Opportunities for seagrass research derived from remote sensing: a review of current methods

Bijeesh Kozhikkodan Veettil [1,2], Raymond D Ward [3,4], Mariana Do Amaral Camara Lima [3], Milica Stankovic [5], Pham Ngoc Hoai[6], Ngo Xuan Quang [7,8,*]

[1] Institute of Fundamental and Applied Sciences,
Duy Tan University, Ho Chi Minh City, 700000, Vietnam
Email: bijeeshkozhikkodanveettil@duytan.edu.vn

[2] Faculty of Information Technology,
Duy Tan University, Da Nang, 550000, Vietnam

[3] Centre for Aquatic Environments,
University of Brighton, Moulsecoomb, Brighton, BN2 4GJ, United Kingdom
Email: R.D.Ward@brighton.ac.uk

[4] Institute of Agriculture and Environmental Sciences,
Estonian University of Life Sciences, Kreutzwaldi 5, EE-51014, Tartu, Estonia

[5] Excellence Centre for Biodiversity of Peninsular Thailand,
Faculty of Science,
Prince of Songkla University, Hat Yai, Songkhla, 90110, Thailand

[6] Faculty of Natural Sciences,
Thu Dau Mot University, Thu Dau Mot City, Binh Duong, Vietnam
Email: hoaipn@tdmu.edu.vn

[7] Environmental Quality, Atmospheric Science and Climate Change Research Group,
Ton Duc Thang University, Ho Chi Minh City, Vietnam

[8] Faculty of Environment and Labour Safety,
Ton Duc Thang University, Ho Chi Minh City, Vietnam

*Correspondence: ngoxuanquang@tdtu.edu.vn

Abstract: Seagrass communities provide critical ecosystem and provisioning services for both human populations and a wide range of associated species globally. However, it has been reported that seagrass area is decreasing at a rapid rate in many parts of the world, mostly due to anthropogenic activities including global change (pollution and climate change). The aim of this review article is to highlight the range of current tools for studying seagrasses as well as identify the benefits and limitations of a range of remote sensing and traditional methodologies. This paper provides a discussion of the ecological importance of seagrass meadows, and recent trends and developments in seagrass research methods are discussed including the use of satellite images and aerial photographs for seagrass monitoring and various image processing steps that are frequently utilised for seagrass mapping. The extensive use of various optical, Radar and LiDAR data for seagrass research in recent years has also been described in detail. The review concludes that the recent explosion of new methods and tools available from a wide range of platforms combined with the recent recognition of the importance of seagrasses provides the research community with an excellent opportunity to undertake a range of timely research. This research should include mapping the extent and distribution of seagrasses, identifying the drivers of change and factors that confer resilience, as well as quantification of the ecosystem services provided. Whilst remotely sensed data provides an important new tool it should be used in conjunction with traditional methods for validation and with a knowledge of the limitations of results and careful interpretation.

Keywords: Submerged marine vegetation, Coastal ecosystems, Marine environment, Coastal management.

1. Introduction: Seagrass communities and their ecological importance

Seagrass meadows are regarded as some of the most productive ecosystems, together with salt marshes, mangroves and coral reefs (Costanza et al. 1998; Ward et al. 2016a; Veettil et al. 2018a,b). Seagrasses occupy lower elevation coastal areas than salt marshes and mangroves (Short et al. 2007; Fortes 2012; Lima et al. 2020), and as a result are more frequently inundated, with seagrasses typically located in lower intertidal or subtidal zones.

Seagrasses have a wide global distribution, covering temperate and tropical coastlines across 6 different bioregions, namely: 1. Temperate North Atlantic (from North Carolina, USA to Portugal); 2. Tropical Atlantic (both tropical coats of Atlantic, Caribbean Sea, Gulf of Mexico, Bermuda and the Bahamas); 3. Mediterranean (Mediterranean Sea, the Black, Caspian and Aral Seas and northwest Africa); 4. Temperate North Pacific (From Korean Coast to Baja, Mexico); 5. Tropical Indo-Pacific (tropical Australia to the eastern Pacific, East Africa, and South Asia); 6. Temperate Southern Oceans (New Zealand, temperate Australia, South America and South Africa) (Short et al. 2007) (**Figure 1**). Seagrass species diversity varies with region, with the highest diversity found in South East Asia (Green and Short 2003). Two environmental variables that strongly influence the distribution of seagrasses are sea surface temperature (SST) and salinity (Chefaoui et al. 2016). Other factors include turbidity, water currents, solar radiation, nutrients, oxygen and sulphides (McMahon et al. 2014; Glasby et al. 2015). The distribution of seagrasses in different climate conditions can be species dependent (Short et al. 2007). For example, key seagrass genera along the tropical coastal areas are: *Cymodocea*, *Enhalus*, *Halodule*, *Halophila*, *Syringodium* and *Thalassia*; whereas *Amphilobis*, *Phyllospadix*, *Posidonia* and *Zostera* dominate in temperate regions; and *Thalassodendron* and *Ruppia* have a global distribution (Short et al. 2007). Due to the influence of various environmental factors on seagrass distribution, these species can be considered as excellent bio-indicators of climate change and ecosystem health (Foden et al. 2013; Fourqurean et al. 2012; Marba et al. 2013).

Seagrasses play an important role as primary producers, as well as providing a habitat and breeding ground for many marine animals including shrimp, sea urchins and clams, and endangered species, such as turtles and marine mammals (e.g. dugong and manatee), and migratory birds (e.g. *Egretta* spp.) (Bujang et al. 2006; Short et al. 2007; Fortes 2012). Seagrasses can also act as a buffer between land and marine environments, filtering nutrients and contaminants from the water column, and reducing turbidity by slowing currents and trapping suspended sediment particles (Fortes 2012; Potouroglou et al. 2017). Seagrasses also play an important role in climate change mitigation through rapid rates of carbon sequestration and high carbon storage capacity, which can be locked away for millennia (McLeod et al. 2011; Duarte et al. 2013; Duarte and Krause-Jensen 2017).

In addition to their role as carbon sinks, seagrass meadows have historically provided numerous ecosystem services, directly or indirectly, dating back to the 16th century (Campagne et al. 2015; Cullen-Unsworth et al. 2014; Nordlund et al. 2016, 2017). A few examples of these come from centuries old records of seagrass litter being used as bedding, straw substitutes for thatching stoned roofs in Scotland, and even in agriculture (Urquhart 1824; Willis 1983; Terrados and Bodrum 2004; Nordlund et al. 2016). Furthermore, small-scale fisheries are largely dependent on seagrass communities and their role as nursery areas for various economically valuable fish species and marine invertebrates (Torre-Castro et al. 2014; Unsworth and Unsworth 2016; Nordlund et al. 2017; Vonk et al. 2008; Jones et al. 2018).

Despite the high economic value provided by ecosystem services, seagrass area is in global decline due to a range of factors including alterations in coastal habitat (Micheli et al.

2008), eutrophication (Burkholder et al. 2007), invasive algal or plant species (e.g. *Caulerpa taxifolia*) (Short et al. 2007), invertebrate grazing (Statton et al. 2015), sea level rise (Saunders et al. 2013; Garner et al. 2015), climate extremes (Arias-Oritz et al. 2018) and climate change (Duarte et al. 2018; Lima et al. 2020) and regional climate impacts such as high energy storms (Orth et al. 2006a; Duarte et al. 2008; Waycott et al. 2009). Seagrass decline is one of the factors accelerating the broader degradation of marine habitats around the world (Waycott et al. 2009). Due to the high importance and value of seagrass ecosystem services to global biodiversity, human well-being and climate change mitigation, it is crucial to understand and acknowledge research methodologies for evaluating the location, extent and ecosystem health and benefit of seagrass systems. To date, seagrass monitoring methods have primarily used in-situ approaches including SCUBA/snorkeling surveys (Gotceitas et al. 1997), ground-based sampling (Moore et al. 2000), and hovercraft-based mapping (McKenzie 2003). More recently, active and passive remote sensing approaches have been introduced to estimate the cover and quality of seagrass habitats (Duffy et al. 2018). Other methods, such as active acoustic remote sensing using side scan sonar, have also been deployed to quantify seagrass meadow cover (Barrell et al. 2015; Hossain et al. 2015b), whilst passive spectral sensors on-board platforms such as satellites or light aircraft have proven useful to quantify seagrass meadow dynamics (e.g. Baumstark et al. 2016; Cunha et al. 2005). However, remote sensing techniques still have limitations to be overcome, especially regarding spatial resolution, which restrict the focus of studies to identification and mapping of seagrass areal extent. Even using fine spatial resolution satellite data, individual seagrass plants or shoots cannot be detected (Stekoll et al. 2006; Valle et al. 2015; Duffy et al. 2018). For this reason, the development of new and scale-appropriate methods for quantifying and monitoring changes in seagrass ecosystems remain important, to improve the way that drivers of change are understood, and allow for improved management.

The aim of this review article is to provide an overview of seagrass research methods in order to highlight limitations and benefits to using various techniques derived from a range of remote sensing platforms (spaceborne/airborne/UAV/AUV/boat), including passive and active methods such as hyperspectral, multispectral, stereoscopic aerial imagery, LiDAR, RADAR and side scan, and the added benefits of combining these with traditional research methods.

2. The role of remote sensing in seagrass research

For a global level assessment of various terrestrial and marine ecosystems, field surveys are too time consuming and expensive. Remote sensing, particularly using spaceborne datasets, can provide relevant and long-term data for analysing ecosystem changes (Murray et al. 2018). Spaceborne and airborne remote sensing has been widely used for monitoring and mapping seagrass ecosystems throughout the world (Chauvaud et al. 1998; Dekker et al. 2007). Recently, methods for the acquisition and interpretation of optical/acoustic data for the mapping of seagrass habitats have advanced rapidly (Ferwerda et al. 2007; Hossain et al. 2015b). Remote sensing applications on seagrass ecosystems can be primary (detection of seagrasses, spatial coverage, species-level discrimination, biomass detection, growth patterns and degradation) or secondary (environmental variables influencing seagrasses such as SST, salinity, sea-level rise. pollution, detection of epiphytes, etc.) (Chauvaud et al. 1998; Dekker et al. 2007; Ferwerda etal. 2007; Hossain et al. 2015b, 2019).

