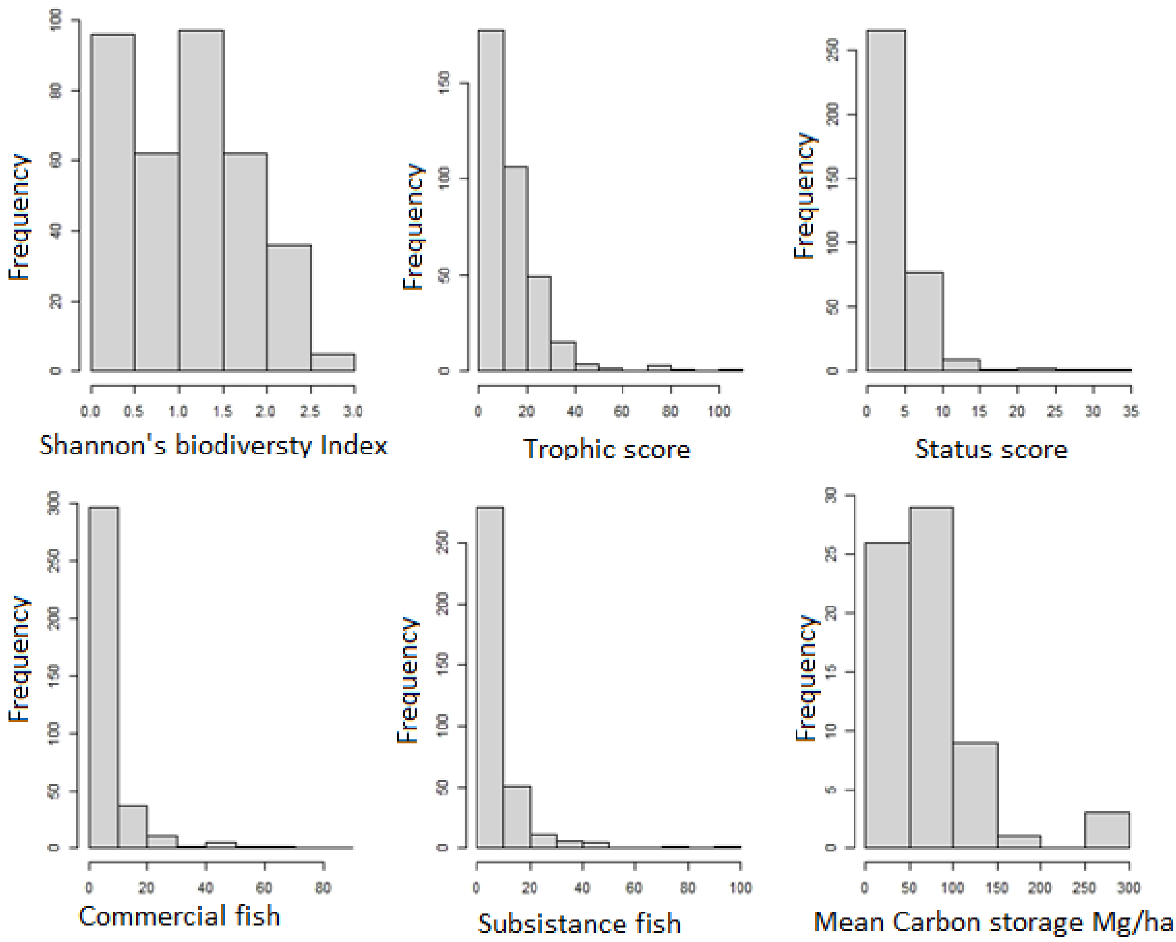

Figure 2. Distribution of the raw data in each category used for analysis. No data set shows a normal distribution.

3.c Species distribution

A wide variety of species was seen in the study (n=313), the ten most commonly found species where *Lethrinus variegatus* (n=426), *Siganus fuscescens* (n=418), *Plotosus lineatus* (n=288), *Lethrinus harak* (n=269), *Siganus luridus* (n=256), *Hypoatherina temminckii* (n=213), *Cheilio inermis* (n=189), *Leptoscarus vaigensis* (n=153) and *Parupeneus barberinus* (n=137). *Lethrinus variegatus, Cheilio inermis* and *Leptoscarus vaigensis* were all recognised as commercially important species in Tanzania, *Cheilio Inermis, Leptoscarus vaigensis,* juveniles were found exclusively in seagrass meadows (Lugendo, 2007). *Lethrinus harak* is recognised as a highly commercially important species in Indonesia (Unsworth et al., 2010). The data compiled from the SeaAroundUs (SeaAroundUs, 2020) identifies *Lethrinus variegatus, Lethrinus* harak, *Siganus luridus, Siganus fuscescens, Cheilio inermis* and *Leptoscarus vaigensis* as part of the families of import to commercial (SeaAroundUs, 2020) and subsistence fisheries (De la Torre-

Castro et al., 2014; De La Torre-Castro and Rönnbäck, 2004; Exton et al., 2019; Unsworth et al., 2014).

The number of species and individuals at each site varied (Figure 3.) Deep water meadows had the highest mean number of species 10.2± 2.86, and the next highest was reef meadows 5.4±3.37 then coastal meadows 4.2±5.17 the lowest was lagoon meadows 2.9±3.60. The number of species seen at each site was significantly different (PERMANOVA, $F=6.17$, $p=0.001$). Significant differences were seen between deep water meadows and all other environment types ($p<0.001$). The same was seen for reef meadows ($p<0.001$). Coastal and lagoon meadows showed no significant difference in the number of species present. A similar trend was seen with the number of individuals present, deep water scoring the highest 70.2±74.80 lagoon the lowest 4.9±7.64, with reef and coastal meadows averaging 18.4±20.48 and 13.43±8.67, respectively. The difference in the number of individuals present at a sample site was significant between environment types (PERMANOVA, $F=12.991$, $P=0.001$). Contrary to the number of species, coastal meadows showed significantly different numbers of individuals compared to all other environment types ($p<0.001$). There was no significant difference in the number of individuals between deep water and reef meadows. Both deep water and reef meadows were significantly different from lagoon meadows ($P<0.001$).

