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Improving data on the distribution and ecological value of temperate subtidal seagrass in Tayaritja Sea Country (Furneaux Group of Islands), Lutruwita (Tasmania)

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Acknowledgement of Country

We acknowledge Palawa (Tasmanian Aboriginal people) as the First People of Lutruwita (Tasmania) who have not ceded their sovereignty to their Country.

Palawa have an unbroken obligation to care for Country.

Country is the English word Palawa use to describe their lands, waterways, seas and sky. However, Country is more than a description of Palawa connection to lands, waterways, seas and sky. Country holds lore, custom and language, and is at the centre of Palawa spiritual and cultural practices and provides sustenance for Palawa as it has for millennia.

We pay our respects to elders past and present and acknowledge the enduring role that Palawa must play in caring for their Country.

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Executive summary

This report presents the outcomes of a Marine and Coastal Hub-funded project focused on improving the mapping, monitoring, and management of seagrass habitats in Lutruwita (Tasmania), with an emphasis on culturally significant Tayaritja Sea Country (seas around the Furneaux Islands) for Palawa (Tasmanian Aboriginal people). The project addressed the critical need for robust baseline data to support seagrass conservation, blue carbon assessment, and enhance the Pakana Rangers efforts to monitor the health of culturally significant seagrass beds in Tayaritja Sea Country.

A total of 1,539.14 km^2 of Tayaritja Sea Country was mapped in 2023 and 2024 using a combination of Sentinel-2 satellite imagery, bathymetric LiDAR and derivatives using a machine learning approach, with training and validation data provided by an underwater BOSS camera system. These tools resulted in the mapping of 755.80 km^2 of seagrass habitat. Core samples were collected at 10 sites to assess sedimentary carbon stocks, contributing to blue carbon knowledge essential for future climate mitigation strategies.

The project also conducted a biodiversity assessment and differentiated habitats such as macroalgae and sand habitat extents and developed methods for monitoring in collaboration with the Tasmanian Aboriginal Centre's Pakana Rangers. This research was conducted in accordance with a Pakana-led collaborative research agreement that was established between the parties to support the application of Indigenous Data Sovereignty principles and respect the rights of Pakana to their Indigenous Cultural and Intellectual Property. Culturally respectful engagement was central to this work, providing Pakana Ranger training in marine monitoring techniques and co-developing protocols aligned with community priorities.

Key outputs include high-resolution habitat maps, seagrass species inventories, blue carbon data, and guidance for long-term monitoring. These deliverables enhance the collective capacity of all rights and interest holders to manage seagrass ecosystems under pressure from human activities such as port development, pollution and climate change. Importantly, the project lays the groundwork for management that recognises and provides for collaborative research and monitoring with and by the Pakana Rangers in the proposed Tayaritja Wukalina milaythina muka Indigenous Protected Area (IPA).

Marine managers can use the findings to inform healthy Country management, spatial planning, ecosystem reporting, restoration prioritisation, and inclusion of blue carbon in climate policy. The co-design model and integration of cultural values serve as a template for future marine science initiatives aiming to deliver both ecological and social outcomes, consistent with best practice principles for the protection of cultural heritage values, Indigenous Data Sovereignty principles and Indigenous Cultural and Intellectual Property regimes.

1. Introduction

Seagrass beds are a dominant marine ecosystem in Tayaritja Sea Country (seas around the Bass Strait Islands in the Furneaux Group of Islands) to the northeast of Lutruwita (Tasmania), with historical coarse mapping indicating extensive beds of *Posidonia, Amphibolis, Heterozostera* and *Zostera* seagrass (Jordan et al., 1998). The beds of *Posidonia* and *Amphibolis* were identified as some of the largest in extent and deepest found in temperate waters of Australia. Lack of data on the distribution and ecological value of these seagrass habitats represents a significant knowledge gap in understanding Australian wetland natural assets that provide a range of ecological, social, cultural and economic values.

This project mapped the extent and ecological composition, population structure and blue carbon value of the seagrass beds in Tayaritja Sea Country in partnership with the Tasmanian Aboriginal Centre's Pakana Rangers. This has helped to improve understanding of the ecological and cultural significance of these seagrass beds and has supported Pakana Ranger capability development, to enable them to undertake ongoing monitoring to support the protection of seagrass beds in the proposed Tayaritja Wukalina milaythina muka IPA.

1.1 Background and scope

Seagrass habitats are among the most important benthic ecosystems in temperate coastal waters of Australia. These marine flowering plants form extensive underwater beds that provide a wide range of ecological functions and services. Seagrass beds support high biological diversity by offering nursery grounds, feeding areas, and shelter for numerous marine species. Their ability to uptake nutrients and reduce pathogens contributes significantly to nearshore water quality, while their dense root systems stabilise sediments and help prevent coastal erosion.

Importantly, the study area for this research is within Tayaritja Sea Country which is a culturally significant area for Palawa. This is reflected in the work that the Tasmanian Aboriginal Centre is doing to establish the first Sea Country Indigenous Protected Area (IPA) in Tasmania. This project supports their efforts by contributing ecological knowledge that can inform Aboriginal-led conservation and by building the capacity of Pakana Rangers through training in seagrass monitoring. As part of this initiative, Pakana Rangers supported the delivery of the fieldwork, received training in methods for seagrass monitoring which will help them monitor the health of these culturally significant seagrass beds into the future.

In addition to improving understanding about the extent and significance of Tayaritja seagrass beds, these habitats are also recognised as critical components of internationally significant bird areas and Ramsar-listed wetlands. They also play a key role in carbon cycling as blue carbon ecosystems, with the capacity to store substantial amounts of organic carbon in their underlying sediments over millennial timescales. However, spatial variability in carbon storage across different seagrass beds is still poorly understood, presenting a challenge for accurate blue carbon assessments and climate mitigation planning.

Despite their cultural and ecological importance, there are major gaps in the availability of spatially continuous seagrass mapping data across much of temperate Australia. This is particularly true in Tayaritja Sea Country, where the distribution and extent of seagrass has

been poorly documented until now. The Marine and Coastal Hub's national wetlands scoping study (Project 1.5) identified this lack of data as a significant knowledge gap. Earlier, coarse-scale mapping from several decades ago identified the presence of *Posidonia*, *Amphibolis*, *Heterozostera*, and *Zostera* seagrass beds in coastal waters of northern Tasmania (including Lutruwita), extending to depths of approximately 25 m and covering an estimated area of at least 500 km² (Jordan et al., 1998). For context, this area is substantially larger than the total mapped seagrass area along the New South Wales coastline, which is approximately 180 km² (West & Glasby, 2021) – highlighting the ecological importance of the habitat in the region.

To address this knowledge gap our project focused on mapping and assessing the seagrass beds in Tayaritja Sea Country in collaboration with the Pakana Rangers. We used an integrated suite of close-range remote sensing methods, including high-resolution satellite imagery from Sentinel-2 (10 m) and Planet Dove (3–5 m), combined with bathymetric LiDAR data and oceanographic variables. These remote sensing datasets were used to create a baseline map of seagrass extent and spatial composition. Fieldwork was conducted to validate the remote sensing data, including underwater video surveys, photographic quadrats, and rhizome coring. Together, these efforts produced detailed maps of seagrass extent, species composition, and spatial variability – within the limits imposed by water depth and clarity.

Importantly, the study area around Tayaritja holds deep cultural significance for the Tasmanian Aboriginal Community. The Tasmanian Aboriginal Centre (TAC) is working toward establishing a Sea Country Indigenous Protected Area in the region. This project supports these efforts by contributing ecological knowledge that can inform Aboriginal-led conservation and by building local capacity through training. As part of this initiative, Pakana Rangers participated in the fieldwork and received training in methods for seagrass monitoring. This collaboration enhances the community's ability to care for Sea Country into the future.

Globally, seagrass ecosystems are facing increasing threats and have experienced significant declines in distribution and health (Waycott et al. 2009). These declines have raised serious concerns given the critical roles seagrasses play in maintaining coastal ecosystem function. While a range of natural and human-induced factors – such as pollution, climate change, and coastal development – have been implicated in these losses, recent research suggests that the resilience of seagrass ecosystems depends heavily on the levels of genetic diversity and connectivity within and among seagrass populations (Hughes & Stachowicz, 2004; Reusch et al., 2005; Ehlers et al., 2008). Therefore, understanding the patterns of genetic variation and population connectivity is essential for predicting the long-term viability of these systems and for informing conservation strategies.

Like seagrass beds, Palawa people face serious consequences from the mismanagement of their Country. They too suffer poor health when the condition of their Country is deteriorating. This is why collaborative research like that conducted in this study is both socially and ecologically important.

Seagrass ecosystems also provide significant socio-economic benefits. They literally sustain people. They support commercial, recreational and cultural fisheries, contribute to food security, and provide opportunities for tourism and recreation (Mtwana Nordlund et al., 2016; Do Amaral Camara Lima et al., 2023). Despite these benefits, and their global distribution

(being found around every continent except Antarctica), many regions still lack the comprehensive mapping and ecological, cultural and social assessments needed for effective management.

In the following section, we explore the ecological roles of temperate seagrass habitats and the challenges they face, including environmental stressors and knowledge gaps. We also review past seagrass mapping efforts in Australia, with particular attention to areas such as Tayaritja Sea Country, where deep-water and temperate seagrass beds remain poorly mapped. Lastly, we consider the cultural importance of these ecosystems for Palawa Tasmanian Aboriginal people and highlight the growing need for collaborative research and monitoring to support enhanced Aboriginal-led conservation efforts aimed at ensuring their long-term protection.

1.2 Ecology and significance of temperate seagrass

1.2.1 Temperate habitats and species

The Temperate Southern Ocean zone is one of six global seagrass bioregions, as defined by distinct assemblages of seagrass species, their geographic distribution, and the interplay of tropical and temperate oceanographic influences (Short et al., 2007). This bioregion encompasses temperate Australia, New Zealand, South Africa, and parts of South America, and is home to 18 recognised seagrass species.

Within this bioregion, seagrass habitats are typically classified into three broad types based on their location and environmental conditions: estuarine, coastal, and deep-water habitats (generally >15 m depth). Other habitat types – such as coastal lakes, tidal rivers, and sheltered beaches – have also been described and contribute to the overall diversity of seagrass environments (Macreadie et al. 2018). Across all habitat types, species distribution is largely influenced by substrate type, water clarity (light availability), temperature, salinity, and hydrodynamic conditions (Hemminga et al., 2000; West & Glasby, 2021).

Estuarine and coastal seagrasses in temperate waters mostly occur from the intertidal zone down to depths of about 15 m. Estuarine species must tolerate highly variable environmental conditions, including fluctuations in temperature, salinity, and light availability. In temperate Australian estuaries, common species include *Zostera muelleri* and *Heterozostera nigricaulis.* Coastal species, in contrast, tend to inhabit more stable environments, such as sheltered bays or exposed coastlines where they must withstand higher wave energy. These habitats support species such as *Posidonia australis*, widespread across southern Australia, and *Amphibolis antarctica,* which can colonise both sandy and occasionally rocky substrates.

Deep-water seagrass habitats – those found at depths greater than 15 m – are less well understood, primarily due to the logistical challenges of studying these environments. In these deeper areas, light becomes the most significant limiting factor for seagrass growth and distribution. Only a small number of species have been documented in such habitats. For instance*, Thalassodendron pachyrhizum* has been recorded at depths of up to 63 m in western and southern Australian waters (Martin et al. 2023). Species of *Halophila* have been observed as deep as 76 m on the Great Barrier Reef (McCormack et al. 2013; York et al. 2015, 2018; Carter et al., 2021). *Zostera* species have been found to 35 m off the coast of Victoria (Ierodiaconou et al., 2007), and *P. australis*, although typically restricted to depths

less than 15 m, has been recorded as deep as 20 m in parts of Bass Strait (Jordan et al., 1998).

Despite these observations, our understanding of species distribution and ecological dynamics in deep-water seagrass systems remains limited. The inaccessibility of these habitats and challenges associated with remote sensing and field surveys mean that significant knowledge gaps persist – particularly regarding the full extent of species ranges, habitat composition, and the role these deep systems play in supporting biodiversity and ecosystem services (Martin et al., 2023).

1.2.2 Ecosystem services

Seagrasses are foundational marine ecosystems that provide a wide range of valuable ecosystem services. These include blue carbon sequestration, maintenance of water quality, sediment stabilisation, and the provision of habitat for diverse marine species. In addition to their ecological roles, seagrass beds offer significant socio-economic benefits. They support fisheries by providing nursery and feeding grounds, protect coastlines from erosion, and underpin recreational, tourism, and cultural activities (Mtwana Nordlund et al., 2016; Do Amaral Camara Lima et al., 2023).

While these services are generally shared across seagrass ecosystems, the extent and intensity of service provision can vary substantially depending on the size of seagrass beds, species traits and environmental conditions. Some species form expansive, dense beds covering up to 10,000 hectares, which greatly enhances their ability to deliver ecosystem services at scale (Hemminga & Duarte 2000; Mtwana Nordlund et al., 2016). The life history strategies of different seagrass species also influence their ecological function. Long-lived, slow-growing species such as *Enhalus* and *Posidonia* tend to provide greater contributions to carbon storage and sediment stabilisation compared to fast-growing, opportunistic species like *Z. muelleri* (Kilminster et al., 2015; Mtwana Nordlund et al., 2016).

Recent research has begun to reveal the ecological significance of deep-water seagrass habitats, which remain relatively understudied compared to their shallow-water counterparts. These deeper systems can support high levels of biodiversity and store considerable amounts of carbon. For instance, over 101 fish species from 50 families have been recorded in deep *Thalassodendron pachyrhizum* beds in southwestern Australia, highlighting their habitat value (Martin et al. 2023). Similarly, deep-water *Halophila* species have demonstrated organic carbon content levels comparable to those in shallow-water habitats, despite the typically lower density of these deeper stands (York, Macreadie & Rasheed, 2018).

In the following section, we explore three key ecosystem services provided by seagrasses in greater detail: their function as blue carbon ecosystems, their role in supporting marine biodiversity, and their contribution to maintaining nearshore water quality.

1.2.3 Seagrass as blue carbon ecosystems

With increasing atmospheric carbon dioxide levels, there is growing global interest in understanding how natural ecosystems can capture and store carbon to help reduce net greenhouse gas emissions. Since 2009, ocean ecosystems have gained recognition for their role in carbon sequestration, particularly mangroves, saltmarshes, and seagrasses – collectively known as “blue carbon ecosystems” (Nellemann et al., 2009).

These ecosystems are highly effective at storing organic carbon because they accumulate organic material and sediment over time. The typically anaerobic conditions in their soils inhibit microbial decomposition, allowing carbon to be stored for millennia provided the sediments are not disturbed. While mangroves and saltmarshes often have higher per-hectare carbon stocks in their plant biomass and soils (McLeod et al., 2011), the global extent of seagrass beds makes them critically important at both national and international scales for long-term carbon storage (Fourqurean et al., 2012; Serrano et al., 2019).