Compared with terrestrial plant ecosystems, seagrass ecosystems are more dynamic and change significantly over space and time (Frederiksen et al. 2004) and the principle difficulty in seagrass remote sensing (particularly when using passive remote sensing systems) arises from the fact that these are often submerged (Duffy et al. 2018). Despite these difficulties remote sensing surveys of seagrass ecosystems can be undertaken using terrestrial/underwater photography and videography, airborne data (including unmanned aerial vehicles - UAVs), and satellite imagery (Hossain et al. 2015b).

2.1 Principles of seagrass remote sensing

Many direct remote sensing methods are based on spectral reflectance measures of chlorophyll and other constituents in leaves (Qiu et al. 2019). Leaf reflectance characteristics are influenced by surface features, structure and biochemical components of the leaf (Thorhaug et al. 2007). The spectral reflectance of seagrasses and attenuation of the useful portion in the electromagnetic spectrum (mainly visible [400-700 nm] and infrared radiation [1000-2000 nm]) by its surrounding aquatic environment are key factors influencing the quality of seagrass remote sensing (Thorhaug et al. 2006). Electromagnetic radiation (EMR) from the sun (for passive remote sensing) or the sensor (for active remote sensing) undergoes atmospheric scattering and underwater attenuation twice for each medium. Large differences in reflectance in visible and infrared wavelengths are used for discriminating terrestrial plants using optical remote sensing (Borregaard et al. 2000). However, in the case of submerged seagrass communities, the visible wavelength penetrates the water column whereas wavelengths beyond 680 nm undergo significant attenuation (Kirk 1994; Kirkman 1996) and hence the most suitable method while using optical data is to utilize the differences in spectral reflectance within visible wavelengths (Dekker et al. 2007). Both atmospheric correction and estimating the water attenuation coefficient are necessary for mapping seagrass communities using optical remote sensing data (Giardino et al. 2019).

In coastal waters, spectral scattering and absorption by phytoplankton, suspended sediments and dissolved organic substances further restricts the application of remote sensing of seagrasses (Dekker et al. 1992; Giardino et al. 2019). Underwater absorption of electromagnetic radiation is principally caused by phytoplankton, organic and inorganic particulates, dissolved organic materials and water molecules whereas scattering is mainly caused by suspended (organic/inorganic) sediments (Hossain et al. 2015b). Absorption and scattering increase with depth through the water column, causing an exponential reduction in light intensity, which means that the possibility of spectral discrimination of seagrass communities using optical remote sensing data rapidly decreases with depth (Zimmerman and Dekker 2007).

2.2 Types of remote sensing data for seagrass research

A wide range of remotely sensed datasets, including aerial, terrestrial and underwater photography, and satellite imagery have been used for seagrass research in recent decades (e.g. Mumby et al. 1997; Dekker et al. 2007; Hossain et al. 2015b, 2015c; Hossain and Hashim 2019). Both active and passive sensors have been employed for data collection in recent years and used for seagrass research (Ferwerda et al. 2007; Hossain et al. 2015b; Duffy et al. 2018). Various factors dependent on the objective of the study need to be considered when selecting remotely sensed data for seagrass research, including: spatial resolution, spectral resolution (e.g. multispectral and hyperspectral), radiometric resolution, temporal coverage, remote sensing system (active or passive), platform (e.g. terrestrial, underwater, airborne, spaceborne) and ranging techniques (e.g. Radar, LiDAR).

In this paper, we have classified the data used for seagrass research as airborne and spaceborne remote sensing. However, active remote sensing systems (Radar and LiDAR) are discussed separately because distinct acquisition and processing techniques are applied while using these datasets. Furthermore, a separate section is provided to discuss new trends in using UAVs for seagrass research.

2.2.1 Seagrass remote sensing using terrestrial and underwater photography/videography

Terrestrial photographs have been traditionally used for documenting coastal areas (Robbins and Bell 1994). However, due to a number of limitations, very few studies exist on the use of this method for mapping or monitoring seagrass habitats (e.g. Andrade and Ferreira 2011; Gonzalez 2015) compared to underwater photography/videography (e.g. Burdick and Kendrick 2001; McDonald et al. 2006). Terrestrial oblique large-scale photography can be used for cost-effective and repeated coverage over short time intervals, even though ground control points (GCPs) using a GPS tracker are required for image rectification and to create orthogonal views for mapping seagrass meadows by applying a suitable image classification algorithm (Andrade and Ferreira 2011). Recently, Alvsvåg (2017) used photographs from a DSLR camera mounted on an Autonomous Surface Vehicle (ASV) to map seagrass habitats in Hopavågen (Norway).

Underwater photography (**Figure 2**) and videography are widely used for regional-scale seagrass habitat studies, including the interrelationship among organisms (Norris et al. 1997; Burdick and Kendrick 2001; McDonald et al. 2006). Images and/or video can be taken by SCUBA divers or using remotely operated or autonomous underwater vehicles (AUVs) (Armstrong et al. 2006), towed or drop-down video (Andrade and Ferreira 2011). Underwater photography can be used for measuring structural characteristics, such as the number of leaves, leaf length, and shoot density (Borg et al. 2006). Underwater videography can also be used for detecting changes in seagrass cover (e.g. McDonald et al. 2006), characterization of seagrass habitat (e.g. Haag et al. 2008) and species identification in extremely shallow or clear water (Norris et al. 1997). AUVs are very time-effective, since they can be used to measure canopy structure and estimate above-ground biomass using captured stereo imagery (Roelfsema et al. 2015). Furthermore, AUV data can be used for calibrating and validating satellite data (Roelfsema et al. 2015), as AUV data collection can have spatial and temporal consistency, repeatability, and be used in deeper waters. AUVs can also capture stereo imagery to measure canopy structure and to estimate above-ground biomass and can be compared with labour intensive in situ data collection (Roelfsema et al. 2015). Light-weight AUVs are used effectively in seagrass research (e.g. Vasilijevic et al. 2014) and the integration of AUV data with high resolution satellite data (e.g. WorldView-2, IKONOS) can offer high quality, multi-temporal mapping of seagrass at a species level as well as biomass estimation (Roelfsema et al. 2014). Tecchiato et al. (2015) successfully utilised underwater photography integrated with sediment data and geomorphological information to understand the influence of geomorphology and sedimentary processes on seagrass habitat distribution. Further improvements to underwater imaging can be achieved by utilising hyperspectral imagery, which can be used as a substitute for in situ data for species-level identification (Bongiorno et al. 2018; Dumke et al. 2018). Improved classification techniques are being tested for real-time classification of seagrass meadows using underwater photography (e.g. Bonin-Font et al. 2018; Martin-Abadal et al. 2018). A summary of studies that utilise underwater photography and videography is provided in **Appendix 1**.

2.2.2 Seagrass remote sensing using aerial photographs and airborne hyperspectral data

Airborne data, including aerial photographs and videography, have been widely used in seagrass research (Kendrick et al. 2000; Pasqualini et al. 2001; Frederiksen et al. 2004; Lathrop et al. 2006; Fletcher et al. 2009; Young et al. 2010). Both passive (photographs, multispectral and hyperspectral data) and LiDAR data can be obtained from airborne platforms.

Quantitative seagrass mapping using aerial photographs has been widely used (e.g. Ferguson et al. 1993; Hernández-Cruz et al. 2006; Orth et al. 2006b; Fletcher et al. 2009; Cuttriss et al. 2013), mainly because the visible wavelength undergoes less attenuation underwater than other spectra (Kirkman 1996). Simple transformation techniques (e.g. Red Green Blue to Intensity Hue Saturation) and image thresholding can be used to map seagrass areas from aerial photographs (Fletcher et al. 2009). Aerial photographs have also been used for long-term temporal and spatial monitoring of seagrass meadows. For example, Hernández-Cruz et al. (2006) documented decadal spatial changes in seagrass meadows between 1937 and 2000 in Puerto Rico. A number of classification techniques, such as on-screen digitizing (Orth et al. 2006b; Murdoch et al. 2007), principal component analysis (PCA) (Ferrat et al. 2003), multi-scale image segmentation and object-oriented image analysis, have been applied to digital photographs for mapping the extent and density of seagrasses (Lathrop et al. 2006). The accuracy of time-series seagrass mapping using aerial photographs depends on the mapping methods applied (Meehan et al. 2005). On-screen digitizing, even though time consuming and depending on the expertise of the researcher, can provide highly accurate seagrass maps if high resolution aerial photographs are used. Even with low resolution aerial photography, the spatial distribution pattern of seagrasses can be estimated (Robbins 1997), particularly where meadows are not patchy. New image processing techniques, such as linear spectral unmixing, have been applied recently on aerial photographs to improve seagrass mapping for identifying small seagrass patches and masking the bare substrate (Uhrin and Townsend 2016). In shallow marine environments where red light reaches the seabed, digital aerial photographs can also be used to differentiate epiphytes (mostly red/brown) from seagrasses (mostly green) (Mount 2006, 2007). At greater depths a combination of aerial photographs for shallow waters (0-20 m) and side-scan sonar (SSS) for deep coastal regions (20-50 m) is recommended due to the differences in light attenuation depth (e.g. Pasqualini et al. 1998; Leriche et al. 2006). Different types (colour, false colour near-infrared, and black and white) of aerial photographs can also be used for seagrass mapping in coastal areas with varying depths (e.g. Ferguson et al. 1993; Pasqualini et al. 2001; Young et al. 2008, 2010).