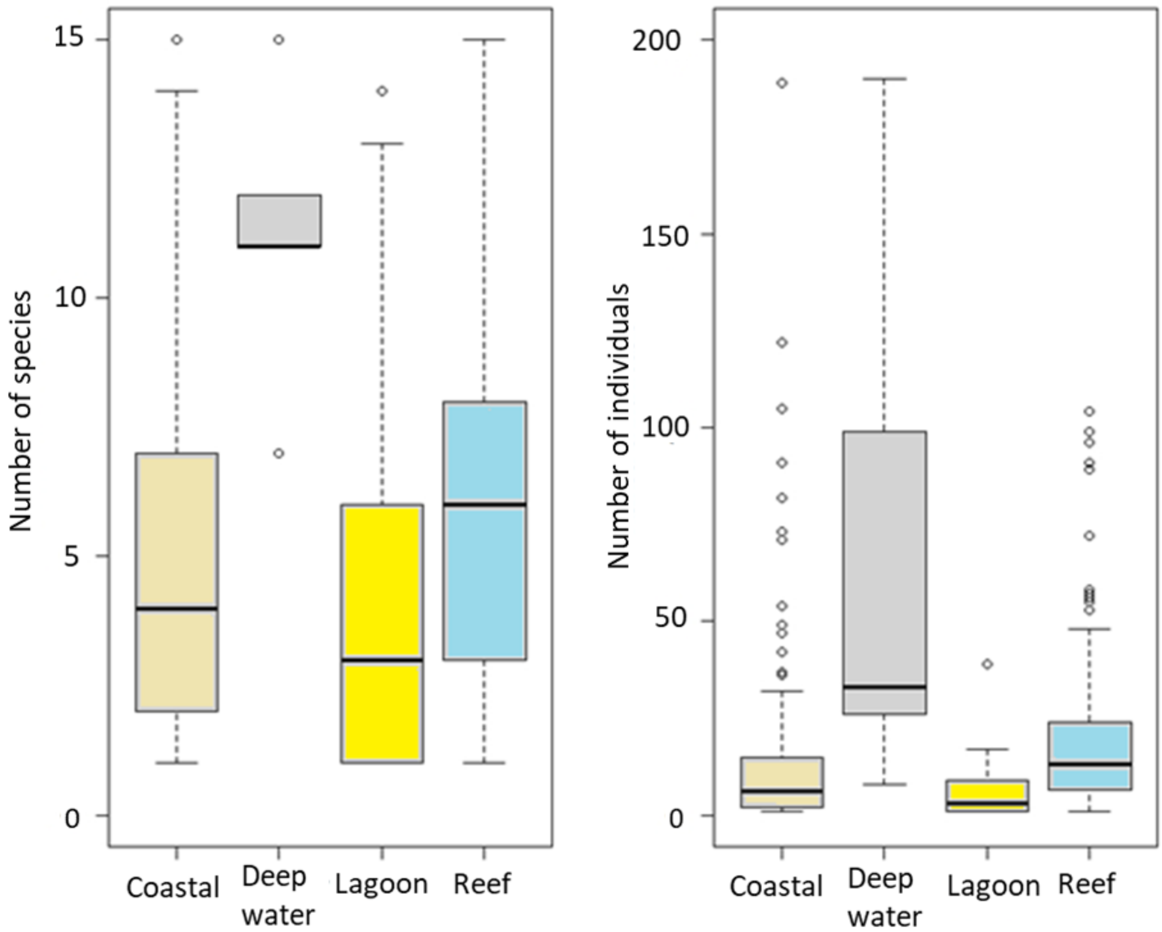

Figure 3. Comparison of the number of species and individuals present at each BRUV drop across environment types. Deep water meadows show higher numbers of individuals and species than other environment types.

3.d IUCN redlist status score

The classification by the IUCN (IUCN, 2020) of species varied with environment type (Figure 4). Across all environment types, there were very few species on average classified as Data Deficient (DD), the highest being coastal meadows at 0.073±0.49 species and reef meadows at 0.24±1.01 species. All other environment types had no DD species present. PERMANOVA showed no significant difference in the number of DD species present between environments (F=1.80, p=0.155). The majority of species sampled were of Least Concern (LC). Deep water meadows had the highest mean presence of LC species with 67.4±75.65 species. Reef coastal and lagoon were 16.08±19.10 species, 10.77±26.78 species and 3.24±4.94 species, respectively. There was a significant difference in the number of LC species between environment types (PERMANOVA, F=19.691, p=0.001). No Near Threatened (NT) species were found in lagoon meadows. Conversely, in deep water meadows, the average was 1.4±1.67 species, which was the highest mean for the IUCN redlist status coastal and reef meadows had 0.04±0.21 species and 0.11±0.32 species, respectively. There was a significant difference in the number of NT species between environments (PERMANOVA, F=39.931, p=0.001). Very few species rated Vulnerable (V) none were seen in deep water or reef meadows. Coastal meadows had a mean value of 0.59±0.19 species lagoon, a mean of 1.7±1.67 species. No significant difference was found between environments for the number of vulnerable species present (PERMANOVA, F=2.37, p=0.084). Lagoon, coastal and reef meadows had a similar number of unlisted species present 0.09±0.38 species, 0.86±4.49 species and 0.81±3.12 species, respectively. There were no unlisted species recorded in the deep water meadows. There was no significant difference between the number of species unlisted between environments (PERMANOVA, F=1.83, p=0.127).

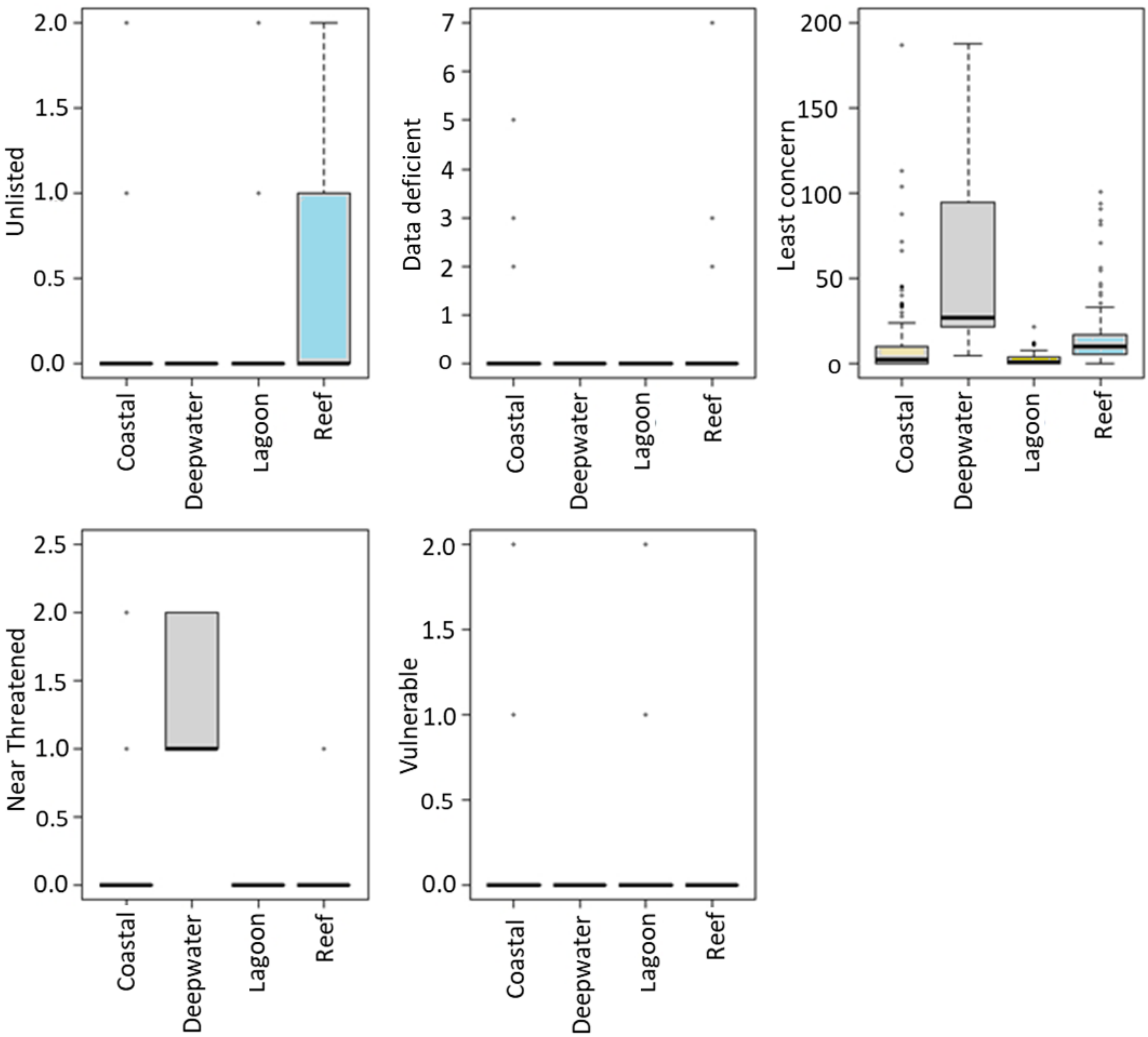

Figure 4. The number of individuals of each status level under the IUCN redlist in each environment type. Deep water meadows show much higher levels of individuals categorised as near threatened and least concern.

3.e Ecosystem service variation

Lagoons showed the highest mean organic carbon storage at 129.04±87.06 Mg/ha, followed by coastal and reef meadows with means of 69.29±41.20 Mg/ha and 64.45±27.49 Mg/ha, respectively. Estuary and deep water meadows showed low levels of organic carbon storage relative to other environments. Estuary meadows had a mean of 32.65±22.53Mg/ha, and deep water meadows had a mean of 29.1±21.05 Mg/ha. Significant differences were found between environment types (Kruskal-Wallis, KW= 12.753, p=0.006154). A Post-hoc Dunn test on the organic carbon data set indicated that there was a significant difference in organic carbon storage between coastal and deep water (p=0.017), coastal and estuary (p=0.0054), deep water and lagoon (p=0.0034), estuary and lagoon (p=0.0007) and reef and estuary meadows (p=0.0061). There was no significant difference between coastal and lagoon or reef, deep water and estuary, or reef and lagoon.