However, the carbon storage capacity of seagrass beds varies significantly across different spatial scales. Soil carbon stocks – referring to carbon stored in both biomass and sediments to a depth of one metre – and annual sequestration rates are influenced by factors such as climate, geomorphology, and species-specific traits. Large-scale drivers like temperature and precipitation patterns can shape variability across countries and continents (Fourqurean et al. 2012; Young et al. 2021), while regional differences may be driven by coastal landforms and sediment dynamics (Ewers Lewis et al., 2020). Even within individual estuaries, carbon storage can differ due to fine-scale variation in hydrodynamics and vegetation cover (Sanders et al., 2019; Ricart et al., 2020). Species-specific traits, such as morphology and habitat preference, also influence carbon storage capacity (Lavery et al., 2013). Understanding these multi-scale patterns is essential for accurate regional assessments of blue carbon potential (Young et al., 2018).

Once we have reliable estimates of current carbon stocks and sequestration rates, there are two key pathways to enhance seagrass blue carbon. The first is through protection: maintaining intact seagrass beds and preventing disturbance of existing carbon-rich sediments. These long-term carbon stores are vulnerable to physical damage from port development, dredging, anchoring, and boat moorings (Serrano et al. 2016), as well as broader ecological pressures such as altered predator-prey dynamics (Atwood et al. 2015; Carnell et al. 2020), climate change, and nutrient enrichment (Arias-Ortiz et al. 2018).

The second approach involves restoration – re-establishing seagrass beds to promote new carbon accumulation in sediments over time (Marbà et al., 2015; Macreadie et al. 2018). This restoration-based method underpins most carbon credit markets, including Australia's Carbon Credit Unit Scheme, which requires evidence of additional carbon sequestration beyond what would have occurred if seagrass was protected (Kelleway et al., 2020; Lovelock et al., 2023).

Together, protection and restoration of seagrass habitats are essential strategies for maintaining and enhancing the blue carbon potential of coastal ecosystems in the face of escalating climate and anthropogenic pressures**.**

1.2.4 The biodiversity contributions of seagrass habitat

Among the many ecosystem services provided by seagrass beds, one of the most widely recognised is their contribution to supporting high levels of marine biodiversity. The structure of seagrass canopies creates a complex three-dimensional habitat that supports a wide array of marine life, offering both food and shelter. This complexity generates diverse microhabitats, which are particularly important for fish, crustaceans, molluscs, and other marine organisms (Do Amaral Camara Lima et al., 2023). As a result, seagrass beds function as biodiversity hotspots across many coastal ecosystems.

A key ecological function of seagrass beds is their role as nursery grounds, especially for juvenile fish. The shelter provided by the canopy reduces predation risk, while the availability of food supports growth and survival. Several studies, including a meta-analysis by Heck et al. (2003), have shown that juvenile fish exhibit higher abundance, growth rates, and survival in seagrass habitats compared to unvegetated substrates. Interestingly, while seagrass beds generally outperform bare areas, comparisons with other structurally complex habitats – such as coral reefs and mangroves – suggest that it is the degree of structural complexity, rather than vegetation type per se, that most strongly influences nursery habitat value.

This nursery role has been observed globally. Numerous studies have reported consistently higher fish abundance in seagrass beds compared to adjacent unvegetated habitats (e.g., Guidetti, 2000; Olesen et al., 2024). In southern Australia, similar patterns have emerged, particularly in shallow systems dominated by *Zostera*, *Heterozostera*, and *Posidonia* species, which support greater densities of fish than adjacent bare sediments (Jenkins et al. 1998; Bloomfield et al., 2005; Smith et al., 2008). However, not all findings are uniform. Some studies (e.g., Hyndes et al., 2003; Gray et al., 2011) have reported no significant differences, highlighting the influence of sampling method, bed size, shoot density, and broader landscape context on observed biodiversity patterns (Hyndes et al., 2018).

Emerging research has also drawn attention to the biodiversity value of deep-water seagrass beds, which remain relatively understudied. For instance, over 100 fish species from 50 families have been documented in deep *Thalassodendron pachyrhizum* beds in southwestern Australia, underscoring the ecological significance of these deeper habitats (Martin et al., 2023).

Beyond vertebrate fauna, seagrasses support rich epiphytic communities, particularly on the leaves of slow-growing, long-lived genera such as *Posidonia* and *Amphibolis*. These epiphytes – comprising micro- and macroalgae, fungi, and sponges – form additional trophic layers within seagrass ecosystems, serving as important food sources for a variety of invertebrates and fish (Borowitzka et al. 2006). While direct grazing on seagrass leaves by temperate marine fauna is generally limited to a few omnivorous fish (Hyndes et al., 2018), the broader food web supported by seagrass-associated epiphytes is extensive.

Seagrass beds also host high densities of motile epifauna and epibenthic organisms, including grazing and predatory molluscs, as well as omnivorous and carnivorous crustaceans (York et al., 2018). Benthic invertebrate communities – particularly macroinfauna such as polychaetes, bivalves, and crustaceans – are significantly more abundant and diverse in seagrass beds than in nearby unvegetated areas (Edgar et al., 1994; Kindeberg et al., 2022). In some cases, invertebrate densities have been observed to decline sharply within just 10 cm beyond the edge of a seagrass bed (Barnes 2017), highlighting the sharp ecological boundary and localised biodiversity enhancement provided by these habitats.

In sum, the role of seagrass beds in supporting biodiversity spans multiple functional groups and trophic levels, reinforcing their importance as foundational habitats in coastal marine ecosystems. Ongoing research, especially in less explored deep-water environments, continues to expand our understanding of the full extent of their ecological contributions.

1.2.5 Contribution of seagrass to nearshore water quality

Seagrass beds play a vital role in maintaining and maintaining nearshore water quality through multiple biophysical mechanisms. Their physical structure slows water movement and reduces wave energy, which in turn limits sediment resuspension and promotes the deposition of fine particles (Hendriks et al., 2008; Christianen et al., 2013). Seagrass leaves trap suspended sediments and organic matter (OM), while belowground roots and rhizomes stabilise the seabed, further enhancing water clarity and sediment retention (Duarte et al., 2005). Over time, these processes lead to the burial and accumulation of OM and fine silts, creating sediment layers with high organic content (Bos et al., 2007). This sediment-trapping function is particularly effective in temperate seagrass species such as *Zostera marina* and *Thalassia testudinum*, which form persistent, high-biomass beds compared to more ephemeral tropical species (Mellors et al., 2002).

The accumulation of organic matter in seagrass sediments supports active microbial communities that drive nutrient cycling, increasing the availability of nutrients for plant uptake and making seagrass beds important nitrogen sinks in coastal systems. For example, *Z. marina* beds have demonstrated substantial nitrogen retention capacity. A study in Sweden estimated that the release of nitrogen from sediments following seagrass loss could cost society over US$141,000 per hectare, equating to a nitrogen release of 6.63 megagrams (Mg) per hectare (Moksnes et al., 2021).

Seagrass blades and their associated epiphytic communities also contribute to water column nutrient uptake. Epiphytes – such as those found on *Posidonia oceanica* – can absorb nitrogen and ammonia more rapidly than the seagrass itself; however, due to the larger overall biomass of the seagrass, the host plant generally acts as the more significant nutrient sink (Lepoint et al., 2007). While moderate nutrient enrichment can stimulate seagrass growth, chronic or excessive nutrient loading – often from anthropogenic sources – can be harmful. Elevated nutrient levels alter the internal balance of carbon, nitrogen, and phosphorus in seagrass tissues, reducing carbon sequestration potential and compromising plant health and ecosystem function (Touchette et al., 2000; Invers et al., 2004; Ashikin et al., 2020).

In addition to sediment and nutrient dynamics, seagrasses have also been shown to improve microbial water quality. They can significantly reduce bacterial pathogen loads in nearshore waters. A global field study reported up to a 50% reduction in potential bacterial pathogens in seagrass-covered areas compared to adjacent unvegetated sites, highlighting their role in mitigating coastal waterborne disease risks (Lamb et al., 2017; Deng et al., 2021).

Together, these functions illustrate the central role of seagrass beds in regulating nearshore water quality, not only through physical processes like sediment trapping and stabilisation, but also by supporting nutrient cycling, buffering against eutrophication, and reducing microbial contamination.

1.2.6 Cultural significance of seagrass beds

Seagrasses have been entwined with human culture for millennia, constituting a natural resource that has supported humanity throughout this history. Understanding their values fosters appreciation of these ecosystems, encouraging conservation and restoration actions to counteract historic and predicted losses (Foster et al., 2025). For First Nations peoples, recognising the cultural significance of seagrass beds, and empowering them to engage in

the management of seagrass beds also acts to overcome dispossession as it recognises their rights and interests, and cultural responsibilities to care for Country.

Country is the English word Palawa use to describe the lands, waterways, seas and sky that they are a part of. For Palawa, Country is more than a description of their connection to lands, waterways, seas and sky. For Palawa, Country is them and they are Country. Country is intrinsic to their culture and knowledge. They understand that they are a part of Country, not separate to it, and that they have an unbroken obligation to care for it, as they do for family.

For them, Country holds lore, custom and language, and is at the centre of their spiritual and cultural practices and provides sustenance for their people as it has for millennia. Under their law, Country is a living being that has its' own right to exist and thrive; and they have an ongoing, unbroken and reciprocal relationship with Country. Central to their relationship to Country is that they are interconnected with all elements of Country within a reciprocal relationship. For them, "Healthy Country Healthy People" is real, and it goes both ways too - healthy people also leads to healthy Country.

Within the proposed IPA seagrass beds are extensive, being a dominant marine ecosystem. This makes seagrass beds in Tayaritja Sea Country an important element of Country, that have sustained Palawa, through providing kipli/food, shelter, and essential nursery areas for marine life, including fish, squid, octopus, and shellfish, and provided for the continuation of ancient cultural practices, like shell-stringing. The seagrass beds in Tayaritja Sea Country are particularly significant for the Palawa community because these islands have been and remain critical to the survival of the Palawa cultural traditions like shell stringing, with many necklace making shells reliant on healthy seagrass beds.

Currently the Pakana Rangers are working to establish the first Sea Country Indigenous Protected Area (IPA). The proposed IPA is the Tayaritja Wukalina milaythina muka IPA, which is located in northeast Lutruwita/Tasmania extending across the land bridge that once connected the Furneaux Islands to mainland Australia (Figure 1).

For these reasons, the Palawa community have identified seagrass beds (defined as an ecosystem, including marine animals that rely on these habitats) as a target for the proposed Tayaritja Wukalina milaythina muka IPA.

Through establishing a collaborative partnership to deliver this project, the Pakana Rangers have exerted Palawa rights to data on culturally significant shellfish, and supported and facilitated the fieldwork, including by deploying a BOSS camera system to assess the extent and composition of seagrass beds throughout the study area. This has enabled Pakana Rangers to exercise their unceded rights and recognised their enduring responsibility to care for their Country. It has also supported capability development, and raised awareness of the value of consistent, standards-based approaches to collecting, analysing, managing, and reporting environmental data on seagrass beds in the area.

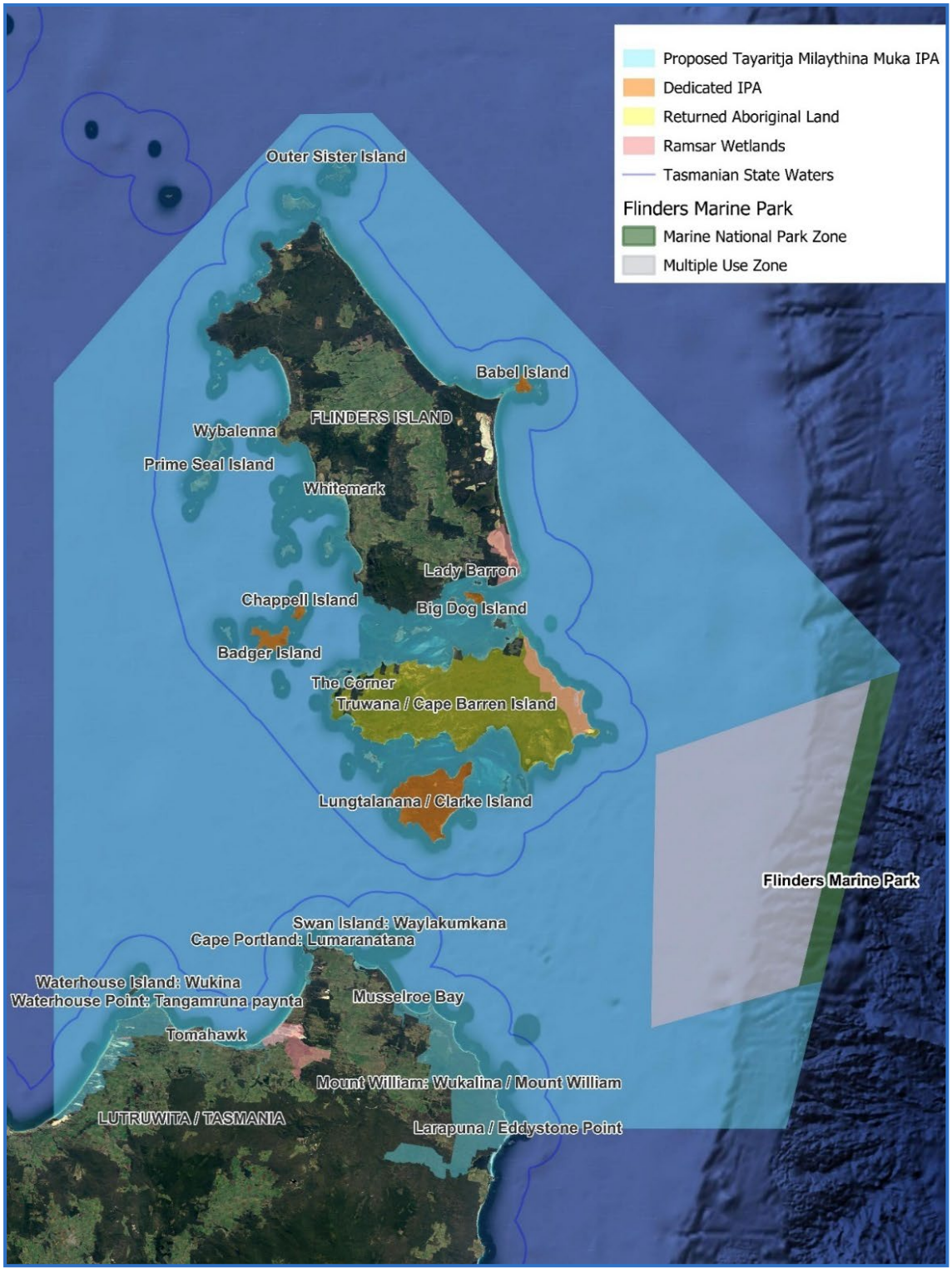


Figure 1 Proposed Tayaritja Wukalina milaythina muka Indigenous Protected Area and the project study area.

While the Palawa community is concerned about some observed changes to seagrass beds around the islands and are deeply concerned about how the seagrass beds will respond to

increasing oceanic changes being driven by climate change; seagrass beds around the islands are in generally good condition.

In this context, Palawa community have identified the following focal areas for seagrass monitoring:

- Focusing monitoring effort on culturally important areas
- Monitoring the abundance and condition of cultural resources at seagrass monitoring sites (such as shellfish)
- Improving understanding of drivers and rates of change.