Even though aerial photographs can be used to discriminate habitat features at a fine scale, the lack of capacity to record in multiple bands (Hossain et al. 2015b) and dependency on water quality (Hernández-Cruz et al. 2006) are drawbacks to the use of these data. Furthermore, photometric variation inherent in aerial photography is a major source of misclassification when seagrass mapping/classification relies solely on digital image processing techniques (Meehan et al. 2005; Young et al. 201). Airplane-derived aerial photographs are also expensive to acquire, in place of these a few recent studies (e.g. Barrel and Grand 2015) used cost-effective methods such as low altitude, high resolution photographs from balloon-mounted digital camera platforms. More recent studies have used cost and time-effective UAV-aerial photographs for seagrass mapping (See section 2.2.6). However, historical records of seagrasses in many regions (e.g. Rees 1993; Cunha et al. 2005; Meehan et al. 2005) are often only available from aerial photographs, in which an error margin of up to 20% in quantifying the aerial extent of seagrasses may still be acceptable where these are the only data available (Meehan et al. 2005).

Airborne platforms are commonly used for the acquisition of hyperspectral images (e.g. AISA+, CASI-2, HyMap, Ocean PHILLS, PRISM), and these have been increasingly used for: mapping the extent of seagrasses (e.g. Mumby et al. 1997; Dierssen et al. 2003; Bostater et al. 2004; Peneva et al. 2008); species level discrimination (e.g. Ferwerda et al. 2007; Phinn et al. 2008; Dierssen and Russel 2015); estimation of biomass and productivity (Hill et al. 2014); as well as other factors including water column depth, water quality, bottom types (e.g. Garono et al. 2004; Garcia et al. 2015; Hossain et al. 2015b) and spectral separation between seagrasses and other bottom substrates such as algae (e.g. O´Neill et al. 2011; Pe'eri et al. 2016). Compared to the use of a field spectroradiometer, airborne hyperspectral data has the advantage of greater areal coverage. However, for mapping large areas of seagrass meadows, spaceborne multispectral imagery is preferred due to the high cost per unit area for airborne derived imagery (Hossain et al. 2015b). For mapping individual seagrass patches using airborne hyperspectral data, some studies suggest that at least a 3m spatial resolution is required (e.g. Peneva et al. 2008). However, a number of recent studies (e.g. Phinn et al. 2008; Valle et al. 2015) obtained species-level discrimination and biomass estimation of seagrass using CASI and CASI-2 images with a spatial resolution of 4m alone or together with QuickBird-2 and Landsat data. **Appendix 2** lists a number of studies published in the last two decades using airborne hyperspectral data for seagrass studies and the mapping techniques used. A few recent studies have also used cost-effective hyperspectral data from UAV platform (e.g. Uto et al. 2017; Manfreda et al. 2018), which used pattern matching algorithms and vegetation indices for accurate seagrass mapping improving bot resolution and repeatability.

A number of Vegetation indices, including various modified versions of the normalized vegetation index (NDVI), have been developed for the automatic discrimination of seagrass meadows (e.g. Valle et al. 2015). It is possible to use airborne hyperspectral data with proper atmospheric correction to estimate detailed pigment composition of seagrass meadows using vegetation indices, calibrated by hyperspectral field data (spectroradiometer) (e.g. Bargain et al. 2013; Hedley et al. 2016, 2017). Marcello et al. (2018) observed that a combination of hyperspectral and multispectral data increased the robustness and performance, respectively, of seagrass mapping when maximum likelihood and support vector machine (SVM) methods were applied.

Unlike spaceborne data, which has fixed sensor altitudes and orbits, airborne hyperspectral data can have a variety of deployment altitudes and measurement geometry that pose significant difficulties in dealing with atmospheric effects (Castillo-López et al. 2017), particularly for shallow coastal waters with unknown aerosol properties, spatial heterogeneity in the water column, and sensor artefacts (Zhang et al. 2015). Zhang et al. (2016) recommend an iterative atmospheric correction in such cases, for monitoring short-term changes in shallow water environments. However, accuracies in excess of 80% can be obtained without applying any atmospheric correction methods to airborne hyperspectral data (e.g. Zhang et al. 2013). Furthermore, water-depth correction algorithms using water absorption and scattering factors have been found to improve seagrass mapping using hyperspectral data (Lu and Cho 2012).

2.2.3 Spaceborne remote sensing data for seagrass mapping

Spaceborne remote sensing platforms offer a cost-effective option where higher aerial coverage is required, when mapping larger, monospecific and continuous seagrass meadows (Dekker et al. 2007). In recent decades, a large number of sensors with a wide variety of spatial, spectral and temporal resolutions offer various approaches to seagrass mapping. Even though, spaceborne multispectral and hyperspectral data are constrained in highly turbid environments, with limited applications in optically shallow waters, higher spectral resolutions (for hyperspectral data) can reduce the number of mixed pixels and are relatively cheaper than airborne derived data (Hossain et al. 2015b). More wavelengths in the visible region of the EMS can greatly improve seagrass mapping. A summary of the most recent seagrass studies using spaceborne multispectral and hyperspectral data with varying spatial resolution is provided in **Appendix 3**.

A range of high-resolution multispectral spaceborne data is available from the early 1970s and most have the optical channels required for seagrass mapping (Wicaksono et al. 2017). Recently, a number of high spatial resolution (5 m to 50 cm resolution) multispectral spaceborne data, such as SPOT-5 (e.g. Pasqualini et al. 2005), SPOT-7 (Siregar et al. 2018), IKONOS (e.g. Mumby and Edwards 2002; Wabnitz et al. 2008; Howari et al. 2009; Pu and Bell 2017), GeoEye (Chayhard et al. 2018), WorldView-2 (Misbari and Hashim 2014; Reshitnyk et al. 2014; Roelfsema et al. 2014; Adi 2015; Anggoro et al. 2016; Baumstark et al. 2016; Albert et al. 2017; Halls and Costin 2016; Hoang et al. 2016; Koedsin et al. 2016; Manuputty et al. 2016; Martin et al. 2016; Eugenio et al. 2017; Oguslu et al. 2018; Poursanidis et al. 2018), WorldView-3 (e.g. Jadidi and Vitti 2016; Collin et al. 2017), QuickBird (Wang et al. 2007; Albert et al. 2017; Hisabayashi et al. 2018), KOMPSAT-2 (e.g. Kim et al. 2012, 2015; Matta et al. 2014a, 2014b; Choi et al. 2018), RapidEye (e.g. Matta et al. 2014a; Giardino et al. 2016; Li 2018; Traganos and Reinartz 2018a) and PlanetScope (e.g. Traganos et al. 2017; Wicaksono and Lazuardi 2018), have been used for mapping of seagrass meadows around the globe.

However, a large number of studies have used medium resolution satellite imagery, such as Landsat series (e.g. Ferguson and Korfmacher 1997; Dahdouh-Guebas et al. 1999; Meyer

2008; Wabnitz et al. 2008; Roelfsema et al. 2009; Knudby et al. 2010; Meyer and Pu 2012; Pu et al. 2012a, 2014; Blakey et al. 2015; Hossain et al. 2015a, 2015c; Kim et al. 2015; Chen et al. 2016; Millán et al. 2016; Müller et al. 2016; Ayustina et al. 2018; Geevarghese et al. 2018; Hisabayashi et al. 2018; Topouzelis et al. 2018), ASTER (e.g. Pulliza 2004; Castaño-Gallego and Lozano-Rivera 2006; Dahanayaka et al. 2012; Adi et al. 2013a; Wicaksono and Hafizt 2013; Shofa 2014; Kim et al. 2015; Wicaksono et al. 2017), ALOS AVINIR-2 (e.g. Firdaus 2011; Astuti et al. 2012; Adi et al. 2013b; da Silva et al. 2017), CBERS (e.g. Yang and Yang 2009; Yang and Huang 2011), and Sentinel-2 (e.g. Topouzelis et al. 2016; Fauzan et al. 2017; Hafizt et al. 2017; Thalib 2017; Thalib et al. 2017; Yanuar et al. 2017; Dattola et al. 2018; Fethers 2018; Kovacs et al. 2018; Luo 2018; Traganos and Reinartz 2018b; Traganos et al. 2018a, 2018b; Transon et al. 2018) and these are widely used for seagrass monitoring, particularly where long-term datasets are available. Medium resolution satellite imagery can also be used to estimate carbon stocks in an inexpensive way (Mashoreng et al. 2018). Similar to SPOT 5, ASTER data also lack blue wavelength in the spectrum and hence coastal mapping capacity is limited compared to the data mentioned above (Capolsini et al. 2003), particularly when applying vegetation indices. Medium resolution multispectral imagery must be acquired at low tide for accurate mapping of seagrass meadows.

Although the number of publications is limited, several studies have used low resolution (>100m) spaceborne data, such as MODIS (e.g. Dierssen et al. 2010; Bargain et al. 2012; Adi et al. 2013b; Downie et al. 2013; Barnes et al. 2014a; Petus et al. 2014a, 2014b, 2016, 2018; Tuya et al. 2014; York et al. 2015; Phinn et al. 2017; Beck et al. 2018; Carlson et al. 2018; Champenois and Borges 2018; Perez et al. 2018), AVHRR (e.g. Salas et al. 2000; Amela et al. 2007; Carlson et al. 2018), and MERIS (e.g. Lunetta et al. 2009; Roman et al. 2010; Adi et al.