Reef meadows had the highest mean Shannon's diversity index of 1.26±0.59, followed by coastal meadows at 0.88±0.80, and lagoon meadows had the lowest mean diversity 0.73±0.83. This was found to be significantly different by PERMANOVA (F=18.608, p=0.001). Pairwise tests showed that these differences were between coastal and reef meadows (F=30.59, p=0.003) and lagoon and reef meadows (F=28.59, p=0.003) but not between coastal and lagoon meadows (F=1.44, p=0.702). The same trend was seen with trophic scores with mean values of 16.92±9.01, 11.07±15.23, and 6.56±9.01 for reef coastal lagoon meadows, respectively, and was also found to be significantly different with PERMANOVA analysis (F=27.816, p=0.001). Pairwise testing also showed a significant difference between reef meadows and either coastal (F=42.37, p=0.001) or lagoon (F=49.38, p=0.001) meadows. There was no significant difference between lagoon and coastal meadows (F=3.64, p=0.027). Trophic score showed particularly high variance, implying a lot of variation within the data. Similarly, reef meadows had the highest mean IUCN redlist status score at 5.06±2.81, with coastal meadows showing a mean of 3.33±4.62 and Lagoon meadows 2.15±2.84 and was found to be significantly different using PERMANOVA (F=27.816, p=0.001). The differences between meadows showed the same trend as the other metrics, with reef meadows being significantly different to coastal meadows (F=54.23, p=0.003) and lagoon meadows (F=51.69, p=0.003) with no significant difference between coastal and lagoon meadows (F=2.34, p=0.375). Reef meadows had by far the highest abundance of commercially important species with a mean of 9.62±13.07 individuals, almost double that of coastal meadows at 4.68±10.42 individuals, roughly ten times that of lagoon meadows which had a mean of 0.97±1.71 individuals there was significant differences in commercial species data set shown by PERMANOVA (F=39.052, p=0.001). Pairwise testing showed these significant differences were between coastal and reef meadows (F=58.90, p=0.003) and lagoon and reef meadows (F=49.20, p=0.003). There was no significant difference between coastal and lagoon meadows

(F=4.34, p=0.213). However, the standard deviations on commercial species were high, so there is more overlap than the means suggest. Subsistence species abundance differed from the other metrics as coastal meadows had the highest mean abundance of subsistence species at 8.02±15.05 individuals, reefs had a mean of 7.77±10.02 individuals and lagoons 2.71±3.95 individuals. PERMANOVA analysis showed there were significant differences between environments for subsistence species abundance (F=9.69, p=0.001). Pairwise testing showed that these significant differences followed the same trend as other fish assemblages data. Reef meadows were significantly different from coastal (F=13.098, p=0.003) or lagoon (F=13.63, p=0.003) meadows. The variance in ecosystem services provided by the meadows in this study shows that Lagoons appear to store much larger amounts of carbon than the other environment types. Conversely, reef-associated meadows appear to outperform the other environments in the fisheries services, particularly commercial and subsistence species (Figure 5).

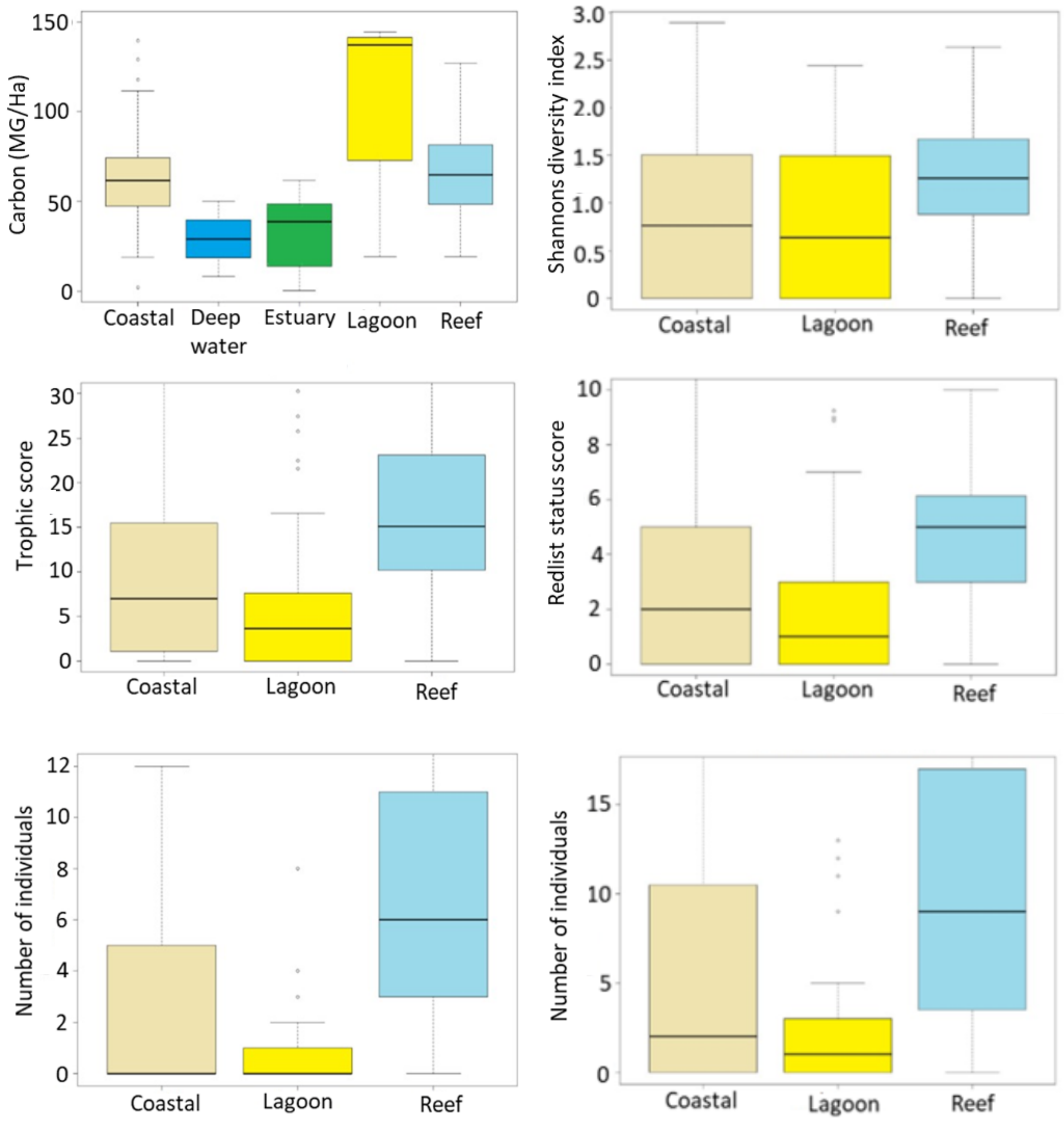

Figure 5. Variation in the collected data between seagrass meadows in different environments in the Indo-Pacific. Colour denotes environment type: Brown (coastal), Dark blue (deep water), Green (estuary), Yellow (lagoon), Light blue (reef). Some of the outliers have been omitted, allowing the majority of the data to be seen more clearly. These outliers were included in all statistical analyses. A) Carbon data are from cores to 1m in depth collected via systematic review. Lagoon meadows show much higher carbon storage than other meadow types, in particularly deep water and estuarine meadows. B) Shannon's diversity index calculated from the MaxN values of the BRUV data. Reef meadows show higher mean diversity than other meadow types. However, there is a reasonable amount of overlap.

C) Trophic scores calculated from MaxN data. Reef meadows show the highest values, but there is some overlap with coastal meadows. Lagoon meadows score much lower than reef meadows. D) IUCN redlist status scores calculated from MaxN data. Coastal and lagoon meadows show similar means, whereas reef meadows have much higher mean scores. E) Number of individual fish which were determined to be of commercial importance from SeaAroundUs data. Counts are from BRUV samples. Reef Meadows have much higher mean numbers of individuals, whereas lagoon and coastal meadows average around 0. F) Number of individual fish which were determined to be of importance to subsistence fisheries by compiling data from the literature. Count data were from BRUV samples. Lagoon average very low numbers of individuals, as do coastal meadows, but these have slightly higher numbers. Reef meadows have much higher mean numbers of individuals. As species composition can be a significant factor predicting carbon storage (Lavery et al. 2013), mixed and mono-specific meadows were compared. Figure 6 shows this comparison. Single species meadows showed mean storage to 1 m of 96.31± 93.47 Mg/ha and mixed-species meadows a mean of 65.75±37.04 Mg/ha. There was, however, no significant difference in the means of my data set (Kruskal-Wallis, KW=0.079, p= 0.78).