1.3 Conservation and management in temperate Australia

1.3.1 Threats to seagrass

Seagrass beds are experiencing global decline, with measured losses in seagrass extent estimated between 3,370 km² (Waycott et al., 2009) and 5,602 km² (Dunic et al., 2021). Recent assessments suggest that approximately 19% of the known global seagrass area has been lost. However, this figure likely underrepresents the true extent of decline, as current data may account for only a tenth of the actual global distribution (Dunic et al., 2021). In the Southern Temperate Ocean Zone, seagrass beds are undergoing slow but consistent reductions in area, with Australia estimated to have lost approximately 5.5% of its seagrass coverage since the 1930s (Statton et al., 2018).

The primary drivers of seagrass loss are anthropogenic, particularly declines in water quality resulting from nutrient enrichment (Van Katwijk et al. 2016). Excessive nitrogen and phosphorus inputs from catchment run-off can disrupt seagrass physiology, stimulate the overgrowth of epiphytes, and promote blooms of phytoplankton, all of which reduce light availability critical for photosynthesis. Turbidity is further exacerbated by sediment runoff from agriculture, stormwater discharge, and coastal development (Burkholder et al., 2007). In addition to water quality degradation, direct physical disturbances such as land reclamation, dredging, and damage from boat moorings and anchors contribute significantly to seagrass habitat loss (Duarte, 2002).

Climate change intensifies these existing pressures through a range of mechanisms. Rising sea surface temperatures are likely to increase thermal stress and mortality in some seagrass species, while driving latitudinal shifts in species distributions (Olsen et al., 2018). Climate-related events – including intensified storm activity, cyclones, and marine heatwaves – can cause acute habitat degradation. For example, a marine heatwave in 2010–2011 in Shark Bay, Western Australia, led to the loss of over 1,310 km² of dense seagrass meadow (Strydom et al., 2020).

In response to these declines, several conservation measures have been implemented. These include legislative protections for vulnerable species, such as *Posidonia australis* in some region of New South Wales (NSW Government 2012) and *Ruppia megacarpa* in Tasmania (Threatened Species Section 2025). In addition, many seagrass habitats benefit from inclusion in protected areas such as World Heritage Areas, Ramsar-listed wetlands, and other marine conservation zones, offering some safeguards against further degradation.

1.3.2 Protection for seagrass

Globally, 29% of documented seagrass areas are located within 40 of the 50 marine World Heritage Areas (WHAs), although significant portions of seagrass habitat remain unmapped (Losciale et al. 2022). It is estimated that WHAs store around 25% of global seagrass carbon, yet at least 19 of the 50 marine WHAs have not been assessed for their carbon stock contributions. Additionally, at least 10 WHA sites contain more than 10,000 hectares of seagrass habitat (UNESCO 2020).

In temperate Australia, however, the representation of seagrass within WHAs is limited. The only designated marine WHA in temperate waters is the Lord Howe Island Group, which includes at least one lagoon with seagrass, accounting for just 0.9% of the surveyed benthic habitat (Edgar et al., 2009).

More extensive protection for seagrass in temperate Australia is provided through Ramsar-listed wetlands. These include Corner Inlet, Western Port, and Gippsland Lakes in Victoria; The Coorong in South Australia; and Pittwater–Orielton Lagoon and Moulting Lagoon in Tasmania. These sites safeguard a range of seagrass habitats, although some have experienced significant degradation. For instance, *Ruppia* stocks in the Coorong have been severely affected by drought, prompting substantial restoration efforts (Sinclair et al., 2021). In Tasmania, Pittwater–Orielton Lagoon lost an estimated 94% of its seagrass cover prior to 1990 (Rees, 1993), with ongoing declines reported in more recent assessments (NRM South, 2012).

Many Ramsar sites in Australia were designated primarily due to their importance for waterbirds, including migratory shorebirds (Australian Government Department of the Environment and Energy, 2017). Seagrasses support these bird populations directly by providing foraging grounds and indirectly by serving as nursery habitats for fish, which in turn benefit piscivorous birds (Unsworth et al., 2021). As a result, conservation efforts focused on protecting waterbirds have indirectly contributed to the preservation of seagrass ecosystems.

Despite these protective measures, large areas of temperate Australia's seagrass habitats remain inadequately documented (McKenzie et al., 2020), limiting the effectiveness of conservation and management strategies. Accurate and up-to-date maps are critical for understanding seagrass distribution, monitoring change over time, and setting informed conservation priorities.

1.4 Seagrass mapping in Australia

Over 31% of the mapped global area of seagrass (266,562 km^2) is located in Australia (83,013 km^2), largely due to the extensive mapping efforts in the Great Barrier Reef (GBR), with at least 395 individual mapping events recorded in the GBR before 2020 (McKenzie et al., 2022). In contrast, less than 10% of mapped area is in temperate regions of Australia (8433 km^2) (McKenzie et al., 2020). This suggests a large gap in mapping of seagrass habitats in temperate Australia.

1.4.1 Recent mapping in Australia (past decade)

Much of the extensive mapping conducted in the GBR has been field based (McKenzie et al., 2022), however a range of remote sensing options are offering potential solutions to

some of the difficulties of seagrass mapping. To identify mapping methods and ground validation methods being currently utilised around Australia for mapping seagrass we reviewed literature using Scopus and Google (search terms: (remote sensing OR satellite OR ground truth* OR mapping OR monitoring) AND (seagrass)) restricting returned citations to those from Australia from Jan 1, 2014, onward. A total of 308 references were generated and 65 references retained after title and abstracts were scanned for relevance. Many of these references related to technical aspects of mapping such as developments in machine learning or comparisons between remote sensing approaches, only 26 references described actual seagrass mapping events which were of interest to us in understanding seagrass remote sensing and ground validation methodologies currently in use in Australia.

A summary of the recent published seagrass mapping reports across each state conducted through remote sensing, including the field validation approaches used, are presented in Table 1. Unsurprisingly, the number of mapping studies from Queensland was equal to the total number for all other states combined. Notably there have been no seagrass mapping studies published for Lutruwita/Tasmania in the last decade (2015–2025) (Table 1).

Table 1. Published accounts of seagrass mapping through remote sensing since 2014, identified by Australian state and highlighting the mapping approaches and field validation approaches used.

State	Seagrass genus	Mapping approach	Field validation methods	Mapping accuracy	Reference
New South Wales	*Posidonia*	Nearmap	Photoquadrats for species and % cover	Not stated	Evans et al. 2018
	Zostera, Posidonia	UAV	None	Not stated	Simpson et al. 2024
	Posidonia, Zostera, Halophila, Ruppia	Satellite	Older maps validated with sidescan and underwater video	Not stated	West & Glasby 2021
Queensland	*Halodule, Halophila, Zostera, Cymodocea*	MODIS Level-0 data acquired from the NASA Ocean Colour website	Helicopter, diver, visual survey quadrats for biomass ranking, species composition, percent cover algae, depth, sediment type, GPS position	Not stated	Petus et al. 2014
	not stated	Sentinel-2 surface reflectance dataset in Google Earth Engine.	Field-measured bathymetry samples	Not stated	Li et al. 2021
	not stated	Quickbird 2	Photoquadrats - benthic cover	61.6% habitat type	Roelfsema et al. 2018
	Halophila, Halodule, Zostera, Cymodocea, Syringodium	Landsat 8, Sentinel-2, Ziyuan-3A, and WorldView-3	Photo transects for species composition and % cover	66% for species, 57% percent cover	Kovacs et al. 2018
	Halophila, Halodule, Zostera, Cymodocea, Syringodium	LANDSAT	Spot checks - seagrass presence/absence	63^% presence/absence	Kovacs et al. 2022
	Halophila, Halodule, Zostera, Cymodocea, Syringodium	WorldView-2, IKONOS andQuickbird-2 - creating semi-automated approach	Photoquadrats with towed GPS for species, % cover - point count analysed, Biomass cores	77% species composition, 52% percent cover,	Roelfsema et al. 2014

	Halophila, Halodule, Zostera, Cymodocea, Syringodium		AUV - comparison of snorkeller and AUV data - species composition	Not stated	Roelfsema et al. 2015
	Cymodocea, Enhalus, Halophila, Halodule, Syringodium, Thalassodendron, Thalassia, Zostera	Synthesis of prior mapping – various methods	None	Not stated	Carter et al. 2023
	Halophila, Halodule, Zostera, Cymodocea, Syringodium	Synthesis of prior mapping – various methods	None	Not stated	Carter et al. 2021
	not stated	In situ high-resolution hyperspectral mapping of the seafloor utilizing a spectrometer colocated and coregistered with a high-resolution color stereo camera system onboard an autonomous underwater vehicle (AUV)		Classification accuracy 91.99%	Bongiorno et al. 2018
	Halophila ovalis, H. spinulosa, Halodule uninervis, Zostera muelleri, Syringodium isoetifolium and Cymodocea serrulata	Quickbird-2 and Worldview-2 maps previsouly created	Photo transect surveys with GPS for seagrass %cover and species composition + biomass cores	48-58% for seagrass cover, 68-80% for species	Lyons et al, 2015
	15 species (not listed)	PlanetScope and UAV-captured imagery, UAS acquired nadir imagery			McKenzie, Langlois & Roelfsema 2022
South Australia	*Posidonia, Amphibolis*	Hyperspectral airborne imagery - A Diamond HK-36 ECO-Dimona aircraft fitted with a hyperspectral linescanner	Video transects	98%	Clarke et al. 2021
	Posidonia, Amphibolis, Zostera, Halophila	Video transects	Video transects - manually classified for changes in seagrass extent, species composition and extent cover	not stated	Tanner, 2021

	Posidonia, Amphibolis	Landsat imagery (1988–2018)	Underwater video transects	88-90%	Fernandes et al. 2022
Victoria	*Zostera, Heterozostera, Halophila, Amphibolis*	Sentinel 2 imagery and supervised object-based imagery analysis	Photoquadrats	77%-85%	Dalby et al. 2023
	General benthic habitat	Collation of various sources	Collation of various sources incl. monitoring, impact assessment, historical surveys	Not stated	Mazor et al. 2023
	Zostera, Halophila, Amphibolis	Historic aerial images	Underwater video, differential GPS for seagrass species, canopy height, seagrass density	>85% except at certain sites	Ball, Soto-Berelov & Young 2014
Western Australia	*Amphibolis, Posidonia*	Landsat and Sentinel 2	Drop camera Photoquadrats for % cover and species	65% - 80% depending on the year	Strydom et al. 2020
	Species not specified	Satellite imagery - type not specified	Video transects, manually classified by benthic habitat type	not stated	BMT Commercial Australia, 2023
	Halophila sp., P australis, A. antarctica	Satellite imagery - type not specified	Video transects, manually classified by benthic habitat type	not stated	BMT Commercial Australia, 2020
	Heterozostera sp., Halophila sp.	Aerial imagery used to update previous mapping	Video transects, manually classified by benthic habitat type	not stated	RPS, 2023
	Amphibolis, Posidonia, Halophila, Heterozostera, Syringodium	Nearmaps, Landsat,	none - Nearmaps compared to google earth points	Field biomass data collected	Vidyan 2018
Northern Territory	*Cymodocea, Enhalus, Halophila, Halodule, Syringodium, Thalassodendron, Thalassia, Zostera*	Synthesis of prior mapping – various methods	None	Not stated	Carter et al. 2023

1.4.2 Challenges in mapping from remote sensing

Mapping seagrass habitats using remote sensing techniques presents a suite of challenges arising from both environmental conditions and the biological characteristics of seagrass ecosystems. While satellite and aerial imagery can effectively detect seagrass in clear, shallow waters; sometimes down to depths of 20 m; such ideal conditions are relatively uncommon (Phinn et al., 2018; McKenzie et al., 2020). In many instances, seagrass occurs in turbid coastal and estuarine waters, where suspended sediments, coloured dissolved organic matter, and surface reflectance obscure bottom features, significantly reducing the accuracy of optical remote sensing.

The spectral similarity between seagrass and other benthic features, such as macroalgae, reef substrates, and detrital matter, further complicates classification. Seagrasses also exhibit considerable morphological variation between species, ranging from long, strap-like *Posidonia* to finer-leaved species such as *Zostera*, and occur at varying densities within beds. These factors alter the spectral signature and hinder reliable species-level or structural classification using remote sensing alone (McKenzie et al., 2020). Additionally, the presence of epiphytes and mixed-species assemblages can modify the reflectance properties of seagrass canopies, making it difficult to distinguish them from surrounding habitats without field-based validation (Phinn et al., 2018; Wicaksono et al., 2024).

High-quality ground-truthing data (see next section) is essential for calibrating and validating remotely sensed products, yet acquiring such data is time-consuming, expensive, and often logistically constrained, particularly in remote or deep-water environments. This limits the frequency and spatial extent of reliable seagrass mapping, especially in regions where long-term funding and capacity are lacking.

Temporal variability in seagrass coverage adds an additional layer of complexity. Some beds are highly dynamic, ephemeral beds exhibiting seasonal or interannual fluctuations in biomass and extent as part of natural ecological cycles (Walker et al., 2006). Capturing these dynamics requires repeated, high-resolution monitoring over time; something that current remote sensing and observation programs are not always equipped to provide.

Consequently, significant knowledge gaps persist in national and global seagrass datasets. For example, 27% of countries within seagrass bioregions remain data-poor in terms of mapped seagrass area (McKenzie et al., 2020). In Australia, a scoping study for the development of a national wetland inventory highlighted the need for a consistent, attribute-based classification approach to improve the mapping of intertidal and subtidal habitats. Priority areas identified include deep-water seagrass beds, turbid coastal systems in northern Australia, and large parts of Southern Australia and Lutruwita/Tasmania where mapping is either incomplete or outdated (Brock et al., 2022).

An illustrative case is the west coast of the Furneaux Group in Tasmania, where seagrass is known to extend beyond 20 m depth. Despite its ecological significance, this region has not been mapped since the late 1990s (Jordan et al., 1998), underscoring the limitations of existing remote sensing efforts and the pressing need for updated, systematic surveys.

1.4.3 Field Validation

Field validation, or ground-truthing, plays a critical role in enhancing the accuracy and reliability of remotely sensed seagrass data (Bremner et al., 2023). The choice of validation method depends on the specific objectives of the mapping exercise, the spatial scale of the project, time constraints, and the availability of resources. Common field-based approaches include the use of photoquadrats, video transects, video or photo point surveys, visual surveys, core sampling, and GPS-based boundary mapping.

Photoquadrats – typically ranging from 0.25 to 0.5 m² in size – are among the most frequently used validation tools. They can be deployed randomly, along transects, or targeted to specific features identified in remote imagery. These quadrats provide detailed visual records that support assessments of canopy cover, species composition, epiphyte load, and signs of stress such as bleaching or necrosis. Similarly, video methods, including the use of drop cameras (such as the BOSS used in this survey), remotely operated vehicles (ROVs), and autonomous underwater vehicles (AUVs), allow for spatially balanced or random sampling across broader areas. Multiple camera angles, including 360-degree footage, can be used to improve coverage and support fine-scale analysis, though understorey species may remain difficult to detect without physical sampling.