2013b; Saulquin et al. 2013; Matta et al. 2014b; Roman and Dupouy 2014; Zucchetta et al. 2016), for large-scale seagrass research. These data were mostly used for understanding biochemical, physical and other environmental variables associated with seagrass meadows (e.g. Amela et al. 2007; Madrinan et al. 2010; Peirano et al. 2011; Madrinan and Fischer 2013; York et al. 2015; Phinn et al. 2017) rather than estimating their spatial extent due to the low spatial resolution. Some studies (e.g. Bargain et al. 2012) used vegetation indices such as the ARVI (Atmospherically Resistant Vegetation Index) developed by Kaufman and Tanre (1992) for seagrass studies, which is derived from MODIS data. These vegetation indices were found to be, on average, four times less sensitive to atmospheric effects when compared with NDVI (Bargain et al. 2012).

Although it is more common to use airborne hyperspectral data for seagrass mapping, spaceborne hyperspectral data, such as Hyperion (e.g. Lee et al. 2005; Pu et al. 2010, 2012; Li et al. 2012; Yuan 2012; Meyer 2013; Pu and Bell 2013; Zhao et al. 2013; Kisevic 2015) and Hyperspectral Imager for the Coastal Ocean (HICO) (Cho et al. 2013, 2014, 2016; Garcia et al. 2014a, 2014b, 2015; Adi 2015; Huang and Cho 2016; Jay et al. 2018), have been employed for detailed spectral characterization of seagrass meadows as well as the assessment of surrounding environments. NASA's ongoing HyspIRI mission is also expected to have a number of applications in seagrass monitoring and mapping (Lee et al. 2015) with its 30 m spatial resolution. A key limitation of the available hyperspectral data is their limited application in highly turbid environments and/or deeper ocean areas (Hossain et al. 2015b).

2.2.4 Seagrass studies using LiDAR, Laser scanners and Sonar

In recent years airborne LiDAR and terrestrial laser scanners (TLS) have been used to detect alterations to seagrass habitat characteristics, particularly 3D analysis (e.g. Hannam and Moskal 2015; Corbi et al. 2018; Collings et al. 2019). Although costs are typically higher than most satellite data, bathymetric LiDAR has good capabilities for high resolution seagrass mapping due to its greater water penetration compared to higher wavelengths. Recently, sound navigation and ranging (Sonar) systems, including multi-beam echosounders, have also been employed for mapping and classification of seagrass meadows (e.g. Komatsu et al. 2003; Asada et al. 2005; Lefebvre et al. 2009; Hamana and Komatsu 2016), even though complex post-processing algorithms are required for data extraction. Different algorithms for processing LiDAR data for seagrass mapping include maximum likelihood classification (Tulldahl and Wikström 2012) and object-oriented image classification (Parrish et al. 2016). Parrish et al. (2016), using object-oriented methods, obtained a user's accuracy of 100% and producer's accuracy of 82% in eelgrass mapping using LiDAR data. Using airborne Hawk Eye LiDAR, even with the limited application in high turbidity areas, Chust et al. (2010) obtained an accuracy between 84.5% and 92.1% in coastal habitat mapping. LiDAR data taken from a tripod/vehicle/boat/underwater can also be used for small-scale seagrass surveys (Hannam 2013; Hannam and Moskal 2015; Corbi et al. 2018). Terrestrial Laser Scanning devices provide an accurate method for monitoring coastal areas that are subjected to erosion characterized by the accumulation of seagrass berm (Corbi et al. 2018).

Airborne LiDAR data, usually taken from a helicopter, aeroplane or drone, have been widely used for bathymetric surveys to identify the location and extent of seagrass meadows (Brock et al. 2006; Wang and Philpot 2007; Chust et al. 2008, 2010; Valle et al. 2011, 2014; Collin et al. 2012; Tulldahl and Wikström 2012; Pan et al. 2014; Zavalas et al. 2014; Ishiguro et al. 2016; Parrish et al. 2016; Webster et al. 2016; Webster 2017; Collings et al. 2019). Chust et al. (2008) observed higher accuracy in coastal habitat mapping using LiDAR data compared to multispectral imagery. Full waveform bathymetric LiDAR can also be used to classify different types of seagrasses (Pan et al. 2014). A combination of both LiDAR and multispectral data (e.g. WorldView-2) further improves the accuracy of coastal habitat mapping (Chust et al. 2008; Collings et al. 2019). Zavalas et al. (2014) used a combination of LiDAR and underwater videography for mapping marine algal and seagrass communities. Webster et al. (2016) and Webster (2017) used a MPIX RCD30 camera, which can be used to capture NIR and RGB imagery co-aligned with LiDAR sensor (500 kHz for NIR and 35 kHz for Green) and can be directly georeferenced. Multispectral LiDAR data has been proved its capabilities in assessing structurally complex coastal habitat (Collin et al. 2012). Airborne LiDAR surveys have also been used to map seagrass meadows studded with coral reefs (Brock et al. 2006) as well as for modelling suitable habitat for seagrass meadows (Valle et al. 2011). Ishiguro et al. (2016) used a combination of airborne LiDAR bathymetry data and aerial photographs for the successful classification of seagrass meadows. Such studies using LiDAR data have helped the scientific communities in understanding anthropogenic and climatic stresses on seagrass communities (Al-Nasrawi et al. 2018) and projecting future distribution of seagrasses under global warming and sea level rise (Valle et al. 2014). O´Hare et al. (2018) and Ventura et al. (2018) mentioned that UAV's with LiDAR sensors have a high potential to improve plant-sediment studies in aquatic environments, further developing on those in other coastal environments (Chadwick 2011; Ward et al. 2013, 2016b).

2.2.5 Seagrass remote sensing using Radar imagery

Radar remote sensing has been used in a small number of studies, particularly for the study of seagrass bed structure (Adolph et al. 2018; Gade et al. 2018). Although general detection of seagrasses was nearly impossible, the elevated structures of sediments induced by the seagrass cover can be mapped by the diffusely elevated backscatter values of Synthetic Aperture Radar (SAR) (Adolph et al. 2018). Seagrasses enhance radar backscattering due to high local surface roughness (even though they lay flat on the ground at low tide) (Gade et al. 2018), and it is possible to apply a combination of structural analysis and unsupervised (ISODATA) classification to SAR data for mapping seagrass beds (Adolph et al. 2018). Seagrass meadows produce characteristic surface structures, different from green algae or diatoms, and merging SAR data with multispectral data can be efficient in understanding the surface roughness information and sediment types of seagrass meadows (Adolph et al. 2018). Combining Sentinel-1 SAR with Sentinel-2 time series imagery and using NDVI statistic parameters showed an improvement in seagrass mapping accuracy, with an overall accuracy of 77.7% and Kappa coefficient of 0.75 (Luo 2018). Furthermore, bare soil can be separated from seagrass meadows using SAR such as TerraSAR-X, which provides complimentary information to optical imagery (Dehouck et al. 2011). Available SAR data for seagrass bed mapping include: TerraSAR-X, TanDEM-X, COSMO-SkyMed, Sentinel-1 and Radarsat-2.

2.2.6 Application of UAVs in seagrass research

The use of UAVs for coastal ecosystem research (**Figure 3**), including seagrasses, mangroves, saltmarshes, and coral reefs, is a new trend and a number of recent studies used visible, thermal and infrared cameras on UAVs (Duffy et al. 2018; Konar and Iken 2018; Villoslada et al. 2020). Notable advantages of using UAVs compared to aircraft, ships or satellite platforms, are low operational costs, high operational flexibility and high spatial resolution (Matese et al. 2015). Furthermore, UAVs are operated at a lower altitude and the disadvantage of cloud cover, particularly in the tropics, while acquiring optical imagery can be avoided. Since the images are taken from low altitudes, atmospheric absorption and other effects in a UAV derived image are negligible compared to satellite data (Lomax et al. 2006). Although, images obtained from UAVs have a clear advantage in terms of spatial resolution, the distinction of the species is still only possible for exposed or shallow monospecific seagrass beds (Duffy et al. 2018). An obvious disadvantage for UAV platforms compared to satellite platforms is the lower areal coverage, particularly where meadows have monospecific species composition (high resolution imagery is not required in this case) and detailed structural and morphological features are not required. A list of the most recent seagrass studies utilising UAV platforms is given in **Appendix 4**.

The use of UAV platforms for seagrass mapping and monitoring has become more sophisticated in recent years. Advances in technology are now focussing on light weight sensors with multispectral, hyperspectral and LiDAR systems on UAVs (Uto et al. 2017; Manfreda et al. 2018). Multispectral satellite images combined with data taken from UAV produced high quality maps of seagrass meadow in some cases. For example, Topouzelis et al. (2016) used a combination of Sentinel-2 imagery and UAV-based data for seagrass mapping in Lesvos Island in Greece. To avoid problems with surface water reflections, sun glint correction using polarised filters can be utilised (Muslim et al. 2019) or image acquisition can be done in the morning and evening (Chayhard et al. 2018).

3. Seagrass research methods: traditional vs. remote sensing

Seagrass meadow distribution mapping using remotely sensed data has predominantly been undertaken using aerial photographs or satellite imagery (Ferwerda et al. 2007). However, the previously discussed limitations of remote sensing (i.e. atmospheric effects and underwater attenuation of EMR) must still be overcome.

Species-level discrimination of seagrasses using traditional methods is mainly conducted by physical collection of samples and measuring morphological features (e.g. stem length, flower), which can be costly and time consuming (Short et al. 2007). To undertake this using remote sensing methods requires the inclusion of spectral features of pigments (chlorophyll-a and chlorophyll-b) within visible wavelengths (Fyfe 2003), as the relative concentrations of pigments (and hence the spectral reflectance) vary within species. Few studies (e.g. Fyfe 2003; Pu et al. 2012; Casal et al. 2013) have utilized the differences in photosynthetic pigment content, which in turn results in differences in the spectral reflectance for different seagrass species. One of the difficulties in using this method, is the bias caused by the pigments contained in epiphytes (Zimmerman and Dekker 2007) or other associated plant communities. However, high spectral resolution data can be employed to solve this problem to some extent (Ferwerda et al. 2007). Moreover, spectral reflectance of the same species at different wavelengths may vary with depth and seasonality (Fyfe 2003). This indicates that traditional methods to collect and discriminate seagrass samples must be done at the initial stage to create spectral libraries, which in turn can be used to map species distribution. Spectral libraries can be created using hyperspectral data (e.g. Pu et al. 2012; Casal et al. 2013) or in situ measurements using a spectroradiometer (e.g. Fyfe 2003; Thorhaug et al. 2007).