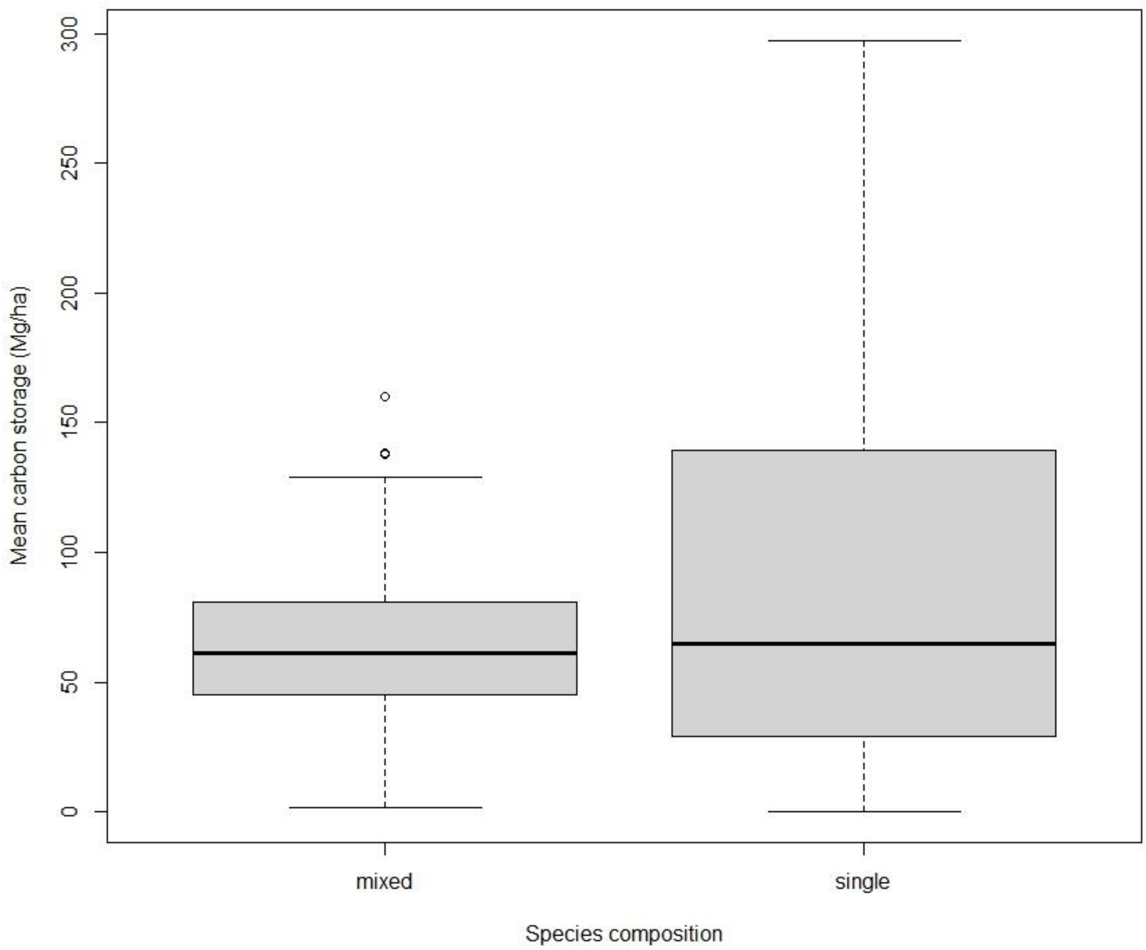

Figure 6. Comparison of organic carbon storage to 1m depth between mixed species and single-species meadows. There is a much larger range in carbon storage values for single species meadows than mixed species. However, mean values appear to be similar.

3.f Scorecards

As the ecosystem services vary between environment types and these are typically easy to identify from one another. The scorecards for each environment type can be seen in tables 3, 4 and 5. Scores for carbon storage in deep water and estuarine meadows can be seen in appendix IV. The range of values in each row represents 10% of the data within the 95% confidence interval for that ecosystem service. These scorecards allow each service to be easily converted into a comparable ecosystem score. The results of two meadows from Zanzibar and Australia can be seen in Figure 6.

Table 3. Scorecard for Reef seagrass meadows.

Reef Scorecard						
Subsistence species (individuals present)	Commercial species (individuals present)	Organic carbon storage Mg/Ha	Trophic score	Diversity	IUCN redlist status score	Ecosystem services score
0-3.924	0-3.87	0-5.84	0-3.7	0-0.4	0-0.99	0
3.925-7.84	3.88-7.75	5.85-11.94	3.8-7.3	0.5-0.68	1-1.99	1
7.85-11.774	7.76-11.63	11.95-17.24	7.4-10.7	0.69-1.00	2-2.99	2
11.775-15.69	11.64-15.51	17.25-18.94	10.8-12.7	1.01-1.08	3-3.99	3
15.7-19.624	15.52-19.39	18.95-35.44	12.8-14.4	1.09-1.20	4-4.62	4
19.625-23.54	19.4-23.27	35.45-41.34	14.5-16.2	1.21-1.38	4.63-4.99	5
23.55-27.474	23.28-27.15	41.35-44.19	16.3-18.6	1.39-1.54	5-5.99	6
27.475-31.39	27.16-31.3	44.19-53.99	18.7-23.0	1.55-1.66	6-6.05	7
31.4-35.324	31.03-34.91	54.00-59.44	23.1-27.4	1.67-1.87	6.06-7.02	8
35.325-39.24	34.92-38.8	59.45-64.14	27.5-32.2	1.88-2.21	7.03-9.69	9
>39.24	>38.8	>64.14	>33.3	>2.22	>9.7	10

Table 4. Scorecard for Coastal seagrass meadows

Coastal Scorecard						
Subsistence species (individuals present)	Commercial species (individuals present)	Organic carbon storage Mg/Ha	Trophic score	Diversity	IUCN redlist status score	Ecosystem services score
0-3.74	0-2.124	0-2.1	0-2.63	0-0.635	0-1.075	0
3.75-7.49	2.125-4.24	2.2-6.1	2.64-5.27	0.636-0.693	7.076-2.150	1
7.5-11.24	4.25-6.374	6.2-12.5	5.28-7.90	0.694-0.993	2.151-3.225	2
11.25-14.99	6.375-8.49	12.6-19.2	7.91-10.54	0.994-1.096	3.226-4.300	3
15-18.74	8.5-10.624	19.3-22.9	10.55-13.18	1.097-1.260	4.301-5.375	4
18.75-22.49	10.625-12.749	22.9-25.9	13.19-15.82	1.261-1.389	5.376-6.450	5
22.5-26.24	12.75-14.874	26-36.3	15.83-18.45	1.390-1.558	6.451-7.525	6
26.25-29.99	14.875-16.99	36.4-44.0	18.46-21.09	1.559-1.789	7.526-8.600	7
30-33.74	17-19.124	44.0-49.7	21.10-23.72	1.790-2.048	8.601-9.675	8
33.75-37.49	19.125-21.25	49.7	23.73-26.36	2.049-2.307	9.676-10.75	9
>37.5	>21.25	>49.7	>26.36	>2.307	>10.75	10

Table 5. Scorecard for lagoon seagrass meadows

Lagoon Scorecard						
Subsistence species (individuals present)	Commercial species (individuals present)	Organic carbon storage Mg/Ha	Trophic score	Diversity	IUCN redlist status score	Ecosystem services score
0-1.135	0-0.4	0-8.4	0-2.63	0-0.228	0-0.893	0
1.36-2.27	0.41-0.8	8.41-9.2	2.64-5.27	0.229-0.456	0.894-1.786	1
2.28-3.41	0.81-1.2	9.21-10.0	5.28-7.91	0.457-0.684	1.787-2.678	2
3.42-4.54	1.21-1.6	10.01-18	7.92-10.54	0.685-0.912	2.679-3.571	3
4.55-5.68	1.61-2	18.01-23.6	10.55-13.17	0.913-1.140	3.572-4.464	4
5.69-6.81	2.01-2.4	23.61-29.6	13.18-15.81	1.141-1.368	4.465-5.357	5
6.82-7.95	2.41-2.8	29.61-38.4	15.82-18.45	1.369-1.596	5.358-6.249	6
7.96-9.08	2.81-3.2	38.41-52.01	18.46-21.09	1.597-1.824	6.249-7.142	7
9.09-10.22	3.21-3.6	52.0-54.4	21.10-23.72	1.825-2.052	7.143-8.035	8
10.23-11.35	3.61-4	54.41-5.2	23.73-26.36	2.053-2.280	8.036-8.928	9
>11.35	>4	>56.2	>26.36	>2.280	>8.928	10