Visual confirmation surveys – conducted on foot in intertidal areas, or by snorkelling, diving, or aerial observation – offer a practical means of recording biomass, percent cover, species composition, and other biological indicators such as reproductive status and epiphyte load (McKenzie et al., 2003; Petus et al., 2014; Roelfsema et al., 2014; Kovacs et al., 2018). For example, helicopter-based assessments have been applied along the Great Barrier Reef (Carter et al., 2016). While these approaches can provide detailed field assessments at fine spatial scales, they require considerable observer expertise and training to ensure data consistency and accuracy.

Boundary tracking using handheld or vessel mounted GPS is another low-tech yet effective method, particularly for delineating meadow edges in turbid systems (Ierodiaconou et al., 2002). However, it assumes habitat homogeneity unless supported by additional sampling and can be subjective in areas where seagrass distribution is patchy (Bremner et al., 2023).

Core sampling provides precise estimates of above- and below-ground biomass, but it is labour-intensive and particularly challenging in dense, subtidal beds with large-rooted species such as *Posidonia*. Typically, samples are dried and weighed to quantify biomass, a process that can be time-consuming and destructive. Nevertheless, when combined with percent cover data from imagery, core samples provide valuable calibration data for estimating biomass indirectly and for generating biomass maps (Lyons et al., 2015).

Each field validation method has distinct strengths and limitations, and the choice of approach is often determined by the specific research questions and resource availability. For instance, while photoquadrats and video data require detailed post-processing – often using point-count software – they offer repeatable, verifiable records. In contrast, visual surveys are quicker to implement but depend heavily on observer accuracy in the field. Advances in deep learning and image classification tools are beginning to reduce the burden

of manual analysis and show promise for automating the interpretation of imagery (Langlois et al. 2023).

Although resource-intensive, field validation remains an essential component of robust seagrass mapping programs. These methods not only increase the confidence in remotely sensed outputs but also provide critical ecological insights into seagrass condition, composition, and change over time.

1.4.4 Filling a gap – mapping Tayaritja

Separating Lutruwita/Tasmania and Victoria is the Bass Strait, with a maximum width of 240 km and average depth between 50 – 70 m, it is bordered in the west by King Island and the Indian Ocean and Tayaritja/The Furneaux Group on its eastern side. Tayaritja contains more than 50 islands covering around 1992 km^2, the largest islands being Flinders Island, Truwana/Cape Barren and Lungtalanana/Clarke (Mort, 2006). Benthic surveys in the 1990s revealed extensive beds of *P. australis* on the west coast, as well as beds of intertidal *Z. muelleri*, and mixed beds of *P. australis*, *H. nigricaulis* (formerly *Heterozostera tasmanica*) and *A. antarctica,* as well as recording reports of deep water *Halophila australis* (Rees, 1993). LANDSAT imagery was later used to guide field activities involving underwater video surveys where seagrass was recorded to a depth of 20 m which was the limit of the survey. It was suggested, based on the standing stock biomass of *Posidonia* at that depth that the range for seagrass may be considerably deeper. This survey also recorded *Posidonia spp* which represented the most eastern extent of range recorded (Jordan et al., 1998).

Early surveys documented 212 km^2 of seagrass area encompassing Tayaritja, reported as part of a larger mapping project (Edyvane et al., 2000), with further mapping of Tayaritja, the north coast and the northwestern corner of Lutruwita/Tasmania, near Robbins Island, identifying 530 km^2 of previously unrecorded seagrass beds. At the time this added to the known area of seagrass for Lutruwita/Tasmania to be over 1000 km^2. (Jordan et al., 1998). Tayaritja represents a known area of extensive seagrass beds from shallow through to deep water where only very coarse scale mapping has previously been applied. This makes the region a priority for improving understanding of a potentially vast natural asset, as well as the development of remote sensing workflows that can be applied to deep water temperate seagrass beds.

1.6 Conclusion

Seagrass habitats in temperate Australia represent ecologically and functionally significant systems, supporting biodiversity, contributing to coastal productivity, and offering substantial potential for blue carbon storage. Despite their importance, large areas of temperate seagrass remain poorly mapped or monitored, constraining effective conservation and management. Current protective mechanisms – such as World Heritage Areas and Ramsar-listed wetlands – have offered some indirect benefits, yet gaps in spatial data, particularly in turbid and deep-water environments, limit the ability to assess habitat extent, condition, and change over time. Remote sensing offers valuable tools for large-scale seagrass mapping;

however, environmental and biological complexities present persistent challenges, underscoring the continued need for field validation and methodological refinement.

To fully realise the role of seagrass in climate mitigation and biodiversity conservation, future efforts must prioritise sustained research, update national inventories, and support inclusive approaches that engage Aboriginal communities, recognising the critical role that Aboriginal people have to play in caring for culturally significant ecosystems.

Addressing knowledge gaps in mapping and carbon stock assessment will require coordinated investment in technological innovation, consistent classification frameworks, and long-term monitoring programs. Together, these actions will enhance the scientific foundation needed to safeguard seagrass ecosystems and their associated values in a changing coastal landscape

2. Ecological significance of seagrass in the Tayaritja Sea Country

Seagrass is an aquatic angiosperm found in estuarine and brackish waters across the globe (Larkum, 2006; do Amaral Camara Lima et al., 2023). Collectively seagrasses form dense beds where they make important contributions to coastal biodiversity and provide a range of critical ecosystem services (Cullen-Unsworth and Unsworth, 2013; Nordlund et al., 2018). The biodiversity in these habitats is influenced by seagrass biomass and epiphyte cover, which provide structural complexity and food resources for epifaunal communities (Orth et al., 1984; Borg et al., 2010; McHenry et al., 2021; Rodil et al., 2021). Different seagrass species form beds of varying densities, which can affect epiphyte growth and, in turn, the abundance and diversity of associated fauna (Edgar, 1990; Como et al., 2008; Micheli et al., 2008; Morris et al., 2008; Brun et al., 2009; van der Heide et al., 2012; Gartner et al., 2013). The extent to which different seagrass species drive variation in epifaunal communities remains debated, with conflicting evidence on functional redundancy – the idea that different species support similar epifaunal compositions (Loreau, 2004; Micheli et al., 2008; Christiaen et al., 2016). While some studies suggest that shifts in dominant seagrass species have minimal effects on epifauna communities, others highlight the role of morphological differences or epiphyte cover and facilitation cascades in structuring habitat and food availability (Hemminga and Duarte, 2000; Duffy, 2006; Christiaen et al., 2016).

Molluscs are key ecosystem indicators due to their sensitivity to environmental changes, including variations in nutrient availability, temperature, and hydrodynamic forces (Reguera et al., 2018; Fales et al., 2020). Their abundance, richness, and productivity are strongly influenced by the presence of epiphytes, which provide essential food sources, particularly for grazing gastropods (Kitting et al., 1984; Borowitzka et al., 2006). In regions like Tayaritja (Furneaux Islands), where seagrass beds remain understudied, understanding how different seagrass species contribute to mollusc biodiversity is crucial to predicting any consequences of habitat change (Macreadie et al., 2018; Ridgway and Ling, 2023).

Despite the ecological and cultural importance of these seagrass habitats, the last comprehensive study of seagrass beds in the region occurred decades ago and without any engagement with the Palawa community, leaving significant gaps in knowledge (Rees, 1993; Department of Primary Industries, Water and Environment Tasmania, 1999; Edgar et al., 2000). Further research is needed to determine whether facilitation cascades between seagrass, epiphytes, and molluscs drive mollusc biodiversity patterns in this region, or if different seagrass species provide similar ecological functions, mitigating biodiversity loss through functional redundancy (Thomsen et al., 2010; Thomsen et al., 2018; Zhang and Silliman, 2019).

To address these gaps, this study assessed the influence of the three most common seagrass species (*Amphibolis antarctica, Posidonia australis,* and *Heterozostera nigricaulis*), their biomass and cover, as well as associated epiphytes, on the mollusc communities in Tayaritja Sea Country. Given that seagrass biomass and cover directly relate to habitat complexity and resource availability (Orth et al., 1984; Borg et al., 2010; McHenry et al.,

2021; Rodil et al., 2021), we hypothesised that increases in seagrass biomass, cover, and epiphyte cover would enhance mollusc abundance and richness. Additionally, we expect variation in mollusc communities between sites, seagrass species, and epiphyte cover, as each seagrass species supports distinct epiphyte assemblages that influence mollusc diversity (Hemminga and Duarte, 2000; Duffy, 2006; Christiaen et al., 2016). The results of this study will help to understand the role of these the dominant seagrass species on mollusc biodiversity and will provide a baseline for future research and conservation efforts in Tayaritja Sea Country.

2.1 Methods for mollusc and seagrass surveys

Fieldwork was conducted in the Tayaritja Sea Country, located off the north-east coast of Tasmania, between February 6 and February 15, 2024. Five sites were sampled: two on Flinders Island (Petrification Bay and Killiecrankie) and three on nearby offshore islands (Truwana/Cape Barren Island, Big Dog Island, and Chalky Island), (Figure 2). These sites were chosen in collaboration with Pakana Rangers due to the presence of three seagrass species (*A. antarctica, P. australis,* and *H. nigricaulis*) at each location excluding Petrification Bay which only had *P. australis, and H. nigricaulis*. A sonde (EXO2 Multiparameter Sonde) was deployed for approximately 4 hours at each site to measure dissolved oxygen, temperature, and turbidity in the seawater. Water temperature ranged from 22.4°C to 24.4°C, dissolved oxygen from 8.66 mg/l to 10.03 mg/l, and turbidity ranged from 2.5 FNU to 5.1 FNU. Water temperature, dissolved oxygen, and turbidity were generally similar across sites, with no clear links to seagrass or epiphyte cover.

Sampling occurred around low tide, lasting ~ 4 hours, with snorkelling conducted in the subtidal zone (see Sinclair 2024 for details). At each site, 15 patches of different sizes were selected for each seagrass species, spaced at least 5 meters apart, to capture varying levels of epiphyte cover. Nine of these patches were randomly selected for sampling. A 0.5 x 0.5 m quadrat was used, divided into four 0.25 x 0.25 m subsections, and one subsection per quadrat was sampled based on epiphyte cover. One subsection per quadrat was photographed for quantification of seagrass and epiphyte cover, and the seagrass leaves, and associated epiphytes were collected by hand and placed into plastic bags.

Molluscs were separated from the samples by washing samples through a 1 mm mesh nylon bag. The molluscs were preserved in 3% formalin and later identified to the lowest taxonomic level possible under a microscope. Seagrass and epiphyte percentage cover for each quadrat was quantified using ImageJ software, and the dry biomass of leaves was measured after oven drying the samples at 80C for 48 hours. Mollusc biomass per gram of seagrass and secondary productivity per gram of seagrass were calculated following the methods of Edgar et al. (1990).

Generalised linear models (GLMs) or multivariate GLMs were used to assess the influence of 1) site, seagrass species, seagrass dry biomass, seagrass cover, and total epiphyte cover on mollusc species richness, abundance and community composition, and 2) the influence of site, seagrass species, seagrass dry biomass and total epiphyte cover on mollusc biomass and secondary productivity. GLMs or multivariate GLMs were then used to assess the

interactions between site, seagrass species, and total epiphyte cover on 3) mollusc species richness, abundance, biomass and secondary productivity and 4) community composition. The results from the multivariate analyses were shown in non-metric multidimensional scaling (NMDS) plots. Pairwise comparisons were conducted to explore sources of variation in the explanatory variables. Statistical analyses were performed using R and RStudio, with assistance from ChatGPT for code development.

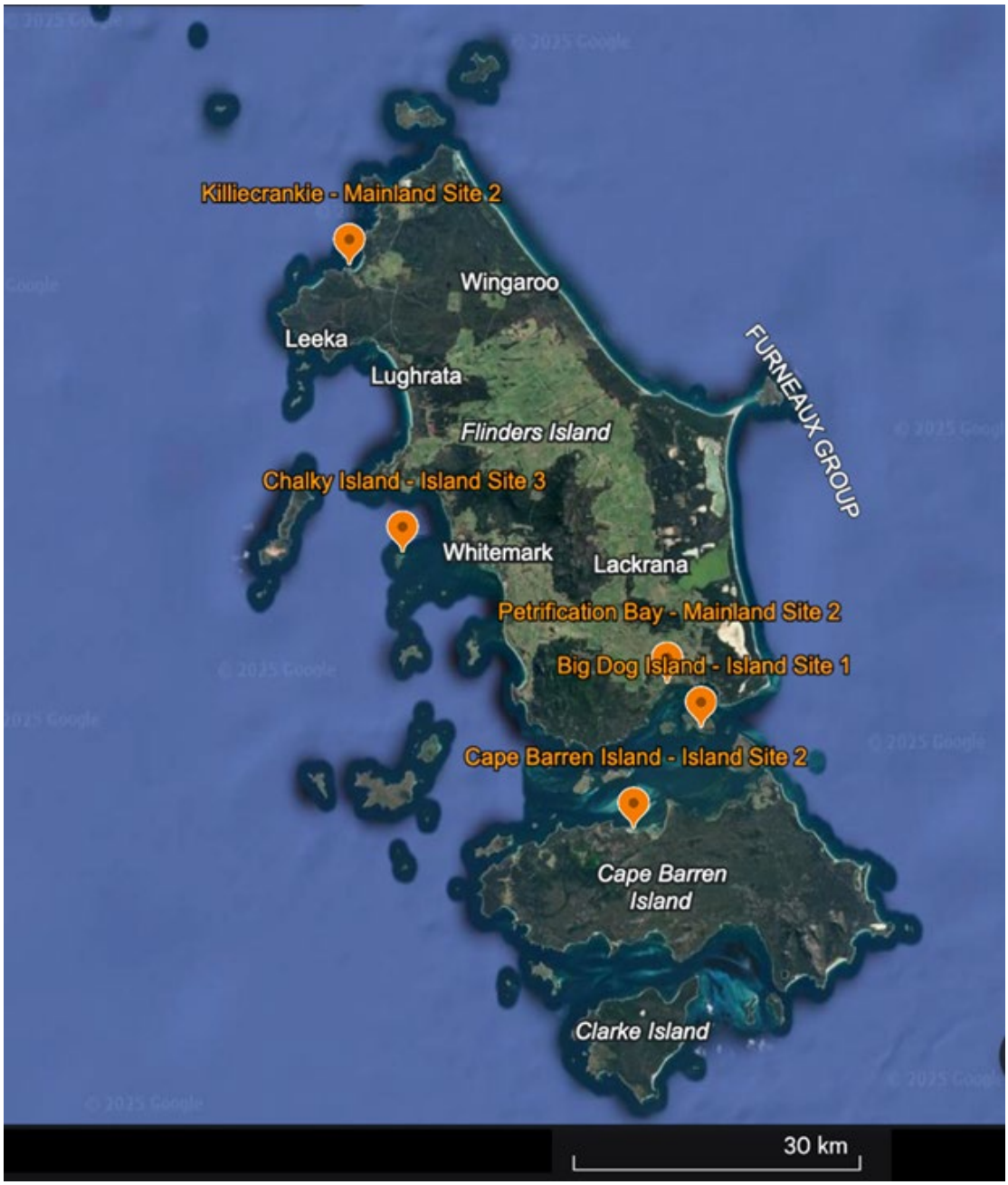


Figure 2 The location of the five sample sites in the Furneaux Group/Tayaritja Sea Country. Shown here with Truwana/Cape Barren Island and Lungtalanana/Clarke Island. Produced on Google Earth (2019).