Robbins and Bell (1994) classified seagrass communities based on spatial structure and pattern at three different levels as: (1) meadow, which has a contiguous areal distribution with varying per cent cover composition, (2) bed, which has a contiguous areal distribution with similar per cent cover composition, and (3) patch, which is a small and discrete clump of seagrass or gap. Compared to species-level discrimination of seagrasses, this information (spatial structure and pattern) can be obtained more easily with multispectral imagery (Lathrop et al. 2006).

Other traditional seagrass research methods include a number of constituents/steps that can be done or improved using remove sensing, such as: seagrass abundance and depth distribution, biomass, growth and production measurement, photosynthetic rates, algal epiphytes, animal associates and seagrass decomposition (Short and Coles 2001; Krause-Jensen et al. 2004). Additionally, a number of environmental parameters used in seagrass research, such as sediment type and water quality of seagrass habitats, can also be evaluated using remotely sensed data (Hossain et al. 2015b).

3.1. Seagrass distribution, cover and biomass

Seagrass meadows can be found in diverse environments with varying salinity, turbidity and depth levels (Short and Coles 2001). Traditional methods of estimating seagrass distribution use pre-recorded grid patterns or a combination of transects and sampling points (McKenzie et al. 2001), which is time consuming with associated high costs (Short and Coles 2001). However, mapping the occurrence of seagrasses can be done accurately with remotely sensed imagery. In fact, seagrass distribution mapping using remotely sensed data has been undertaken since the mid-20th century (Kelly 1980) using photographs taken from balloons, aircraft or a spacecraft, suggesting that a range of acquisition platforms can be used in mapping seagrass parameters (Hossain et al. 2015b). The success of remote sensing methods is highly dependent on the spatial/spectral resolution of the data, accuracy (e.g. accuracy in georeferencing) and mapping methods (Short and Coles 2001). Satellite images with high spatial resolution can be used to map areas occupied by seagrasses more effectively than low spatial resolution data but the spatial coverage of a high resolution data is often lower than that of low or medium resolution data (e.g. single Landsat data scene covers 34,000 km^2 whereas a SPOT panchromatic tile covers only 60km^2. The same problems arise when using aerial photographs – the use of fine spatial resolution aerial photographs requires a large number of frames where large areas are covered (Short and Coles 2001). Seagrass distribution changes have been estimated using NDVI derived from multispectral visible-infrared satellite data (e.g. Landsat, SPOT) calibrated using in situ spectroradiometric data (e.g. Barille et al. 2010).

Hossain et al. (2015b) suggest that depth (e.g. intertidal, shallow subtidal, deep subtidal) and water clarity (clear or turbid) are the principal factors that need to be considered when using remote sensing to map seagrass parameters such as: distribution, percentage cover, species composition and biomass. Data acquisition platform is also an important factor in mapping seagrass parameters: space borne, aerial or boat (Hossain et al. 2015b).

There are many traditional ways of assessing seagrass cover, with the most common method of non-destructive estimation being in-situ visual assessment of percentage cover using predetermined quadrats (Duarte and Kirkman 2001). However, this methodology only includes estimations of cover at specific points and does not evaluate the spatial variation of seagrass cover within the meadow. In recent years, seagrass cover and change detection has been estimated using a range of active and passive remote sensing systems with various resolutions (high - Pu and Bell 2017; Chayhard et al. 2018; Dattola et al. 2018; Hisabayashi et al. 2018; Kovacs et al. 2018; Su and Huang 2019; medium - Yang and Huang 2011; Shofa 2014;

Wicaksono et al. 2017; Ayustina et al. 2018; Dattola et al. 2018; Siregar et al. 2018 and low - Phinn et al. 2017; Beck et al. 2018; Carlson et al. 2018; Petus et al. 2018). Various active boat-mounted acoustic sensors, e.g. SSS, side beam echo sounder (SBES), multi beam echo sounder (MBES), and acoustic Doppler current profiler (ADCP) (Warren and Peterson 2007; Micallef et al. 2012; Montefalcone et al. 2013; Greene et al. 2018; McIntyre et al. 2018; Held and Deimling 2019), have also been used for mapping seagrass cover. Although, many studies have estimated seagrass cover using remotely sensed data, the majority are still reliant on field measurements for estimating shoot density and canopy height (e.g. Gullstrom et al. 2006).

Traditional seagrass biomass estimation involves both destructive and non-destructive sampling of leaves and shoots within a number of quadrats (Short and Coles 2001). Destructive sampling involves actual removal of the above (leaves and sheaths) and belowground (roots and rhizomes) parts of the seagrass within each quadrat. On the other hand, non-destructive sampling can be undertaken using photographs or video images of the quadrats with known biomass to visually estimate biomass (Short and Coles 2001). As these methods involve extensive field work with destructive sampling, newer methods have recently been applied using remote sensing tools. Estimation of seagrass biomass can be done using multispectral/hyperspectral data, particularly using the visible bands (Armstrong 1993; Phinn et al. 2008) and their correlation with the actual biomass of the seagrasses. Aboveground biomass can also be estimated using the quantitative relationship between NDVI from SPOT imagery and dry weight of leaves (Barille et al. 2010).

Recently, a large number of active remote sensing data from aircraft or UAVs have been used for mapping seagrass cover. These airborne sensors are mainly LiDAR (Tulldahl and Wikström 2012; Pan et al. 2014; Zavalas et al. 2014; Ishiguro et al. 2016; Parrish et al. 2016;

Webster et al. 2016; Webster 2017; Collings et al. 2019), hyperspectral sensors like CASI-2, PHILLS and HyMap (Dierssen et al. 2003; Garcia et al. 2015; Valle et al. 2015; Pan et al. 2016; Castillo-López et al. 2017) or photographs (Frederiksen et al. 2004; Lathrop et al. 2006; Fletcher et al. 2009; Young et al. 2010; Uhrin and Townsend 2016).

3.2. Seagrass growth measurement

Growth measurement in seagrasses at different levels (e.g. shoots, whole plant, and population level) is best done by either direct (marking leaves, rhizomes and shoots) or indirect (reconstruction of past growth from plant anatomical patters) methods (Short and Coles 2001). In addition to leaf and rhizome growth, additional calculations, such as stem growth, leaf elongation, root growth, rhizome elongation, and shoot plastochrone interval may be required for detailed growth measurements (Short and Coles 2001). Even though leaf marking methods have been applied to a number of species (e.g. *Cymodocea nodosa*, *Enhalus acoroides*, *Halophila ovalis*), a number of issues were reported related to the growth forms of various seagrasses (Short and Coles 2001). Gaeckle and Short (2001) reported that comparison of leaf marking methods in *Zostera marina* indicated that direct weight measurement of new tissue can introduce significant errors in leaf growth. Furthermore, different growth forms of seagrasses may require different marking methods (Short and Coles 2001). This method demands a number of logistic facilities and hence is expensive in implementation.

Even though traditional methods for quantifying seagrass growth can provide very accurate data, they are highly time consuming and difficult to produce large-scale quantitative maps (Ferguson et al. 1993; Su and Huang 2019). Remote sensing methods can be applied to indirectly measure growth patterns of seagrass meadows as they are modular plants exhibiting clonal growth patterns (Baumstark 2018). Retrospective mapping of seagrass distribution can provide accurate indirect assessment of trends in growth and decline. Time series analysis of datasets, such as the Landsat series, has already proven to be effective in monitoring long-term growth patterns of seagrasses (Lyons et al. 2010; 2013; Pu et al. 2014). For example, Vidyan (2018) used a combination of Landsat data and Nearmap high resolution imagery for estimating the seagrass growth pattern, extent and biomass in Cockburn Sound in Western Australia. Baumstark (2018) used high resolution WorldView-2 data acquired in 2011 and 2013 to estimate seagrass growth near the Indian River Lagoon in Florida. A combination of ALOS satellite imagery acquired between 2008 and 2009 and in situ data collected between 2011 and 2012 have been used by Rustam et al. (2013) for measuring growth rates and productivity dynamics of *Enhalus acoroides* in Pari Island, Indonesia. However, it has to be noted that remote sensing methods can only evaluate dynamic changes in seagrass extent/distribution/growth patterns, actual growth rates cannot be measured.

3.3. Photosynthesis in seagrass meadows

Due to their photosynthetic activity, seagrasses are considered as important producers in marine environment (Duarte and Chiscano 1999). Seagrass photosynthetic efficiency is also considered as an indicator of broader coastal ecosystem health (Fonseca et al. 2003). Laboratory measurement of seagrass photosynthesis is extremely intrusive, as the plants need to be removed from their natural environment and a high degree of manipulation is required (Silva et al. 2009). One of the oldest and simplest ways to quantify photosynthetic activity of plant is to measure the O_2 evolved (Silva et al. 2009). In situ measurement of seagrass photosynthesis can be evaluated using submersible pulse-amplitude modulated (PAM) fluorometers (e.g. Beer and Björk 2000).

Silva et al. (2008) used an infrared gas analysis (IRGA) technique for measuring photosynthesis in seagrasses by a continuous measurement of dissolved CO_2 flux using incubation chambers connected to an analyser at the surface. Even though in-situ measurement of seagrass photosynthesis is not as exhaustive as laboratory analysis, and control of experimental conditions is limited, these at least provide results representative of natural conditions (Silva et al. 2009).