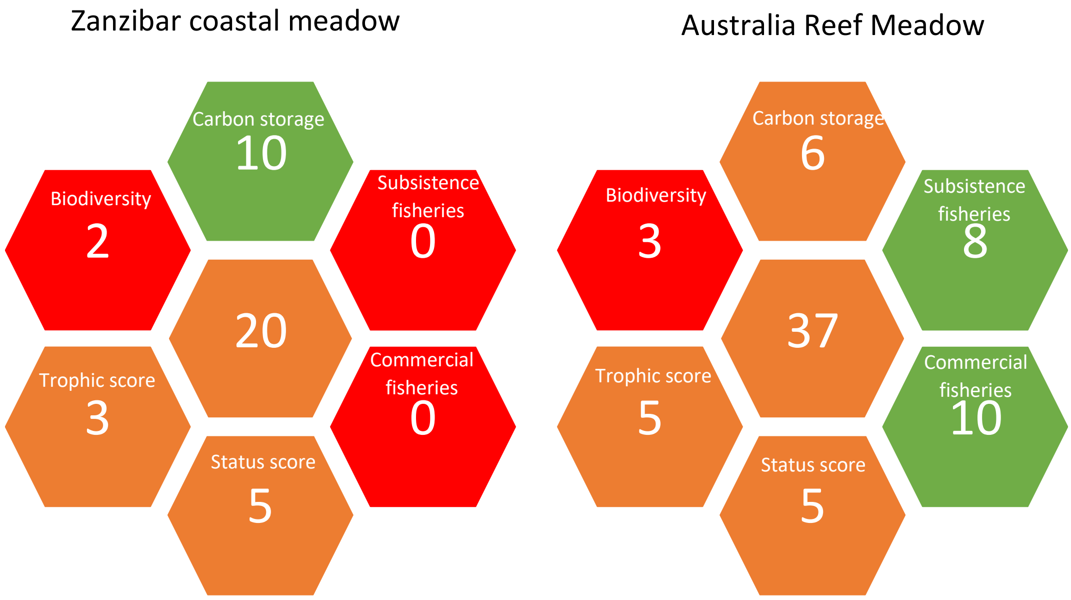

Figure 7. Ecosystem service scores for two meadows from my data. The central number denotes the total ecosystem services score out of 60. External numbers denote the score of that service out of 10. These scores were derived using the scorecards presented in tables 3-5.

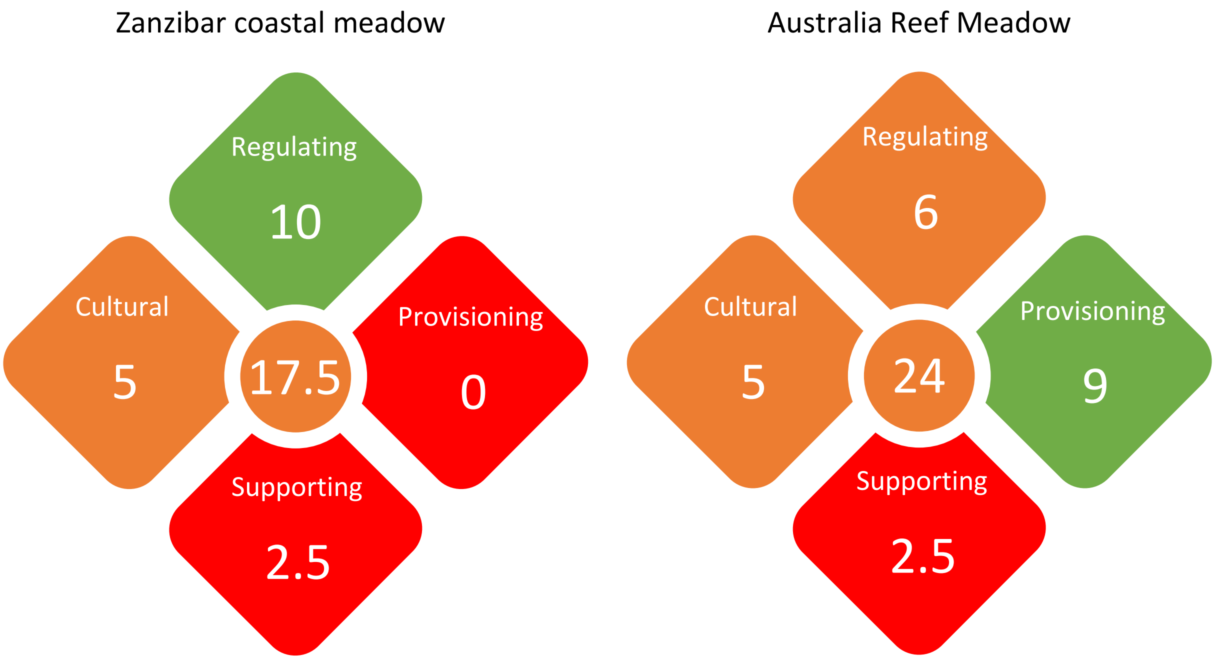

Figure 8. Scores for the same two meadows as in Figure 7 but categorised under the four types of service stated in the Millennium ecosystem assessment (MA, 2005). Scores which are in the same category i.e. provisioning services (subsistence and commercial fisheries) are

averaged to give a score out of 10 for the category. Resulting in an overall score of 40 for the ecosystem services provided.

4. Discussion

Ecosystem services in the Indo-Pacific are shown in my thesis data to vary by the environment type in which the meadow can be found. Notably, lagoon meadows show significantly higher carbon storage than meadows found in other environment types, possibly attributed to their enclosed nature. Conversely, Reef meadows have significantly higher levels of commercial and subsistence fishery species, likely due to their proximity to coral reefs. Reef meadows also showed significantly higher IUCN redlist status scores, trophic levels and diversity than other meadow types, which would imply a larger number of rare predators such as sharks being present in these meadows. The data also alludes to variation in the provision of ecosystem services, but more data would be needed to confirm this. Contrary to Lavery *et al.*, (2013), there was no significant difference in the carbon storage between mono-specific and mixed-species meadows.

My literature search for organic carbon storage papers returned 68 data points which is less than might be expected for a region this size which was noted by Fourqurean et al. (2012), who found 8 data points for the Indo-Pacific but 29 in the Mediterranean. This shows that there has been an increase in research focus on seagrass organic carbon storage in the Indo-Pacific since 2012. These papers show a reasonable spread across the Indo-Pacific, but some countries show more research than others, for example, China and Australia had 6 and 7 meadows studied, respectively. Similar to Fourqurean et al. (2012), few studies were found in Africa. However, the number of studies in Africa does appear to be rising as I found 5 data sets on the African coastline. Equally, the Arabian Gulf had data for 30 meadows, and South East Asia, excluding China, had 20. Areas such as Hainan Island in China appeared to have been studied a disproportionately high amount regarding carbon content, while other areas and countries had none or very little data, for example, India, which only had two data points. Showing a disparity in research in the region, which was noticed by Lavery et al. (2013) and Miyajima et al. (2015), resulting in incomplete reporting of seagrass carbon storage in the Indo-Pacific.