2.2 Results

There were 657 individual molluscs, representing 15 unique taxa identified across the three seagrass species and five sampling sites. *Cantharidella tiberiana* was the most abundant species. Of the five sites sampled, Big Dog Island had the highest abundance of molluscs, with 198 individuals, in comparison to Killiecrankie which had only 80 individuals.

2.2.1 Relationships between seagrass species and dry biomass and cover and epiphyte cover and site and mollusc richness, abundance, biomass and secondary production

There were significant positive relationships between mollusc richness and abundance and the dry biomass of seagrass (Table 2; Figs. 2–3). Seagrass species also significantly influenced mollusc richness with *P. australis* and *A. antarctica* supporting a greater number of mollusc taxa compared to *H. nigricaulis* (Fig. 3). In contrast, mollusc biomass was higher on *H. nigricaulis* than *P. australis* and *A. antarctica* (Fig. 4) while the secondary production of molluscs was greater on *P. australis* than on either *A. antarctica* or *H. nigricaulis* reflecting the presence of more small-bodied molluscs on *H. nigricaulis* and a greater number of larger *individuals on P. australis* (Fig. 5). These findings suggest that certain seagrass species – particularly those with higher biomass – may offer more abundant food and shelter resources, thereby supporting greater mollusc diversity and secondary productivity.

There were also significant differences between sites in mollusc abundances and secondary production with higher abundances recorded at Big Dog and Truwana/Cape Barren Islands than Chalky Island or Killiecrankie (Figs 2, 4). Big Dog Island had a higher secondary production of molluscs than all the other sites sampled (Fig. 4). Mollusc abundance did not differ significantly among seagrass species (Table 2). Additionally, seagrass cover and epiphyte cover had no detectable effects on bivalve richness, abundance, biomass or secondary production (Table 2).

Table 2. Results of GLMs (generalised linear models) testing the effects of seagrass species, biomass and cover, epiphyte cover and site on mollusc a) richness, b) abundance, c) biomass (mg per g seagrass), and d) secondary production (mg/day per g seagrass). Significant p-values are indicated in bold print.

a) Richness					
	df	Deviance	Residual df	Residual deviance	P-value
Epiphyte cover	1	0.173	103	101.14	>0.05
Seagrass biomass	1	14.724	104	101.32	**<0.001**
Seagrass cover	1	0.317	102	100.83	>0.05
Seagrass species	2	10.527	105	116.04	**0.005**
Site	4	6.628	107	126.67	>0.05
b) Abundance					
	df	Deviance	Residual df	Residual deviance	P-value
Epiphyte cover	1	53.84	103	3604.4	>0.05
Seagrass biomass	1	325.00	104	3658.2	**0.002**
Seagrass cover	1	25.09	102	3579.3	>0.05
Seagrass species	2	102.80	105	3983.2	>0.05
Site	4	326.95	107	4086.0	**0.008**
c) Biomass (mg per g seagrass)					
	df	Deviance	Residual df	Residual deviance	P-value
Epiphyte cover	1	0.017	103	8.58	>0.05
Seagrass cover	1	0.001	102	8.582	>0.05
Seagrass species	2	2.882	105	8.645	**<0.001**
Site	4	2.882	105	8.645	<0.05
d) Secondary production (mg/day per g seagrass)					
	df	Deviance	Residual df	Residual deviance	P-value
Epiphyte cover	1	0.131	103	3.22	>0.05
Seagrass cover	1	0.633	102	3.19	>0.05
Seagrass species	2	2.321	105	3.24	**0.016**
Site	4	37.147	107	3.56	**<0.001**

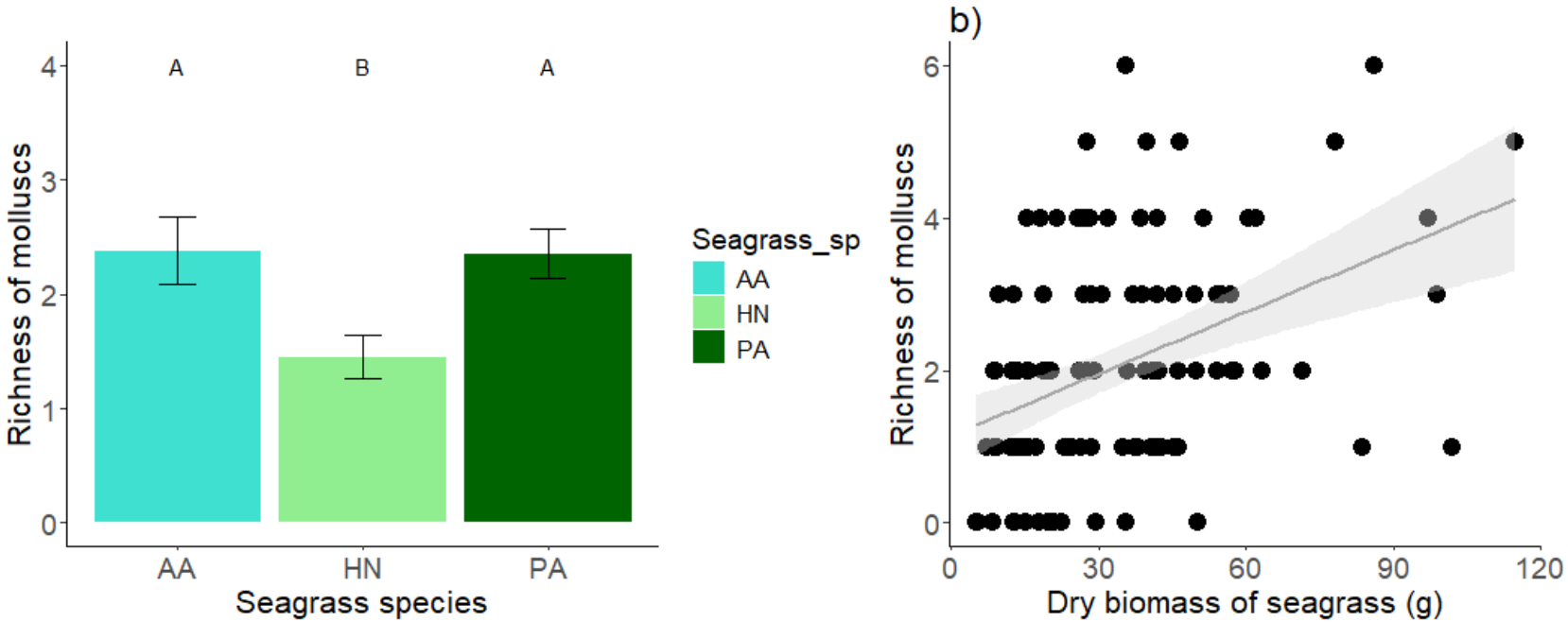


Figure 3. Effects of a) seagrass species (AA = *A. antarctica*, HN = *H. nigricaulis*, and PA = *P. australis*) on the mean (± SE) mollusc richness, and b) the relationship (± CI) between seagrass dry biomass and mollusc richness across all sites in Tayaritja/Furneaux Group. Different letters denote significant differences between groups based on post hoc pairwise comparisons.

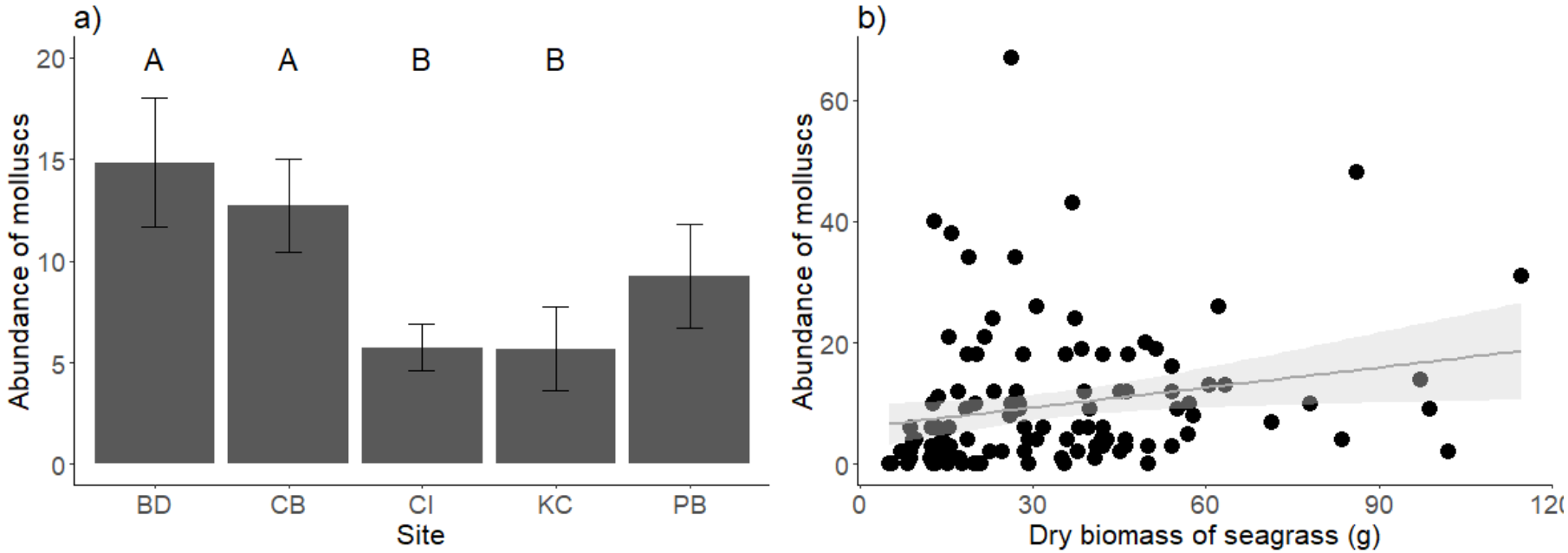


Figure 4. Effects of a) site on mean (± SE) bivalve abundances, and b) relationship (± CI) between seagrass dry biomass (g) and bivalve abundances across all sites (BD = Big Dog Island, CB = Truwana/Cape Barren, CI = Chalky Island, KC = Killiecrankie, PB = Petrification Bay) in Tayaritja/Furneaux group. Different letters denote significant differences between groups based on post hoc pairwise comparisons.

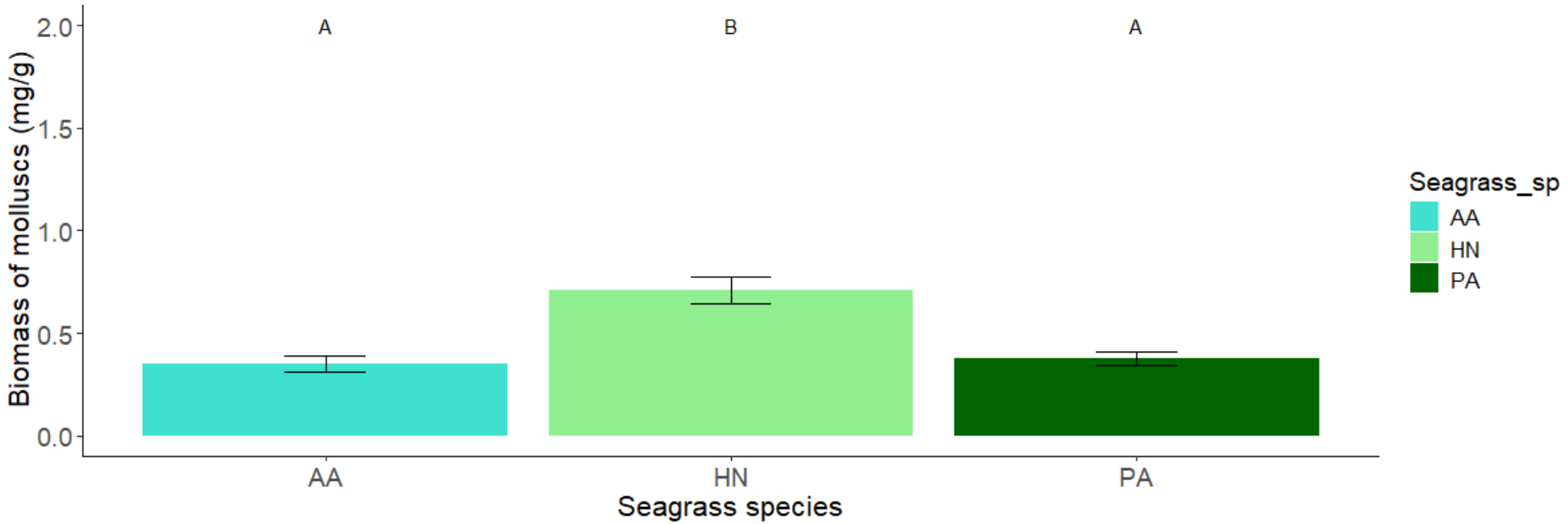


Figure 5. Effects of seagrass species on the mean (± SE) bivalve biomass (mg per gram of seagrass) across all sites in Tayaritja/Furneaux group. Seagrass species abbreviations as in Figure 2. Different letters denote significant differences between groups based on post hoc pairwise comparisons.