One of the possible remote sensing methods to measure photosynthesis in seagrass meadows is by studying the relationships between spectral reflectance (using a spectroradiometer or other sensors) and photosynthesis, which has been applied to terrestrial plants (e.g. Richardson and Berlyn 2002). However, underwater light attenuation properties may restrict the applicability of this method. This method can also be restricted by depth and the physical and chemical properties of water. Zimmerman (2003) proposed a two-flow bio-optical model for predicting downwelling spectral irradiance distributions, which is a robust tool for the measurement of photosynthesis in seagrasses as a function of water quality, depth, canopy structure, and leaf orientation. A remote sensing approach for the indirect measurement of photosynthesis in seagrasses can be based on the changes in pCO_2/acidification, salinity or chlorophyll concentration, which can be obtained from airborne/spaceborne sensors (discussed in section 4). Acoustic methods for photosynthesis measurement are preferred by some authors (e.g. Hermand 2004a, 2004b). Wilson et al. (2012) observed that high frequency acoustic methods perform better in estimating seagrass photosynthesis compared to low frequency. The photochemical response index (PRI), which is a measure of photosynthetic radiation use efficiency (PRUE) (Thorhaug et al. 2006), is another option to be explored further for photosynthesis measurement of seagrasses using remote sensing.

3.4. Seagrass associated species (epiphytes and epifauna)

Epiphytes are commonly associated with seagrasses and can be an indicator of excessive nutrients in the marine ecosystem (Teng et al. 2013) having a negative impact on seagrass environments. Epiphytes are traditionally assessed by sampling individual shoots or visual estimation within quadrats (Kendrick and Lavery 2001). Quantifying algal epiphytic biomass is important in assessing changes in biomass due to eutrophication (Kendrick and Lavery 2001). The preferred method for determining epiphyte biomass is dry weight and ash free dry weight (Kendrick and Lavery 2001). Studies on epiphyte productivity are used to determine the relative contribution of epiphytes to total meadow production and to understand the contribution of epiphyte production to higher trophic levels (Pollard and Kogure 1993).

In general, species or community level discrimination of seagrass meadows and associated vegetation, even though difficult, is theoretically possible using hyperspectral data (Mutanga and Skidmore 2004). For example, Dierssen and Russel (2015) used hyperspectral data (Portable Remote Imaging Spectrometer – PRISM) for the assessment of the hyperspectral properties of the macroalgae *Sargassum* and the seagrass *Syringodium filiforme* in Greater Florida Bay. Drake et al. (2003) developed a model that yielded a robust and positive relationship between epiphyte biomass and its absorption of photons and a strong negative relationship between epiphyte biomass and spectral photosynthesis of seagrass hosts.

Field sampling methods for the study of seagrass-associated fauna, such as those species living within the bottom sediments (infauna) and those living in the canopy or seabed (epifauna), include: hand-held corers, suction samplers, deep water sampling, and grabs and box corers for infauna and: small beam net to suction samplers and deep water samplers for epifauna (Guzman and Grizzle 2001). Large and mobile epibenthos, including fish, crabs and shrimps), require more sophisticated mechanical assistance such as adequately sized boats and underwater cameras (Edgar et al. 2001).

Remote sensing methods for studying seagrass-associated fauna are more complex than those for seagrass-associated flora. Teng et al. (2013) used hyperspectral data from a spectrophotometer to investigate epifauna cover on seagrass blades using the N-FINDR algorithm for spectral analysis. Indirect methods could include using remotely sensed data to evaluate changes in the foliar structure of seagrasses as a result of invertebrate grazing (Nakaoka 2005), which can be differentiated from foliar changes resulting from other factors such as natural decomposition based on spectral properties.

4. Secondary applications of remote sensing in seagrass research

In addition to the direct measurement of seagrass distribution, cover and biomass, remote sensing applications have been widely applied to measure water quality, light penetration, sediment type and other physical parameters (e.g. temperature, salinity, pH, water currents, waves and turbulence) in seagrass meadows (Dekker et al. 2007; Daud et al. 2019; Gumusay et al. 2019). Knowledge concerning environmental conditions within seagrass meadows is important to understand the spatial distribution and the relationship with environmental variables (Roelfsema et al. 2013; Daud et al. 2019; Lima et al. 2020). Remote sensing data from multiple sources can be used to analyse some of the aforementioned environmental variables. For example, Nezlin et al. (2005) used SeaWiFS imagery to analyse the extent of sediment plumes. Seagrass growth is highly dependent on sea surface temperature, suspended sediment and salinity (in order), with optimal seagrass growth occurring in ideal temperature and salinity conditions, dependent on species, with low suspended sediment concentrations (Daud et al.

2019). Radar remote sensing is another option for measuring environmental variables, such as ocean currents and tides, wave height, surface wind speed and direction, which are important for seagrass communities (Hossain and Hashim 2019).

4.1. Suspended sediments

Water quality and other environmental variables, including suspended and dissolved sediment concentrations, have been monitored in seagrass environments using remote sensing data by applying linear and non-linear algorithms (Ferwerda et al. 2007; Devlin et al. 2015; Han et al. 2016; Petus et al. 2016). Current satellite imagery have a high enough spatial and spectral resolution for monitoring sediment plumes, which is a key factor influencing light availability in coastal and estuarine regions based on surface reflectance differences due to varying turbidity conditions (Barnes et al. 2015). The analysis of coastal waters, however, is complex, due to the presence of organic and inorganic sediments, microorganisms, phytoplankton and water currents (Pozdnyakov et al. 2005). Landsat data has been widely used in shallow water remote sensing for monitoring suspended sediments (e.g. Harrington et al. 1992; Brivio et al. 2001; Thiemann and Kaufmann 2002; Barnes et al. 2015; Veettil and Quang 2018), while other spaceborne data with comparable spatial resolution to Landsat, such as ASTER, have also been used for the estimating suspended solids in coastal waters (e.g. Kishino et al. 2005). Low resolution satellite data such as AVHRR (Ruhl et al. 2001) or MODIS (Hu et al. 2004; Devlin et al. 2012; Schroeder et al. 2012; Petus et al. 2014b; Kumar et al. 2016) or VIIRS (Han et al. 2016) have been used successfully for analysing suspended sediment concentrations in estuarine and coastal water environments. Most of these studies were based on the establishment of empirical relationships between in situ observations and remote sensing data for estimating suspended solids. Airborne microwave data can also be employed for estimating sediment plumes in marine environments. For example, Burrage et al. (2003) used airborne data from the scanning low frequency microwave radiometer (SLFMR) for mapping plumes and salinity by applying regression methods in the Great Barrier Reef. Zhang et al. (2002) used a combination of Landsat and spaceborne microwave data (ERS2 SAR data) for surface water quality estimation in the Gulf of Finland. Aerial images, combined with satellite data, have also been used in some studies (e.g. Devlin and Brodie 2005; Devlin and Schaffelke 2009) for estimating the extent of sediment plumes. For small areas with detailed observations on turbidity and sediment plumes in seagrass environments, even though expensive, high resolution airborne data such as portable remote imaging spectrometer (PRISM) can be used effectively (Fichot et al. 2016). However, for understanding sediment dynamics at the seabed, seagrass sediment coring is recommended where specific horizons can be dated (e.g. ^{210}Pb, ^{14}C) (Ward et al. 2014).

4.2. Light penetration

One of the variables that determines the percentage of light penetrating the water column and available for seagrasses is the diffuse attenuation of solar light (K_d m^{-1}), which can be measured using remote sensing data such as MODIS imagery (Barnes et al. 2014a). Spaceborne ocean colour sensors, such as the Coastal Zone Color Scanner (CZCS) and SeaWiFS, can be used to estimate light penetration in the water column based on the blue-to-green reflectance ratio (Gattuso et al. 2006). However, in practice, such relationships may not be straight forward in coastal waters with high suspended particle concentrations (two main contributors of light attenuation in coastal waters are phytoplankton and suspended particles). Hill et al. (2014) used airborne hyperspectral data for the evaluation of light availability and biomass of seagrass environment in Saint Joseph's Bay in Florida.

4.3. Ocean temperature

For the measurement of sea surface temperatures (SST), which is an important variable for seagrass habitat, spaceborne datasets such as AVHRR (1 km) or MODIS (250 m) imagery can be used (Esaias et al. 1998), even though these data have a coarse spatial resolution. SST measurements using split window techniques have been found to be superior to that of using single spectral channel or a pair of windows in separate spectral regions (Stramma and Cornillon 1986) by reducing the impact of atmospheric transmittance and water vapour content in the algorithms (Sobrino et al. 1993). Non-linear SST algorithms using AVHRR provide similar accuracies under a wide range of environmental conditions (Li et al. 2001). Another widely used satellite data is from the Geostationary Operational Environmental Satellites (GOES 8/9) launched by NOAA (Wu et al. 1999). For SST measurements, ship-based sensors such as Atmospheric Emitted Radiance Interferometer (AERI) have been used with accuracy less than 0.1^{o}C (e.g. Smith et al. 1996). It is worth to note that SST retrieval from space in most of the ocean areas are sampled from polar orbiting satellites at most twice a day and that surface diurnal variability studies rely on the extrapolation of *in situ* measurements at depth (Gentemann and Minnett 2008). However, the ocean surface responds to changes in fluxes of heat and momentum rapidly and the diurnal variability at the ocean surface can be quite different from heating at depth (Gentemann and Minnett 2008). Scanning radiometers, such as the Advanced Along-Track Scanning Radiometer (AATSR) and Advanced Microwave Scanning Radiometer (AMSR-E), have been observed to have accuracies comparable to in situ data (O'Carroll et al. 2008), although AATSR only functions well in cloud-free environments, whereas microwave measurements can be used in all weather conditions (Hosoda 2010). Even though microwave remote sensing studies have higher accuracies compared with infrared measurements, sea surface wind correction still remains a problem (Hosoda 2010). For regional applications with diurnal cycle SST information, NASA has developed a Short-term Prediction and Research Transition (SPoRT) program based on 1km MODIS data (Haines et al. 2007).