BRUV literature searching returned only two papers using BRUV sampling in tropical seagrass meadows (Jones *et al*. 2018 , Esteban *et al*. 2018) Which is much lower than might be expected considering this areas reliance on fisheries (Cullen et al., 2007; Unsworth and Cullen, 2010) and the high biodiversity of the area. This may be due to BRUV being a relatively new methodology (Lowry *et al*. 2012). Future studies into this area may wish to include alternative sampling methods if ways to limit bias are considered. This data were supplemented with unpublished data from Clayton (2006) and Gourlay (2017) and new data from Green Island, Great Barrier Reef Australia. It shows that there are relatively few BRUV studies in the region. There were only five data points from deep water seagrass meadows, all of which were in the same archipelago of Chagos. These showed much higher results across fisheries categories (diversity, trophic score, IUCN redlist status score and the presence of commercial and subsistence species) than other environment types. However, these higher values were also likely strongly influenced by study sites location as Chagos is highly remote (Andradi-Brown et al., 2019) as no other data were available for this environment type, deep water meadows were excluded from my scoring system due to the bias of the sampling location. However, the Cambodian meadows where included despite a low n value of 6 as these where coastal meadows meaning they could be compared with meadows of the same environment type from other countries reducing the bias of location. No BRUV sampling was carried out in estuarine meadows, likely due to poor visibility and high flow rates. However, Gilby et al. (2017) successfully deployed BRUVS in high visibility estuaries in Australia, so it may be possible in some Indo-Pacific estuaries. However, this relatively small number of studies gave us data from 357 BRUV samples representing lagoon, coastal, reef and deep water meadows. BRUV data were only found representing seven countries and one territory in the Indo-Pacific, highlighting the need for further data collection to understand ecosystem services in the region fully. However, I have collated sufficient data to assess the variation of ecosystem services between, Lagoon, Reef and coastal sights for all metrics used, plus deep water and estuarine carbon storage.

4.a IUCN redlist status score

The IUCN redlist status score evaluates how endangered the meadow's fish population is, allowing easy comparison between sites. IUCN redlist status scoring was born of the notion that people tend to value rare animals more, particularly characteristic megafauna. For example, Sawfish (*Pristidae*) hold cultural importance in Guinea-Bissau (Leeney and Poncelet. 2013). These rare megafauna, such as sharks and rays, are often listed by the IUCN redlist at a higher level, i.e. vulnerable or endangered (Camhi et al., 2009). IUCN redlist status score is used as a proxy for the cultural value of these species presence in an ecosystem.

Additionally, these species typically have many other benefits to ecological functions, such as altering herbivorous species' behaviour preventing trophic cascades (Haggerty et al., 2018). Alternatively, in a more utilitarian perspective, the presence of large marine megafauna, many of which are endangered, can have a large influence on the success of ecotourism such as diving and snorkelling (Anderson et al., 2011; Anderson and Waheed, 1998; Dearden et al., 2008; Gallagher and Hammerschlag, 2011). As a result, these organisms can provide work for many local people (Garrod and Wilson, 2004), having a significant impact on their way of life.

4.b Why BRUV?

BRUV was selected as the methodology to assess fish populations as it is non-destructive and relatively inexpensive to implement, requiring few in field man hours. While the current database for BRUV samples in the Indo-Pacific is limited, this could be fairly rapidly addressed with relatively low research costs as identification and analysis does not need to occur in the field and multiple samples can be taken simultaneously. While BRUV is not perfect, the bias it possesses is towards predatory species, which are generally underrepresented in assemblage studies. These species are also often indicators of healthy ecosystems and are used as umbrella species to protect habitats (Lindenmayer & Westgate 2020). This is because these predatory species cannot survive without an abundance of prey which a healthy ecosystem provides (Osgood et al., 2020).

Although BRUV data are biased towards carnivorous and omnivorous species due to the bait used being primarily oily fish (Langlois et al., 2010). These species are typically lost at an accelerated pace in over-exploited systems (Unsworth et al., 2015, 2014), so this was considered not to be problematic as the presence of these species would indicate a healthy system. BRUV sampling has also been found to be biased against: highly camouflaged, bottom-dwelling species or nocturnal species as they are hard to see on camera (Unsworth et al., 2014). However, BRUV does overcome the avoidance bias of observers or diver operated video methods (Mallet et al., 2014).

4.c Why SeaAroundUs?

Food and agriculture organisation (FAO) collects data globally on fisheries landings (Zeller *et al*. 2016). While this data are useful, it is not ideal for this study for multiple reasons: Firstly, the data are presented in 19 large zones (Zeller *et al*. 2016), these may miss some of the smaller scale variations between countries. Secondly, the FOA data lacks detail taxonomically as the data are compiled into broad taxonomic groups. Greater detail is required to understand ecological processes than can be provided by this data (Zeller *et al*. 2016). Thirdly the FAO data does not include unreported catches. Contrary to the FAO data, SeaAroundUs data are reconstructed from reported catch data along with estimations of unreported catch data giving a clearer picture of the actual catch rates (Pauly and Zeller,2016).

4.d Categorising by environment

This study shows that ecosystem services do vary with environment type, at least in the case of seagrass. Seagrass meadows in lagoons, for example, show higher carbon storage than those situated near reefs or on exposed coastlines. Contrarily, Seagrass meadows associated with reef meadows had higher diversity, higher trophic and IUCN redlist status scores and more subsistence and commercial fish. This variation can be attributed to a variety of factors linked to its location within the landscape and seascape.

The meadow location can affect the amount of runoff and, consequently, the water quality, nutrient availability, light (Alcoverro et al., 2001; Paulson and Simpson, 1977), as well as the connectivity, salinity (Hill and Webb, 1958) and other factors that can change the ecosystem services provided. Meadow depth and water quality can affect the light (Lee et al., 2007) and nutrient availability (Zhang et al., 2017) for the seagrass and the salinity, particularly in areas of high rainfall or temperature (Behara et al., 2019; Katsaros and Buettner, 1969). Wave action can also limit light availability by increasing turbidity (Anthony et al., 2004; Lee and Folkard, 1969), particularly in areas with sandy and muddy sediments where seagrass meadows are often found (van Katwijk et al., 2010). Lower levels of irradiance have been found to reduce seagrass cover and growth (Peralta et al., 2002). Additionally, seagrass shows a preference for areas with reduced wave action for new meadows' settlement (Koch et al., 2006; Barbier et al., 2011). The most important factors affecting organic carbon storage are productivity (Kirwan and Mudd, 2012), the amount of carbon available for burial and the rate of carbon decomposition in the sediment (Ricart et al., 2017), all of which are strongly affected by the irradiance and amount of movement in the water column. As lagoons are largely shallow and sheltered from wave action the light availability is likely to be higher than other meadow types and exportation rates lower which may explain why lagoon meadows where found to have higher carbon storage than other meadow types. Similarly, lagoons are more cut off from the surrounding seascape than other meadow types altering the exportation and importation rates of carbon by animals through herbivory on the seagrass by key groups like parrotfish (Scaridae), with distance being an important factor (Swindells et al., 2017) and therefore increasing exportation to neighbouring systems and reducing production in interconnected meadows. Honda et al. (2016) found that these movements within the seascape are not limited to juvenile fish because adult fish also regularly move between seagrass and coral reefs. This, in turn, allows seagrass to help maintain diversity on coral reefs (Unsworth and Cullen 2010). Coral reef species are known to utilise nearby habitats in the seascape (Sambrook et al., 2020) and so are likely to increase the diversity of nearby seagrass, as seen in my data.

4.e Ecosystem service variation

Environment types showed significant variation in the levels of ecosystem services provided, with Lagoons showing significantly higher organic carbon storage than deep water or estuarine environments (Figure 5). This can likely be attributed to the differing characteristics of lagoons. Lagoons are sheltered environments that typically have higher levels of fine sediment particles and larger organic carbon stores than exposed environments (Samper-Villarreal et al., 2016; Van Keulen and Borowitzka, 2003). Particulate organic matter can show levels up to five times higher in lagoons than oceanic water (Charpy et al., 1997) and show high residence times (Serra et al., 2021). Lagoons were noted by Alongi (1998) as one of the more productive ecosystems on the planet. This paired with high levels of particulate carbon in the water column, which seagrass enhance the settlement of (Duarte et al., 2013; Hendriks et al., 2008), can result in high carbon storage seen in my data set. They have also been noted to be of particular vulnerability to human impacts due to their proximity to terrestrial systems (Pérez-Ruzafa and Marcos, 2012), and so these large carbon stores must be well protected.

Reef and coastal associated meadows also had relatively high levels of carbon storage which where notably significantly higher than those of deep-water and estuarine meadows. Implying that a combination of irradiance and particle residence time may be the most significant factors influencing carbon storage in this region. Lagoon, coastal and reef meadows are typically situated in shallower waters that have good light availability (Ackleson, 2003), these waters are often more sheltered (Ferrario et al., 2014). Conversely, deep water and estuarine meadows both typically have lower irradiances due to depth and murky water, respectively (Gameiro et al., 2011; Schrameyer et al., 2018; Wofsy, 1983). Deep and turbid water seagrasses tend to be smaller with less complex structures and lower rates of photosynthesis, therefore requiring less light (Ooi et al., 2011). Larger species have been found to store more carbon than small species due to higher canopy densities and more complex root structures (Rozaimi et al., 2013) and so it would be expected that sights with lower irradiance such as estuarine and deep water meadows would store less carbon than sights with higher levels of light availability which is seen in my data.