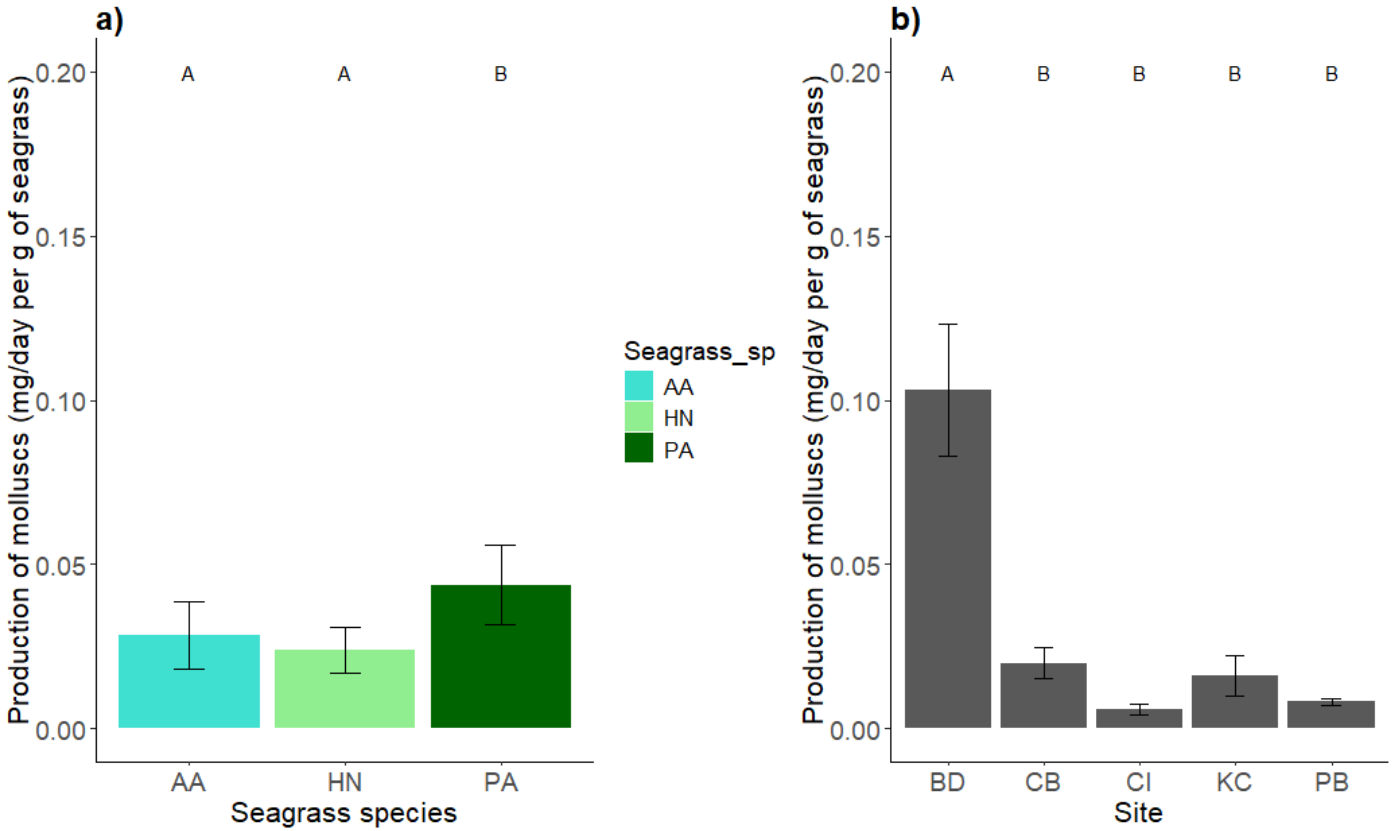


Figure 6. Effects of a) seagrass species and b) site on the mean (± SE) bivalve secondary production (mg per gram of seagrass) across all sites in the Tayaritja/Furneaux group. Seagrass species and site abbreviations as in Figures 2 and 3. Different letters denote significant differences between groups based on post hoc pairwise comparisons

2.2.2 Effects of and interactions between seagrass species, epiphyte cover and site on mollusc richness, abundance, biomass and secondary production

There was a significant effect of seagrass species on mollusc richness, biomass and secondary production, which varied across sites, while total epiphyte cover had no overall effects on these metrics (Figs. 7-9, Table 3). For mollusc secondary production, there was also a significant effect of site but no interactions between total epiphyte cover, seagrass species and site (Table 2). By contrast, the effects of both seagrass species and total epiphyte cover on mollusc abundance varied significantly among sites (Table 2).

Mollusc richness was higher on *P. australis* and *A. antarctica* compared to *H. nigricaulis* at Chalky Island and Killiecrankie, and higher on *P. australis* than *H. nigricaulis* at Petrification Bay (Fig. 6). Mollusc biomass was higher on *H. nigricaulis* than either *P. australis* and *A. antarctica* at Big Dog, Truwana/Cape Barren and Chalky Islands (Fig. 8). The biomass of molluscs was also higher on *A. antarctica* on Big Dog relative to Truwana/Cape Barren and Chalky Islands (Fig. 8). For *H. nigricaulis* there was a higher biomass of molluscs on Truwana/Cape Barren compared with Chalky Island (Fig. 8). For *P. australis* the biomass of molluscs was higher on Petrification Bay than Truwana/Cape Barren Island (Fig. 8). In contrast, no significant differences in mollusc richness and biomass among seagrass species were observed at the other sampled sites (Figs. 6, 8).

The influence of total epiphyte cover on mollusc abundance varied across sites and seagrass species (Fig. 7). At Big Dog and Truwana/Cape Barren islands, increased epiphyte cover on *P. australis* was associated with higher mollusc abundance, suggesting that epiphytes may enhance habitat complexity or resource availability (Fig. 7). In contrast, at Chalky Island and Petrification Bay, increased epiphyte cover on *P. australis* corresponded with lower mollusc abundance, indicating potentially negative effects (Fig. 7).

At Big Dog Island, in contrast to *P. australis*, mollusc abundance decreased with increasing epiphyte cover for *A. antarctica* and *H. nigricaulis* (Fig. 7). Similarly, at Truwana/Cape Barren Island, for *H. nigricaulis* mollusc abundance showed a negative relationship with epiphyte cover (Fig. 7). For *A. antarctica*, the effects of epiphyte cover on mollusc abundance were not significant across sites (Fig. 7).

Table 3. Results of GLMs (generalised linear models) testing the effects of and interactions between seagrass species, epiphyte cover and site on mollusc a) richness, b) abundances, c) biomass and d) secondary production. Significant p-values are indicated in bold print.

a) Richness					
	df	Deviance	Residual df	Residual deviance	P-value
Epiphyte cover	1	1.004	106	125.562	>0.05
Seagrass species	2	10.735	104	114.827	**0.005**
Site	4	1.004	107	126.566	>0.05
Epiphyte cover x Seagrass species	2	0.626	91	87.969	>0.05
Epiphyte cover x Site	4	0.889	100	113.942	>0.05
Seagrass species x Site	7	25.348	93	88.594	**0.007**
Epiphyte cover x Seagrass species x Site	7	9.190	84	78.779	>0.05
b) Abundance					
	df	Deviance	Residual df	Residual deviance	P-value
Epiphyte cover	1	15.124	106	592.28	**<0.001**
Seagrass species	2	19.673	104	572.61	**<0.001**
Site	4	57.99	107	607.41	**<0.001**
Epiphyte cover x Seagrass species	2	5.084	91	364.12	>0.05
Epiphyte cover x Site	4	22.089	100	550.52	**<0.001**
Seagrass species x Site	7	181.320	93	369.20	**<0.001**
Epiphyte cover x Seagrass species x Site	7	33.899	84	330.22	**<0.001**
c) Biomass (mg/g per gram of seagrass)					
	df	Deviance	Residual df	Residual deviance	P-value
Epiphyte cover	1	0.010	106	19.732	>0.05
Seagrass species	2	5.651	104	14.081	**>0.05**
Site	4	3.189	107	19.742	<0.001

Epiphyte cover x Seagrass species	2	0.424	91	9.585	>0.05
Epiphyte cover x Site	4	0.887	100	13.194	>0.05
Seagrass species x Site	7	3.186	93	10.008	**<0.001**
Epiphyte cover x Seagrass species x Site	7	0.537	84	9.048	>0.05
d) Secondary production (mg/day per g of seagrass)					
	df	Deviance	Residual df	Residual deviance	P-value
Epiphyte cover	1	0.034	106	3.528	>0.05
Seagrass species	2	0.301	104	3.228	**0.012**
Site	4	3.901	107	3.561	**<0.001**
Epiphyte cover x Seagrass species	2	0.131	91	2.794	<0.05
Epiphyte cover x Site	4	0.122	100	3.11	<0.05
Seagrass species x Site	7	0.181	93	2.93	<0.05
Epiphyte cover x Seagrass species x Site	7	0.229	84	2.564	<0.05

The composition of mollusc communities was shaped by a combination of total epiphyte cover, seagrass species, and site (Table 4). Non-metric multidimensional scaling (nMDS) ordination revealed distinct groupings of mollusc communities based on both site and seagrass species, and sites and cover of epiphytes of seagrass, with the exception of samples from Killiecrankie, which showed no clear clustering (Fig. 10). These patterns were primarily driven by higher abundances of mollusc taxa (*Bellastraea sp. (c.f) aurea, Calliostoma legrandi, Nassarius sp. (c.f) nigellus, Diala suturalis, Phasianotrochus rutilus*) that graze on or seek shelter within specific seagrass species or their associated epiphytes at particular sites (Fig. 10).

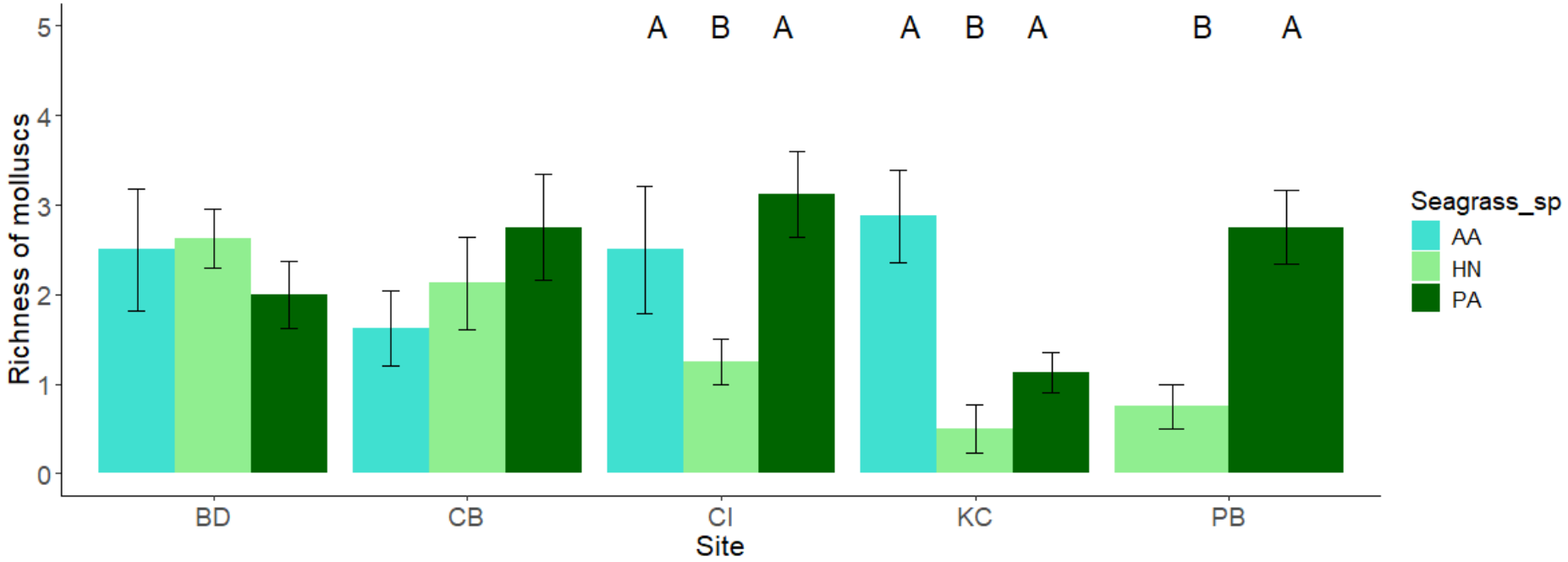


Figure 7. Effects of seagrass species and site on the mean (± SE) mollusc richness in Furneaux/Tayaritja Sea Country. Seagrass species and site abbreviations as in Figures 2 and 3. Different letters denote significant differences between groups based on post hoc pairwise comparisons.

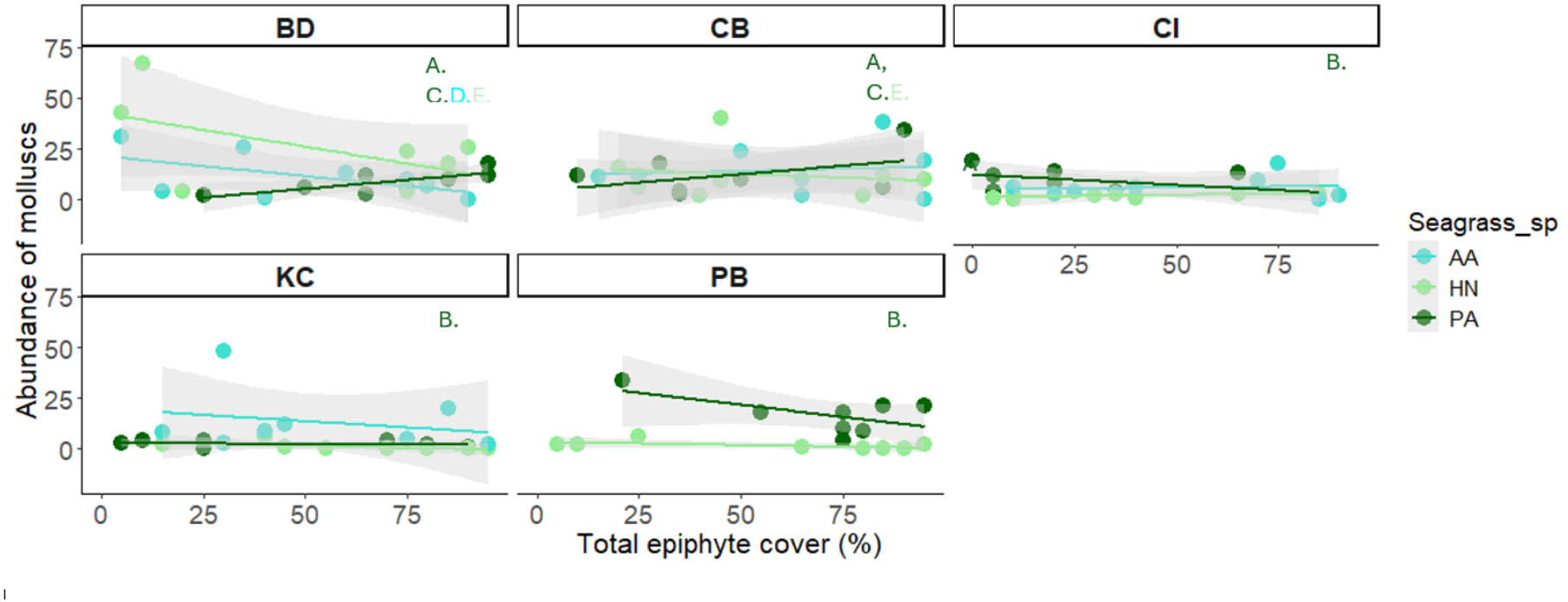


Figure 8. Relationship (± CI) between total epiphyte cover (%) and abundances of molluscs for each seagrass species at each site on the mean (+/-SE) in Furneaux/Tayaritja Sea Country. Seagrass species and site abbreviations as in Figures 2 and 3.