4.4. Ocean salinity

Under a changing climate, variations in ocean salinity occur due to the melting of polar ice caps and sea ice (Wadhams and Munk 2004; Stammer 2008) that may affect seagrass meadow health and photosynthetic abilities (Sandoval-Gil et al. 2012) from regional to global scales. It has been reported that the spectral reflectance of seagrass meadows have been altered due to changes in salinity (Thorhaug et al. 2006), which can be explored using remotely sensed data. Optical and microwave remote sensing can be used to estimate salinity in marine environments. For example, Daud et al. (2019) estimated salinity conditions and suspended sediments in Banten Bay in Indonesia using Sentinel-2 data by applying simple band math algorithms and the normalized mean value error of the results were less than 10% for all the estimated variables (salinity and suspended sediments). Geiger (2011) applied ANN methods to MODIS data for ocean salinity measurements in the mid-Atlantic. Salinity from satellite data can be estimated indirectly based on the coefficient of coloured dissolved organic matter (a_{CDOM}) (Bai et al. 2013). Ocean colour monitoring satellite sensors, such as SeaWiFS, have been used successfully for ocean salinity mapping (Binding and Bowers 2003). Microwave data have also been widely used for salinity mapping in seagrass environments. The European Space Agency

(ESA) has designed and launched (November 2009) a satellite sensor – Soil Moisture and Ocean Salinity (SMOS) – specifically for mapping soil moisture and ocean salinity (Font et al. 2013). Some researchers (e.g. Hernandez et al. 2015) mentioned that the SMOS instrument has a better performance for salinity mapping ($r = 0.57$) than NASA's Aquarius ($r = 0.52$), which has been designed to understand the oceanic thermohaline circulation related to interannual climate variability (Koblinsky et al. 2003). Airborne microwave radiometers, such as Scanning Low-Frequency Microwave radiometer (SLFMR) and the Salinity, Temperature, and Roughness Remote Scanner (STARRS), perform better in mapping sea surface salinity, especially in estuarine and coastal environments (Klemas 2011).

4.5. Coastal ocean acidification

Ocean acidification has been linked to increases in photosynthetic rates in seagrasses due to the increase in concentration of aqueous CO_2 as a primary carbon source, and dissolved inorganic carbon species as bicarbonate ions (Garrard and Beaumont 2014; Repolho et al. 2017). Although some studies suggest an increase in organic carbon sequestration rates by seagrasses under acidic conditions (Palacios and Zimmerman 2007; Hall-Spencer et al. 2008; Fabricius et al. 2011; Russell et al. 2013; Garrard and Beaumont 2014; Mazarrasa et al. 2018) others report a substantial decrease (Martínez-Crego et al. 2014; Repolho et al. 2017).

Furthermore, recent studies (e.g. Unsworth et al. 2012; Hendricks et al. 2014; Koweek et al. 2018; Bergstrom et al. 2019) have suggested that the presence of seagrass communities can mitigate the negative effects of ocean acidification on marine ecosystems. Several studies have used remote sensing methods to monitor ocean acidification (Balch et al. 2007; Gledhill et al. 2008) and Sun et al. (2012) applied five variables (air-sea CO_2 fluxes, total alkalinity, suspended calcite (particulate inorganic carbon), particulate organic carbon and calcification rates) from remote sensing data to indirectly estimate ocean acidification (Takahashi et al. 2014). In addition, since ocean acidification is directly linked to global calcification rates, estimation of calcification rates from satellite data can also be used as a proxy for ocean acidification (Balch et al. 2007; Moses et al. 2009). Recent studies, such as Land et al. (2015) and Sabia et al. (2015) discussed the application of salinity information from satellite data for the assessment of ocean acidification.

4.6. Water currents and waves

Ocean currents influence pollination processes, sedimentation rates and sediment geochemistry of seagrass ecosystems (Koch and Verduin 2001). Seagrass directional characteristics depend on wave direction – seagrass leaves tend to flap back and forth in wave-dominated habitats, whereas the leaves tend to bend in the direction of tidal waves in tide-dominated habitats (Koch and Verduin 2001). Turbulence affects carbon and nutrient transfer and dispersion of pollen, seeds and spores in seagrass habitats (Koch and Verduin 2001). Air-sea turbulent fluxes are important in the exchange of momentum, heat and gas between atmosphere and ocean (Bourassa et al. 2010). Therefore, monitoring ocean currents and waves using remote sensing platforms can be useful in understanding seagrass distribution, reproduction and sedimentary cycles.

Ocean surface currents can be measured indirectly from spaceborne data using physical models involving the variables such as sea surface height, surface winds and sea surface temperature (Dohan and Maximenko 2010). Other methods, such as using surface velocity measured from buoys transmitted to satellite sensors or tracking of surface features and use of

Doppler shift in radar fields from SAR, have also been used (e.g. Dohan and Maximenko 2010). High frequency (HF) radar systems, such as Ocean Surface Current Radar (OSCR), provide periodic, two-dimensional vector estimates of surface currents (Chapman et al. 1997; Shrira et al. 2001; Klemas 2012). Furthermore, microwave data can also be used to estimate ocean currents indirectly from other variables such as SST (Isern-Fontanet et al. 2006). Klemas (2012) mentioned that ocean currents from infrared remote sensing data can be studied by tracking the movement of thermal and colour features in the ocean.

Microwave remote sensing provides information on ocean surface roughness, showing surface waves by analysing backscatter radiation (Goldstein et al. 1994; Pearce and Pattiaratchi 1997). In addition to spaceborne data, airborne sensors can also be used to study ocean surface waves. For example, Dugan et al. (2001) used an airborne optical sensor called Airborne Remote Optical Spotlight System (AROSS), which can also be mounted on a UAV, for measuring surface waves from time-series imagery.

4.7. Marine eutrophication using remote sensing

Nutrient pollution is a key driver of eutrophication and algal blooms within coastal waters (Lapointe et al. 2015). Eutrophication influences seagrass meadows in different ways, for example, Raffaelli (1999) observed that seagrasses in Scottish coastal areas were replaced by green algae following eutrophication. Increases in green algae reduce light penetration in coastal waters thereby reducing seagrasses photosynthetic capabilities and growth rates (Ferreira et al. 2011). In other words, seagrasses were observed to have been replaced by fast growing competitors like microalgae and macroalgae during the initial stages of eutrophication (Waycott et al. 2009).

Eutrophic conditions can be studied using remotely sensed data by examining the growth of phytoplankton (by estimating the chlorophyll content) (Bagheri and Dios 1990). Lee et al. (2004) found that the ratio of leaf nitrogen to leaf mass can be considered as a sensitive and consistent indicator of early eutrophication. Measurement of nitrates, phosphates and other nutrients can also be estimated using spaceborne and airborne data (Veettil and Quang 2018). Cauwer et al. (2004) used spring mean and maximum chlorophyll-a concentrations from satellite data (MODIS, SeaWiFS, MERIS) for estimating the eutrophication status with coastal waters in Belgium. Large-scale marine trophic conditions, such as in the Mediterranean Sea, have been studied using SeaWiFS (Acker et al. 2005; D'Ortenzio and d'Alcala 2009) or MODIS data (Allen et al. 2008) or a combination of both (Banks et al. 2012). Airborne multispectral or hyperspectral data, such as AVIRIS, can be effectively used for retrieving marine water constituents and estimating eutrophication (Bagheri et al. 2005). For small-scale eutrophication measurement based on chlorophyll-a data from space, Landsat TM data has been used since 1980s (e.g. Bagheri and Dios 1990). Coastal Zone Color Scanner Experiment (CZCS), which is the first spaceborne instrument specifically made for the measurement of ocean colour, has been used for eutrophication monitoring in coastal areas based on mapping the extent of algal blooms (Blondeau-Patissier et al. 2014).

The key challenges in using remote sensing data for eutrophication studies is the improvement of algorithms applied and refining the detection limits in different oceanic and coastal environments (Blondeau-Patissier et al. 2014). It is essential to consider the effects of water temperature, turbidity, solar radiation and bathymetry to understand spatio-temporal patterns of algal blooms and eutrophication (Blondeau-Patissier et al. 2014). In order to estimate coastal eutrophication from remote sensing datasets, proper cloud masking schemes need to be applied to improve the accuracy (Banks and Melin 2015). For example, Barnes et al. (2014b) aggregated the Landsat TM 30 m pixels to 240 m pixel to increase the signal-to-noise ratio and applied a MODIS-like atmospheric correction approach for estimating water quality, which was found to have improved the accuracy.

5. Discussion

Seagrass ecosystems are important coastal environments currently under threat throughout their global range (Orth et al. 2006a; Waycott et al. 2009; Jones and Unsworth 2016; Unsworth et al. 2017; Jones et al. 2018). As they cover relatively large areas, sometimes poorly accessible for field research, remote sensing is an alternative tool for mapping and monitoring these ecosystems (Pham et al. 2019). This review has highlighted the significant contributions of remote sensing datasets and various techniques applied to seagrass research, while comparing their use and accessibility to traditional, in-situ, methods. Ecologists and managers can use the information acquired by high spatial resolution maps to provide invaluable information about seagrass ecology and environmental health to be used for the management of protected areas and understanding of biodiversity, functioning, services, and future sustainability of seagrass, particularly in areas with reported seagrass decline (Koedsin et al. 2016; Giardino et al. 2017; Pham et al. 2019).