My data showed no significant difference between mixed-species and mono-specific meadows. This contradicts Lavery et al. (2013), who found that species is an important factor in organic carbon storage. However, this might be attributed to my test structure, comparing only mono-specific vs multi-species, not comparing individual seagrass species. Therefore, species may still be an important factor predicting organic carbon storage independently of whether a meadow is mixed species or mono-specific.

Extrapolating the carbon data to 1 m from a sample depth, while a recognised method with strong supporting evidence, will not always give an accurate estimate of the carbon present, particularly if the characteristics of Indo-Pacific meadows are largely different from those worldwide. This is a particularly prevalent issue for those studies which only sampled to a shallow depth. Fourqurean et al. (2012) stated an average soil organic carbon value for Indo-pacific meadows of 23.6 ± 8.3 Mg C ha^{-1} with an n value of 8 and a global average of 194.2 ± 20.2 Mg C ha^{-1} with an n value of 89. This study found that the mean value of carbon in the top 1m of soil in this region is 67.15 Mg C ha^{-1} with an n value of 68. This is much higher than stated in Fourqurean et al., 2012 for the region; however, more research has been carried out in this field since 2012. On the other hand, this is still much lower than the global average, implying that seagrass meadows in the Indo-Pacific store less carbon than meadows in other regions. An important factor in this may be temperature which is higher in tropical regions such as the Indo-Pacific and can cause increased remineralisation of organic carbon stored in seagrass sediments reducing long term storage (Pedersen et al., 2011).

Whereas reef meadows showed higher scores for diversity, trophic score, IUCN redlist status score, commercial species presence and subsistence species present, indicating the importance of incorporating ecosystem services into the decision-making process. These higher scores were found to be significantly higher than those in lagoon or coastal meadows. However, lagoon and coastal meadows were not significantly different in these categories. Coral reefs are widely regarded as diversity hotspots (Hughes et al., 2002; Roberts et al., 2002), so it is not unexpected that seagrass meadows associated with them would have more diverse and richer fish assemblages than other environments. It is unexpected that lagoons would score low on fish assemblage metrics, as they have been said to have some of the highest fishery yields (Kapetsky, 1984) and implies lagoon fisheries may have declined in the last few decades.

High scores for reef meadows on trophic scoring imply that more predatory species such as Sharks (*Carcharhinus*), Jacks (*Carangidae*), and Barracuda (*Sphyraena*) use seagrass meadows for nursery and feeding purposes depending on their life stage (Ferreira et al., 2017; Unsworth et al., 2009). Conversely, lagoon and coastal meadows typically score lower in this metric. Shallow water sites like lagoons have higher variation in abiotic factors such as salinity and turbidity (Solidoro et al., 2010), affecting their habitability. An alternative explanation for this shift is the overharvesting of species; meadows in nearshore sites such as coastal or lagoon sites may be harvested more extensively than reef meadows as they are more accessible (Baker et al., 2015). Many seagrass assemblages in South-East Asia suffer from overharvesting which causes high trophic level species to be lost first (Jackson et al. 2001; Lotze and Milewski 2004). This can be noticed rapidly due to 'trophic skew' in the ecosystem (Duffy, 2003), as species at high trophic levels are typically lost at a higher rate (Dobson et al., 2006). It is likely a compounded effect of both abiotic and anthropogenic effects causing these variations in the trophic scores. However, marine ecosystems can show some resilience to the adverse effects of this as 'functionally redundant' species can occupy niches when species are lost, reducing the effect (Peterson et al. 1998). These fill-in species are unlikely to fill the same role as those lost, minimising trophic cascades potential but not preventing them entirely.

4.f Ecosystem service score

The scorecard system was kept simplistic to ensure ease of use and allow the addition of new data as it becomes available. My scorecard has similarities to other studies such as Nuttle (2013); however, a decile system was opted for to remove some of the subjective nature of many scorecards which categorise things as ‘poor’ or ‘very good’, which can vary with the opinions and experience of the observer.

The scoring system is easy to implement once data are collected on the ecosystem to be assessed. For example, Figure 7 shows the scoring system implemented on two meadows from my data set; one coastal meadow from Zanzibar and one reef meadow from Australia. The coastal meadow scores very highly on the presence of subsistence and commercial species but mid-level on organic carbon storage, IUCN redlist status score and trophic score. However, diversity in this meadow was low for a coastal meadow, which is unlikely to be accounted for in a monetary system. Conversely, a reef meadow in Australia scores very highly on organic carbon storage but low on diversity and trophic with very few commercial or subsistence species present. The fisheries scores for this reef meadow are all relatively low except for the IUCN redlist status score, which implies that despite the fisheries' values being low, the number of species who scored highly on the IUCN redlist is still average reef-associated meadows. They may not be considered in a monetary scale looking at commercial fisheries, which scored a zero for this reef meadow as the benefit of these species is often valued indirectly, i.e. through ecotourism (Nunes and van de Burgh, 2001). This figure shows how the scoring system may rapidly compare meadows for multiple characteristics that can provide both monetary and socioeconomic value. This also highlights the importance of using multiple characteristics to value ecosystem services. For example, if the example meadows were only valued on their organic carbon storage, the reef meadow may be valued much more highly than the Zanzibar coastal meadow. However, the Zanzibar coastal meadow has high levels of commercial and subsistence species present and is likely highly important to local fisheries. This is particularly important in the Indo-Pacific as there is a large reliance on coastal systems for food in the Indo-Pacific (Cullen et al., 2007; Unsworth and Cullen, 2010).

Figure 8 shows how these score can be incorporated into the categories presented in the Millennium ecosystem assessment (MA 2005). Scores which represent the same type of service i.e. provisioning are averaged to give one score out of 10 for the category

Averaging the scores in each category prevents the imbalance of having larger amounts of data representing one category than the other, for example, provisioning services that have commercial and subsistence species data representing it, whereas for regulating services, we only have data on carbon storage. While not changing the overall outcome of the score, the Zanzibar coastal meadow still scores higher than the Australian reef meadow. The difference in the scores is lessened by the averaging of metrics that assess the same type of score for each meadow, i.e. commercial species and subsistence species.

My scoring system allows for the comparison of ecosystem services between environment types on a level playing field. As my system uses empirical data, it allows comparison independent of the observer's opinion or bias. My scoring system is also comprised of a larger number of scores, I.e. 0-10 rather than 1-5 as in Nuttle 2013 allowing more variation to be shown. My scoring system is relatively independent from whether the meadow is subject to anthropogenic effects such as pollution as it is data based. However, it may highlight the effects of pollution and other anthropogenic effects through potentially reduced scores.

This template for ecosystem scoring could be applied to seagrass outside the Indo-Pacific using the same or similar categories however looking into tropical seagrass in the Indo-Pacific was useful as a case study to create the system. While the limited data availability in particularly for subsistence fisheries may have affected some of the conclusions that can be drawn from the data, it allowed for the creation of the scoring system, which is applicable to much larger data sets that may be found in less specific ecosystems. This scoring system could be applied to other ecosystems which are not necessarily marine such as lakes or forests, using appropriate scoring metrics.

5. Conclusion

This study on tropical seagrass meadows in the Indo-Pacific shows the ecosystem services provided vary between different environments, which is similar to findings by other studies such as Lavery et al. (2013) and Nordlund et al. (2018). For example, lagoon meadows showed significantly higher carbon storage than other environment types for this region. However, no significant difference between mono-specific and mixed-species meadows in this region was found, which is contrary to Lavery et al. (2013). This lack of difference may be attributed to a lack of data on individual species, as many studies did not state the dominant species of seagrass within the meadow studied or studied mixed species meadows. Reef meadows in this region show significantly higher scores for metrics based on fish assemblage data than coastal or lagoon meadows, likely due to use by species associated with reefs (Sambrook et al., 2020) which are a diversity hotspot (Hughes et al., 2002; Roberts et al., 2002). Similarly, deep water meadows, although studied very little currently, may present high ecosystem service values in this area.