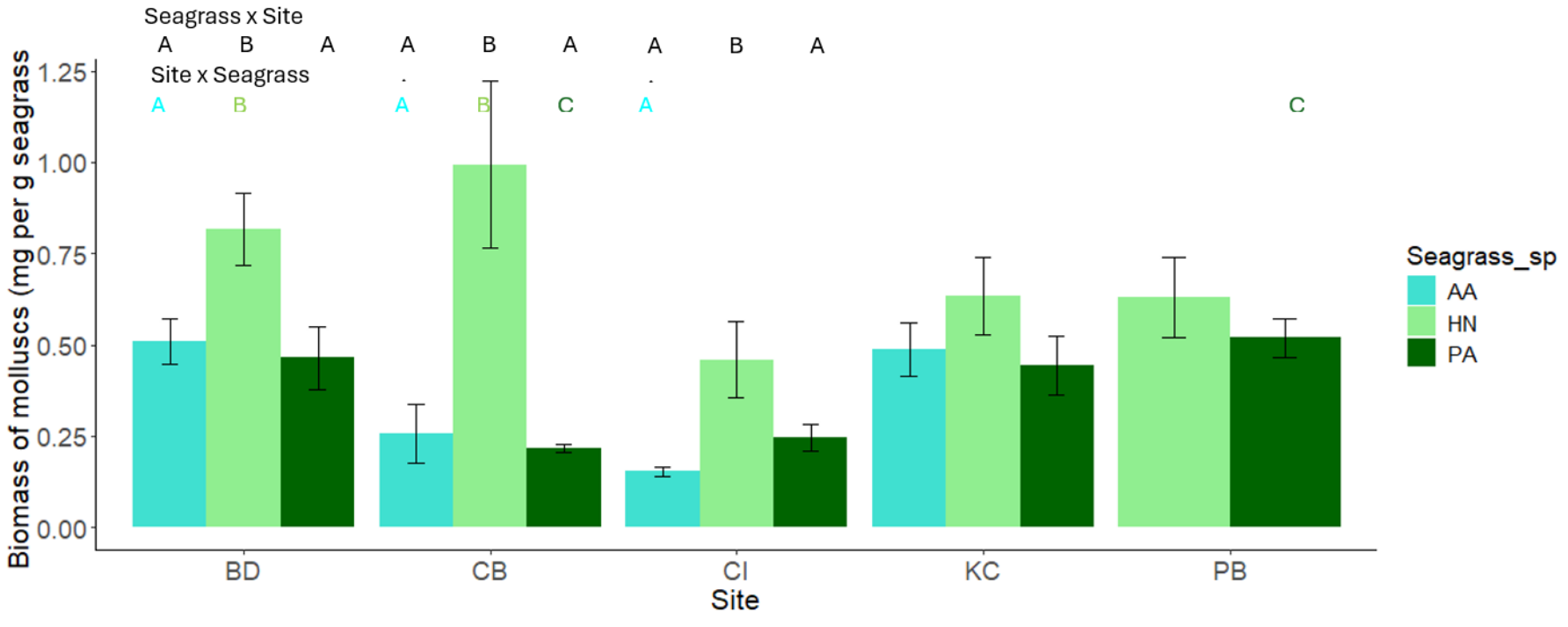


Figure 9. Effects of seagrass species and site on the mean (± SE) mollusc biomass (mg per g seagrass) in Furneaux/Tayaritja Sea Country. Different letters denote significant differences between groups based on post hoc pairwise comparisons i.e. Site x Species = testing the effects of different species within sites and Species x Site = testing the effects of species across different sites. Seagrass species and site abbreviations as in Figures 2 and 3.

Table 4. Results of multivariate GLM testing the effects of and interactions between seagrass species (AA= *A. antarctica*, HN = *H. nigricaulis*, PA = *P. australis*), total epiphyte cover (%) and site (BD = Big Dog Island, CB = Truwana/Cape Barren Island, CI = Chalky Island and KC = Killiecrankie) on the community composition of molluscs. Pair-wise testing was used to identify significant interactions affecting the community composition of molluscs. Significant p-values are indicated in bold print.

Multivariate glm	**Res.df**	**df.diff**	**Dev**	**P-value**
Intercept	111			
Seagrass species	109	2	22.46	**0.001**
Site	105	4	227.0	**0.001**
Epiphyte cover	104	1	22.46	>0.05
Seagrass species x Site	96	8	121.66	**0.001**
Seagrass species x Epiphyte cover	90	2	29.74	>0.05
Epiphytes cover x Site	92	4	116.94	**0.001**
Seagrass species x Epiphyte cover x Site	84	8	29.34	>0.05

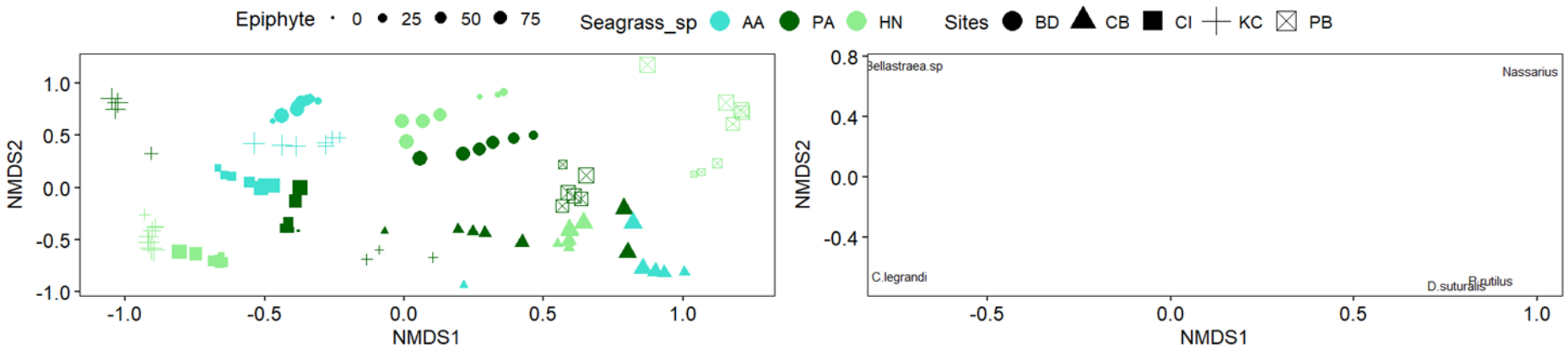


Figure 10. Non-metric Multidimensional Scaling (NMDS) ordination of mollusc community across five sites and three different seagrass species. The left hand NMDS plot shows the spatial arrangement of samples based on dissimilarities in mollusc composition, with each point representing an individual sample. Samples are colour-coded by seagrass species, and shapes represent the site, the size of the point represents the total epiphytic (%) cover in the samples. Seagrass species and site abbreviations as in Figures 2 and 3. The right hand NMDS plot shows the top 5 mollusc taxa (*Bellastraea sp. (c.f) aurea, Calliostoma legrandi, Nassarius sp. (c.f) nigellus, Diala suturalis, Phasianotrochus rutilus*) that contribute to the dissimilarity. NMDS stress = 0.215.

2.3 Species identity and cultural importance of seagrass for Tayaritja Sea Country

This study examined how the three dominant seagrass species in the Tayaritja Sea Country; *Amphibolis antarctica*, *Posidonia australis*, and *Heterozostera nigricaulis* – influence mollusc community structure. The findings reveal that higher seagrass biomass is positively associated with increased mollusc taxa richness and abundance. Seagrass species had a significant effect on mollusc richness, biomass and secondary productivity. Importantly, the impact of seagrass species on mollusc richness and biomass varied across sites, indicating context-dependent interactions. Likewise, the influence of both seagrass species and epiphyte cover on mollusc abundance and community composition differed between sites, underscoring the complexity of these ecological relationships and the critical role of local environmental conditions in also shaping mollusc communities.

At three of the five sampled sites, *A. antarctica* and *P. australis* supported higher mollusc taxa richness despite supporting lower mollusc biomass per gram of seagrass compared to *H. nigricaulis*. This suggests that while these species may promote mollusc diversity, they do not necessarily host a higher density of individuals per unit biomass. Notably, across all sites, mollusc secondary production per gram of seagrass was highest on *P. australis*, suggesting it plays a key role in sustaining not only diversity but also productivity. These results support the notion that seagrass species are functionally distinct in their capacity to support different aspects of mollusc communities, including richness, abundance, and biomass (Edgar et al., 1990).

Facilitation cascades, whereby the structure provided by seagrasses and their epiphytes are utilised by molluscs for habitat or food, were also evident in shaping mollusc community composition (Hemminga and Duarte, 2000; Duffy, 2006; Christiaen et al., 2016). These cascades were not uniform across the region. For example, at Big Dog and Truwana/Cape Barren islands, increased epiphyte cover on *P. australis* was associated with higher mollusc richness, whereas a similar increase in epiphytic cover on *A. antarctica* and *H. nigricaulis* corresponded with declines in richness. These patterns are likely driven by morphological differences among the seagrass species. *P. australis* features wide, flat leaves with relatively simple structure, which may benefit more from added complexity provided by epiphytes. In contrast, the naturally complex, densely packed forms of *A. antarctica* and *H. nigricaulis* may experience diminishing returns – or even negative effects – when epiphyte loads increase, potentially due to shading or smothering effects (Edgar et al., 1990).

However, the role of epiphytes varied substantially among sites, particularly on *P. australis*. At some sites, increased epiphyte cover led to elevated mollusc abundances, while at others the opposite trend was observed. These inconsistencies may reflect variability in epiphyte species composition, density, and structure across sites, each of which can influence mollusc communities via changes in habitat complexity, food availability, or microclimatic conditions. Such findings reinforce the importance of considering local context when interpreting seagrass-epiphyte-mollusc interactions.

Overall, this study highlights the critical role that seagrass habitat characteristics play in influencing the mollusc communities in Tayaritja Sea Country. Differences among *A. antarctica*,

P. australis, and *H. nigricaulis* – in terms of biomass, leaf morphology, and epiphyte cover – had significant and variable effects on mollusc richness, abundance, biomass, secondary production, and community structure among sites. These findings not only contribute to our ecological understanding of seagrass–mollusc dynamics but also have important implications for cultural practices.

Future work should include seasonal and long-term monitoring of seagrass-mollusc interactions, along with culturally guided sampling of necklace making shells; (Confidential report by Sinclair and Strain, 2024 Appendix 2). Such interdisciplinary approaches will support both ecological conservation and the continuation of important Palawa cultural traditions in Lutruwita (Tasmania).

3. Blue carbon storage assessment for seagrasses of Tayaritja

We quantified the belowground (seafloor) carbon stocks of seagrass beds surrounding Tayaritja. Our sampling aimed to address two key questions: (1) What is the spatial variability in carbon stocks within *Posidonia australis* beds, the dominant seagrass species in the area? and (2) Do belowground carbon stocks differ among seagrass species? To investigate spatial variability (Question 1), we sampled 10 sites distributed around Tayaritja (Figure 11). To address species-level differences (Question 2), we measured sediment carbon stocks for the seagrass species present at four of these locations. The species sampled included *Posidonia australis*, *Heterozostera nigricaulis*, and, where present, *Zostera muelleri*. *Amphibolis antarctica* was not sampled, as it typically grows on rocky substrates that do not support sediment carbon accumulation.

3.1 Methods

To sample belowground carbon for each species/site, a 5 cm diameter PVC pipe was used to collect four cores down to 50 cm or to the boundary of the sediment and underlying rock from 10 sites around Tayaritja (Figure 11b). Compaction of the samples during sampling was measured and accounted for. To sample the downcore variation in belowground carbon, on the day of collection while in the field, cores were subsampled at 5 depths down to 50 cm; 0–5 cm, 5–10 cm, 10–20 cm, 24–26 cm (representative of 20-30 cm), and 39-41cm (representative 30-50 cm). Samples were then placed in sterile plastic containers, stored in a 3°C fridge and within two weeks were dried at 60°C until a consistent weight was achieved. Dry bulk density (DBD) was calculated as dry weight divided by the original, compaction-corrected sample volume.

At the sites where we wanted to compare carbon stocks between species, the contribution of the seagrass roots/rhizomes and the soil carbon was determined by sieving the samples using a 500 µm sieve. Samples above 500 µm were assumed to be root/rhizomes while samples below 500 µm were assumed to be soil. Each dry sample was homogenised using an agate mortar and pestle. The total carbon (inorganic plus organic) content of all samples was determined on a subsample of 0.5–1 g. A LECO TruMac CN analyser was used for all analyses. For non-calcareous soils, organic carbon content was equated to the total carbon content. However, most samples contained inorganic carbon, and thus the analysis was completed after applying 4% HCl to remove inorganic carbon.

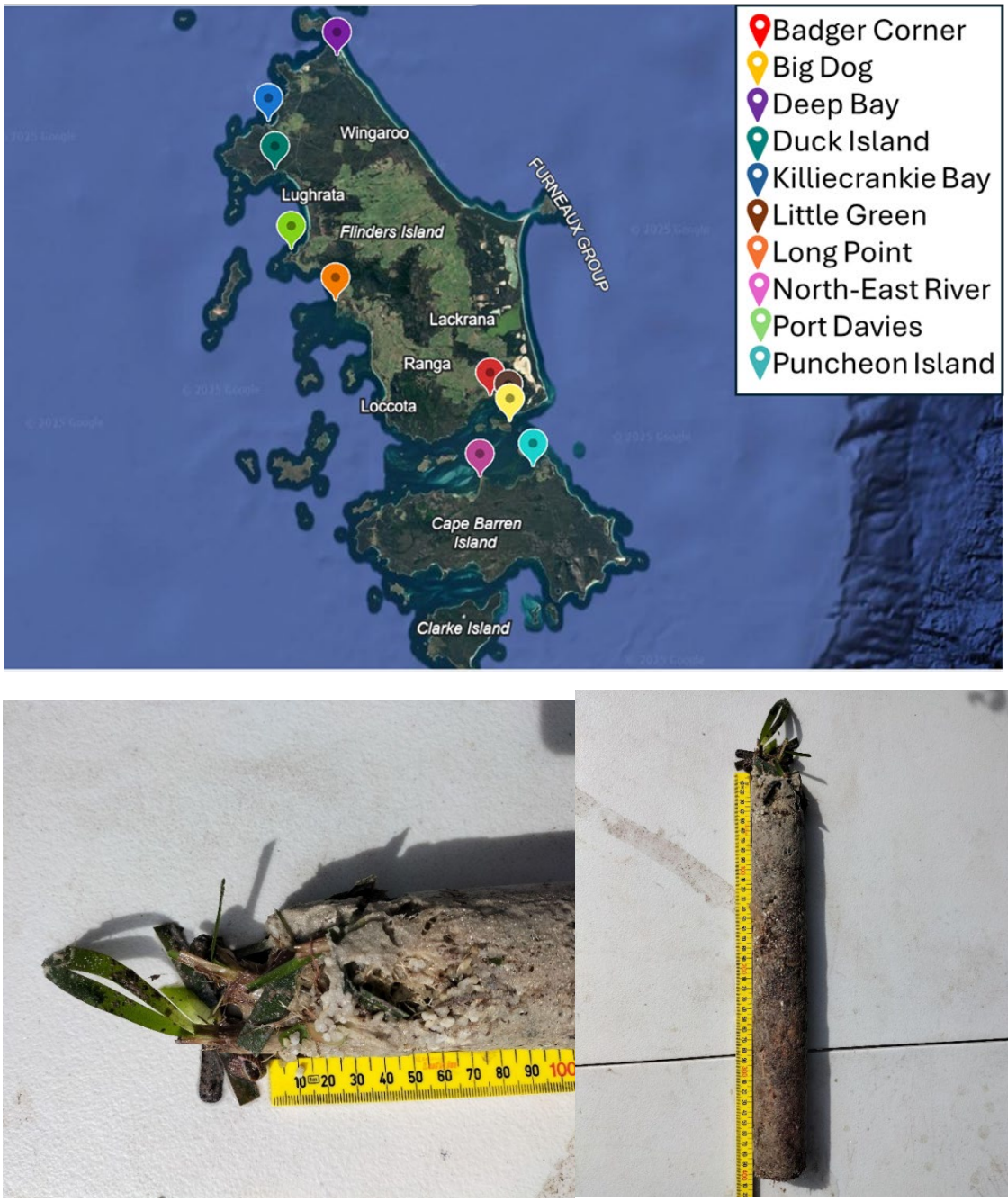


Figure 11. a) Map of Tayaritja Sea Country showing 10 sampling sites on the Islands. b) Example images of *Posidonia* sediment cores.