Overall, it has been demonstrated that high spatial resolution data can be used to improve seagrass classification accuracy (Sagawa et al. 2010; Meyer et al. 2010; Lu and Cho 2012; Tamondong et al. 2013; Saunders et al. 2015; Barrell et al. 2015; Pham et al. 2019). However, medium to high spatial resolution data, such as the Landsat time-series, have been most widely used for monitoring these ecosystems at larger scales (Meyer et al. 2010; Knudby and Nordlund

2011; Ferreira et al. 2012; Borfecchia et al. 2013; Hogrefe et al. 2014; Kim et al. 2015). Moreover, active remotely-sensed data, such as SAR and LiDAR, have been used to ensure higher performance in mapping and monitoring of seagrass ecosystems than some low resolution space borne data (Hogrefe et al. 2014; Duffy et al. 2018), while multi-temporal high spatial resolution images have been used to monitor changes in specific areas (Tuxen et al. 2011; Pu and Bell 2017). The incorporation of multi-resolution and multi-source (SAR, multispectral, and LiDAR) data may be used as a tool to improve accuracy (Pham et al. 2019). For example, research efforts have been made to expand the use of optical sensors, such as multispectral and hyperspectral datasets, in combination with different traditional methods for mapping and monitoring seagrass ecosystems (Qiu et al. 2019; Giardino et al. 2019). However, more attention seems to have been paid to the more advanced, or the hybrid, remote sensing methods using a combination of multi-source and multi-temporal datasets (Pham et al. 2019).

Although multi-spectral imagery has emerged as a popular dataset for seagrass mapping, the limited number of spectral bands may lead to a low accuracy of single species detection (Lyons et al. 2011; Paulose et al. 2013). For this reason, hyper-spectral imagery has been widely combined with physical-based models and various classification algorithms to improve the accuracy of seagrass detection in complex water environments (Koedsin et al. 2016). Generally, the semi-analytical method using hyper-spectral imagery allows a higher mapping accuracy than the empirical approach (Roelfsema et al. 2014). However, it requires an intensive spectral library of different bottom curves as the input for the classification algorithm, which implies an expensive field sampling and storage of the library in the case of large-area and mixed bottom type site monitoring (Roelfsema et al. 2014; Traganos and Reinartz 2018c; Duffy et al. 2018;

Gereon et al. 2018). In addition, hyper-spectral sensors usually have a small coverage and require on-demand flights for specific geographic regions (Pham et al. 2019).

It should be noted, however, that the remote sensing processes vary depending on the environmental conditions of the study area (i.e., depth, tidal level, etc.), highlighting the need to conduct prior field research to appropriately interpret and validate remote sensing data obtained (Short and Coles 2001; Howari et al. 2009; Phinn et al. 2008; Lyons et al. 2011; Koedsin et al. 2016). In the near future, more advanced sensors, such as SAR and LiDAR, and novel machine learning approaches and deep learning methods should be used for mapping and monitoring seagrass ecosystems. Therefore, focus should be placed on the development and selection of state-of-the-art machine learning algorithms for mapping and monitoring in future studies.

6. Conclusions and future research

Despite the fact that seagrass meadows function as primary producers, mitigate climate change through carbon sequestration, and provide a habitat for a wide variety of marine plant and animal species, there has been a decline in global seagrass cover and hence it is important to understand meadow location, distribution and health in a cost-effective way with a greater geographical cover.

Recently, a large number of active and passive remote sensing approaches have been used to estimate various parameters associated with seagrass communities, such as plant cover and habitat quality. Such methods can provide relevant and long-term data for analysing seagrass ecosystem change. Seagrass ecosystem information that can be evaluated using remote sensing are: presence of seagrass, extent, species composition, seagrass-associated communities, stem health, biomass, temporal ecosystem changes, ecosystem services, and environmental quality.

Various types of remote sensing data that can be used to analyse seagrass ecosystem data include terrestrial and underwater photography/videography, aerial photographs and airborne hyperspectral data, spaceborne multispectral and hyperspectral data, LiDAR, Laser scanners, Sonar, Radar, and data from UAVs/AUVs.

There are also a number of environmental parameters that influence seagrass ecosystem location and ecological health that can be evaluated using remotely sensed data, in particular: suspended sediments, light penetration, ocean temperature, ocean salinity, coastal ocean acidification, and marine eutrophication.

Even though remote sensing can be used for a wide variety of applications in seagrass research, there are limitations to be overcome regarding spatial, spectral and radiometric resolution and environmental factors such as depth, turbidity, phytoplankton and pollution. Unlike terrestrial plant ecosystems, seagrass communities are often submerged and hence there are limitations to apply landscape techniques using remote sensing methods to seagrasses. While using optical data, only visible wavelengths pass through water column and can be used for mapping seagrass communities and both atmospheric correction and water column attenuation coefficients must be applied. Furthermore, spectral discrimination of seagrass communities using optical data becomes difficult with increase in depth due to absorption and scattering.

A combination of multi-type, multi-source of remotely sensed datasets combining remote sensing and field data are most effective in understanding the broader aspects of seagrass ecosystems. For example, a combination of multispectral optical data and SAR data can be helpful in discriminating seagrass meadows and green algae based on surface roughness information and sediment types. Recent advancements in data and decreases in costs of UAVs and AUVs provide an excellent opportunity to obtain fine scale data concerning seagrass meadow structure, function, location, and species diversity, although this is best combined with field data. In order to improve data quality from UAVs, polarized filters can be used for sun glint correction. High spatial resolution hyperspectral data from spaceborne sensors with fixed sensor altitude and orbit may reduce the coast of image acquisition and processing compared to airborne data having a variety of deployment altitudes and measurement geometry. Various classification algorithms and physical-based models are being developed resulting in improved accuracy for seagrass mapping. There has historically been a paucity of studies on seagrasses compared with related ecosystems such as mangroves, salt marshes or coral reefs that offer a similar range of ecosystem services. However, in light of the growing interest in seagrasses, particularly concerning ecosystem service provision it should be noted that there are a range of remote sensing techniques and platforms that can be used to study these vital ecosystems. Variations in estimations of extent and distribution of seagrasses are much greater than those for related coastal systems. The authors, therefore would like to encourage a range of studies from extent and distribution mapping, to carbon sequestration and storage to ecosystem health utilising a range of methods, including those highlighted in this review, in order to fill the gap in knowledge of seagrass systems at local, regional and global scales.

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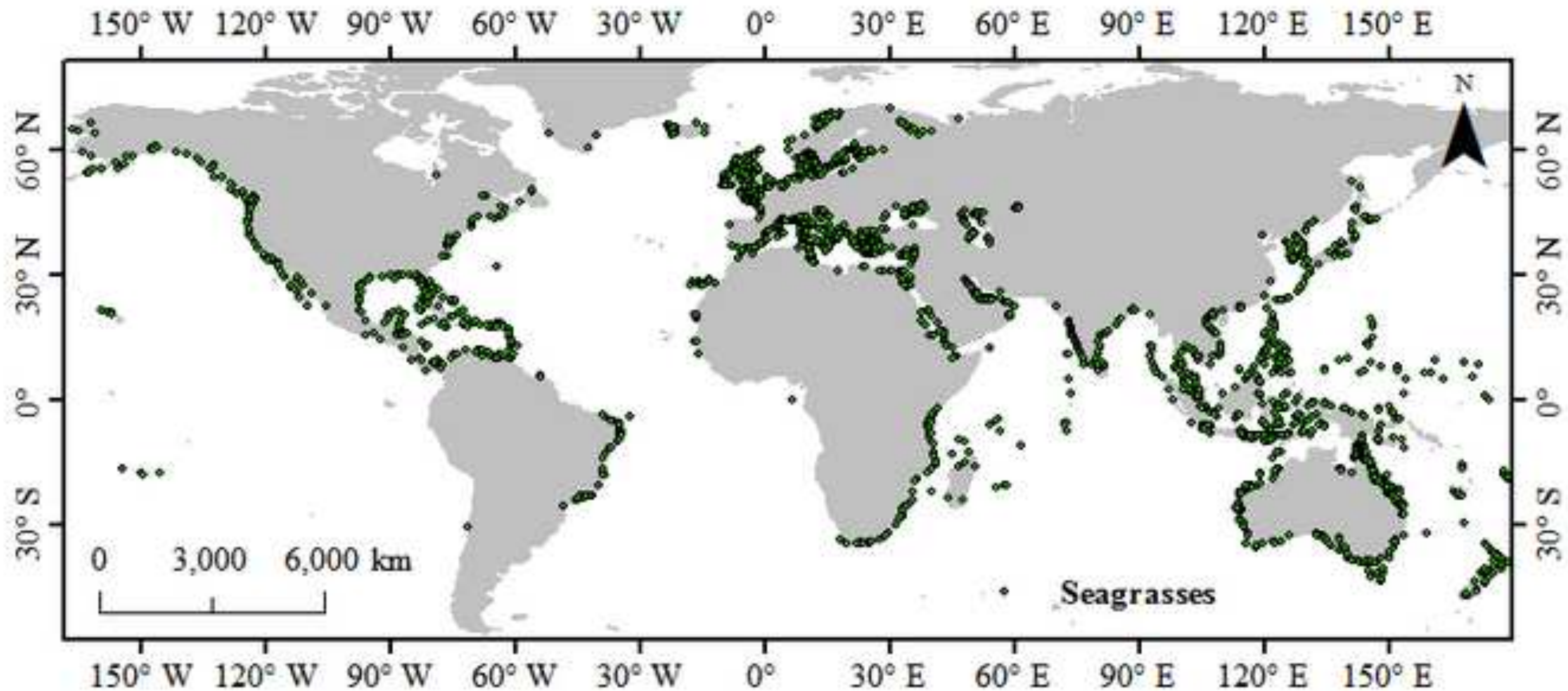

Figure 2A

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