Conversely, lagoon meadows showed significantly higher carbon storage than other types of meadows within the Indo-Pacific region, indicating they may be particularly important carbon sinks. I have created a sound valuation system for ecosystem services which will allow the addition of more data as it becomes available with relative ease. My system allows ecosystems such as cultural value, which can be overlooked in monetary valuation (de Groot et al., 2002; MA 2005), to be incorporated on an equal level to those which do provide large financial benefits, such as commercial fisheries. Additionally, my scoring system is not subjective but data based, preventing the variance or bias created by observers in more subjective scoring systems such as Nuttle (2013), which uses vague categories such as “good” and “poor” which can vary between observers. My system also incorporates a larger range of scores allowing more variation to be captured. Many previous systems have been difficult to operationalise (Kelemen et al., 2014). However, if appropriate data are collected, the system presented here could be used for terrestrial, freshwater or marine systems, allowing comparisons between them more easily. Future research should focus on collecting more BRUV data from seagrass meadows in the Indo-Pacific to allow a more complete picture of the fish assemblage variation between environments, particularly to allow useful comparisons with deep water meadows and other environments independent of variation between countries.

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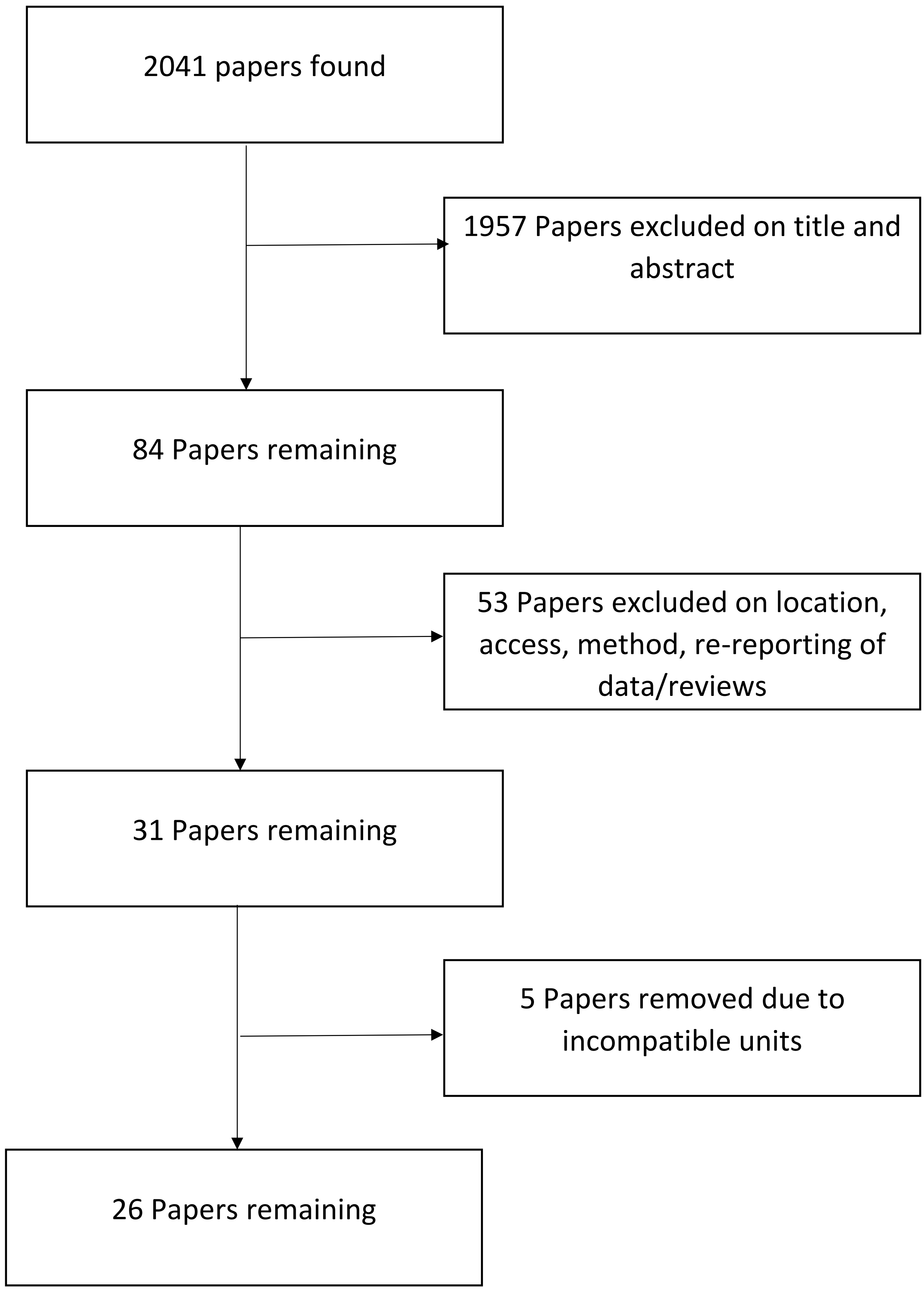
Appendix I List of countries and territories used in literature search

Indo-pacific
Australia
Fiji
Papau new guinea
Indonesia
Malaysia
Thailand
India
Pakistan
South Korea
Japan
Tanzania

Kenya
Oman
Yemen
China
Sri Lanka
Bangladesh
Mauritius
Madagascar
Brunei
Cambodia
Vietnam
Myanmar
Taipei
Solomon Islands
Guam
Hong Kong
Palau
Yap
East Timor or Timor-Leste
Maldives
Phillipines
Seychelles
Mauritius
Reunion
Mayotte
Comoros
British indian ocean territory
Cocos (Keeling) Islands
UAE
Saudi

Appendix II Filtering process of carbon storage papers



Appendix III Results of PERMANOVA analysis comparing between environments and countries of origin

Appendix table 1. Outcomes of PERMANOVA tests on comparing parameters between environment types *** indicates a significant result

PERMANOVA model	Df	Sum sq	F model	R^2	Outcome
Trophic score Residuals Total	2 355 357	7.825 49.931 57.756	27.816	0.13548 0.86452 1.00000	P=0.001***
Status score Residuals Total	2 355 357	7.825 49.931 57.756	27.816	0.13548 0.86452 1.00000	P=0.001***
Commercial species Residuals Total	2 355 357	10.647 48.395 59.043	39.052	0.18033 0.81967 1.0000	P=0.001***
subsistence species Residuals Total	2 355 357	3.180 58.242 61.421	9.6905	0.05177 0.9482 1.00000	P=0.001***
Biodiversity Residuals Total	2 355 357	1.2717 12.1306 13.4023	18.608	0.09489 0.90511 1.00000	P=0.001***

Appendix table 2. Outcomes of PERMANOVA tests on comparing parameters between countries of origin *** indicates a significant result

PERMANOVA model	Df	Sum sq	F model	R^2	Outcome
Trophic score Residuals Total	6 351 357	19.047 38.709 57.756	28.786	0.32979 0.67021 1.00000	P=0.001***
Status score Residuals Total	6 351 357	13.232 24.389 37.621	31.738	0.35171 0.64829 1.00000	P=0.001***
Commercial species Residuals Total	6 351 357	15.014 44.028 59.043	19.949	0.2543 0.7457 1.0000	P=0.001***
subsistence species Residuals Total	6 351 357	8.475 52.946 61.421	9.3645	0.13799 0.86201 1.00000	P=0.001***
Biodiversity Residuals Total	6 351 357	4.1696 9.2327 13.4023	26.419	0.31111 0.68889 1.00000	P=0.001***

Appendix IV Carbon storage ecosystem scores for Estuary and Deep water meadows

Deep water	Estuary	Ecosystem service score
<9.145	<0.688	0
9.145-18.032	0.688-7.190	1
18.033-22.476	7.191-13.692	2
22.477-26.920	13.693-20.195	3
26.921-31.364	20.196-26.697	4
31.365-35.808	26.698-33.199	5
35.809-39.701	33.200-39.701	6
39.702-40.252	39.702-46.204	7
40.253-44.696	46.205-52.706	8
44.697-49.140	52.706-59.208	9
49.140<	59.208<	10

Appendix V Papers used for Carbon storage data

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Appendix VI BRUV data sources

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