3.2 Results

To investigate the spatial variability of belowground carbon stocks, the carbon stock of *Posidonia* was determined across 9 sites, while at North East River, *Heterozostera* was sampled (Figure 12). The carbon stock of *Posidonia* was highest at Killiecrankie Bay (324 ± 132 t OC ha^{-1}) and Badger Corner (279 ± 67 t OC ha^{-1}), followed by Port Davies (165 ± 29 t OC ha^{-1}) compared to the other six sites.

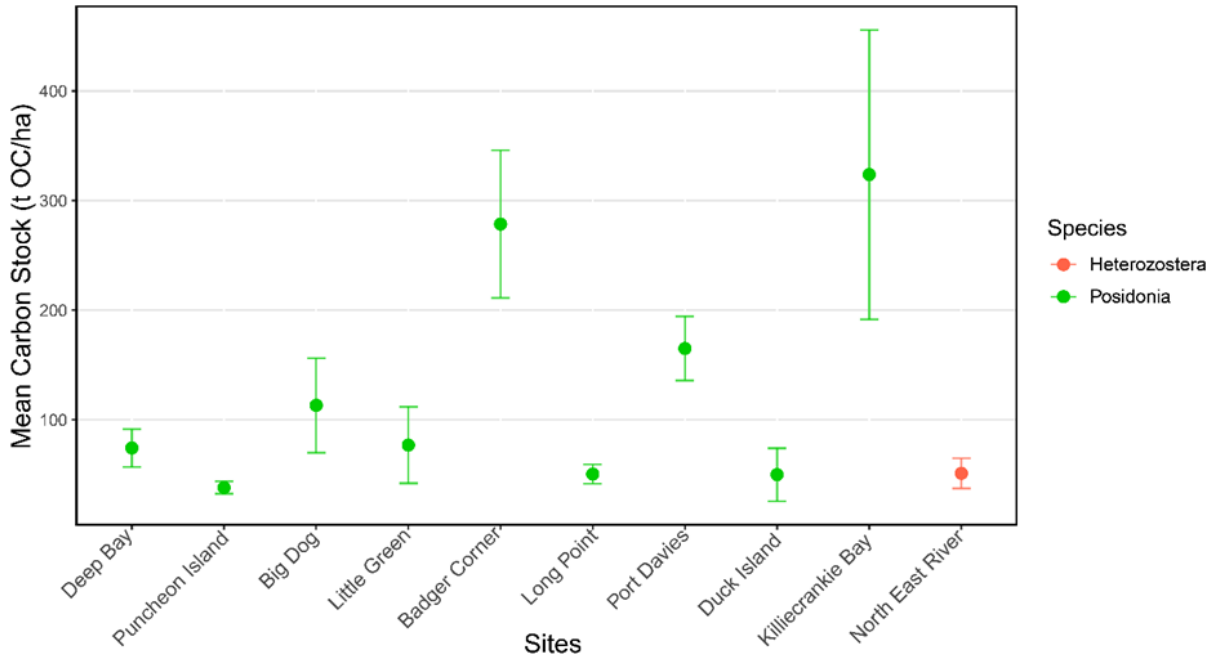


Figure 12. Spatial variability in carbon stocks at the 10 sites in Tayaritja.

To investigate the potential differences between seagrass species, at four of the 10 sites (Badger Corner, Big Dog, Deep Bay, Killecrankie Bay), both *Posidonia* and *Heterozostera* were sampled, with *Zostera* only found at two of these sites (Badger Corner and Big Dog). The mean carbon stock of *Heterozostera* (187 ± 174 t OC ha^{-1}) was slightly higher that of *Posidonia* (130 ± 111 t OC ha^{-1}) but was not significantly different (Figure 13). In comparison, across the two sites it could be sampled at, the mean carbon stock of *Zostera* (76 ± 47 t OC ha^{-1}) was the lowest (Figure 13). However, these results were influenced by an interaction between site and species (Figure 14), with no differences between seagrass species at Big Dog Island, but significant differences between all three species at Badger Corner, with the highest carbon stocks in *Posidonia* (278 ± 67 t OC ha^{-1}), followed by *Heterozostera* (137 ± 15 t OC ha^{-1}) and *Zostera* (36 ± 8 t OC ha^{-1}). The significantly higher carbon stocks of both *Posidonia* and *Heterozostera* at Killiecrankie compared to the other three sites, were also influential in driving some of the overall species' patterns.

Interestingly, when we look at the split between the contribution to the belowground carbon stocks from seagrass biomass, separately to the sediment, we see that the sites with substantially higher carbon stocks (Badger Corner and Killiecrankie Bay), are those with significant seagrass biomass in the belowground samples (Figure 15, Figure 16). This indicates that these sites have been stable seagrass beds for a long period of time, which has allowed this significant biomass to remain stored in the sediment.

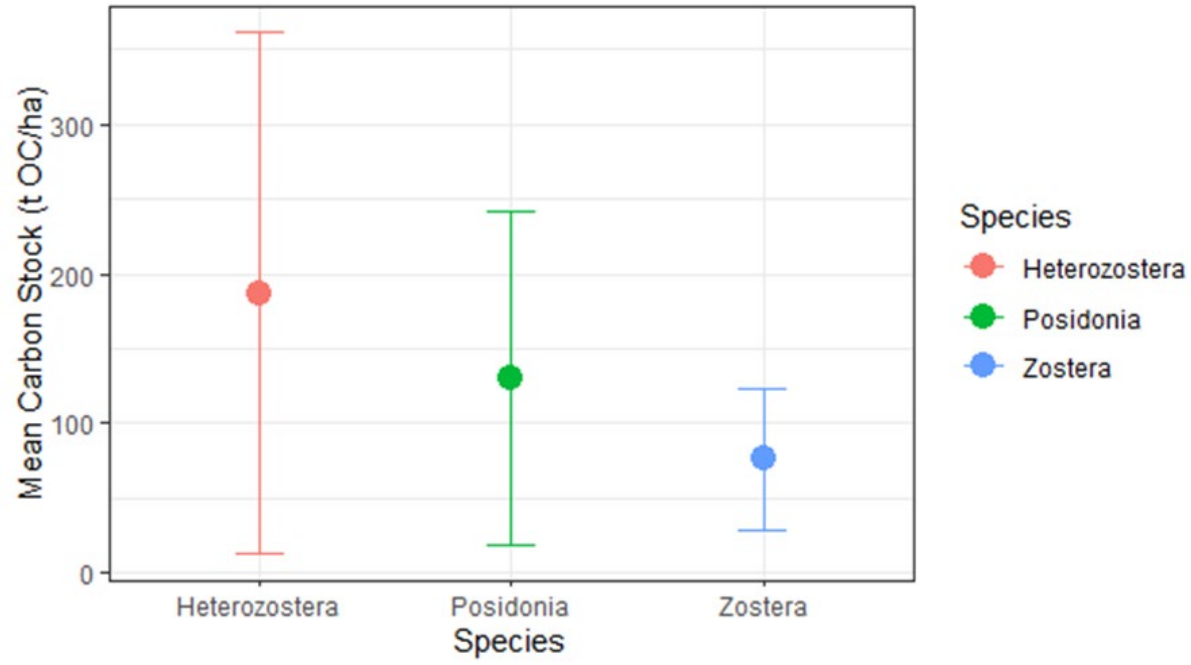


Figure 13. Mean belowground carbon stock (tonnes OC ha-1) in the top 50 cm of three seagrass species sampled from Tayaritja Sea Country. Error bars represent the standard deviation, *Heterozostera* (n=20), Posidonia (n=36), Zostera (n=8).

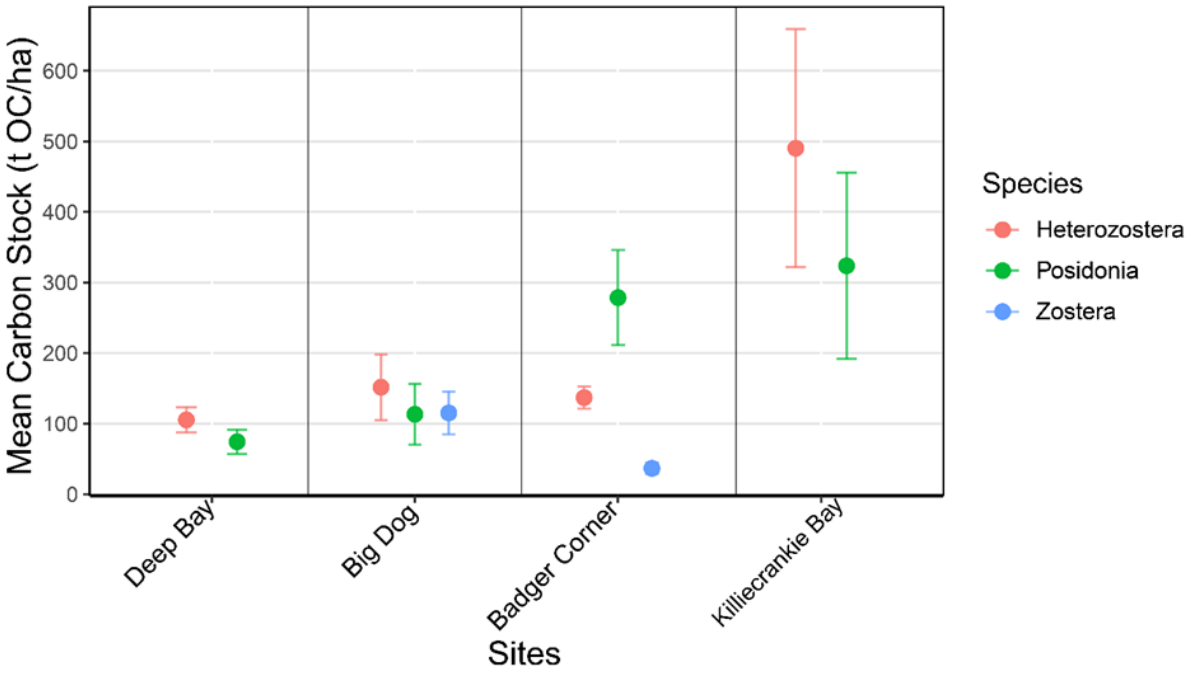


Figure 14. Mean belowground organic carbon stocks (tonnes OC ha-1) in the top 50 cm of sediment in each of the four locations sampled, where the three seagrass species were present. Error bars represent one standard error form the mean (n=4). Locations are listed South to North.

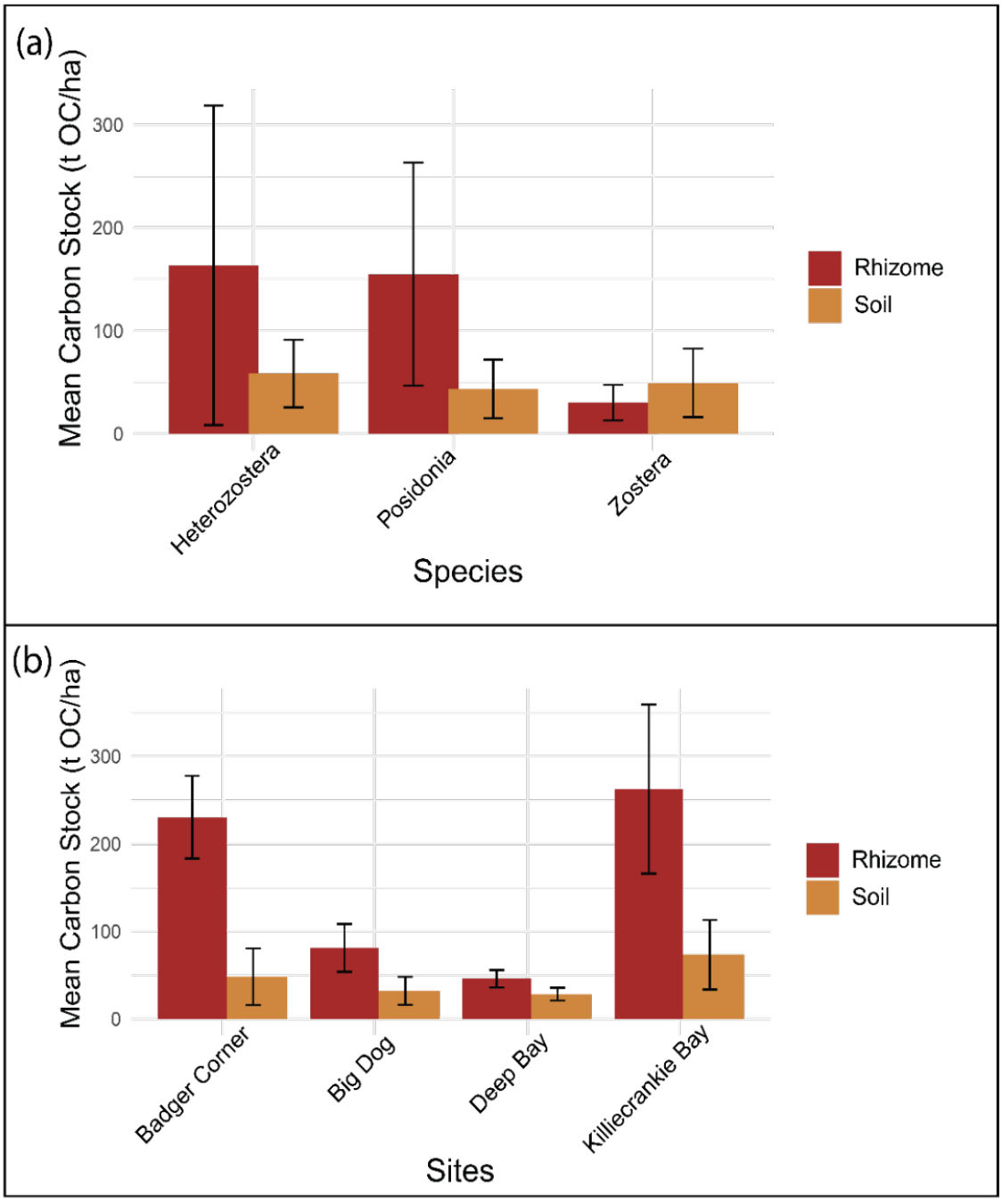


Figure 15. a) Carbon stock associated with each seagrass species between biomass (roots and rhizomes) and sediment, b) Carbon stock of *Posidonia* split between the seagrass biomass (roots and rhizomes) and sediment at Deep Bay, Big Dog, Badger Corner and Killiecrankie Bay.

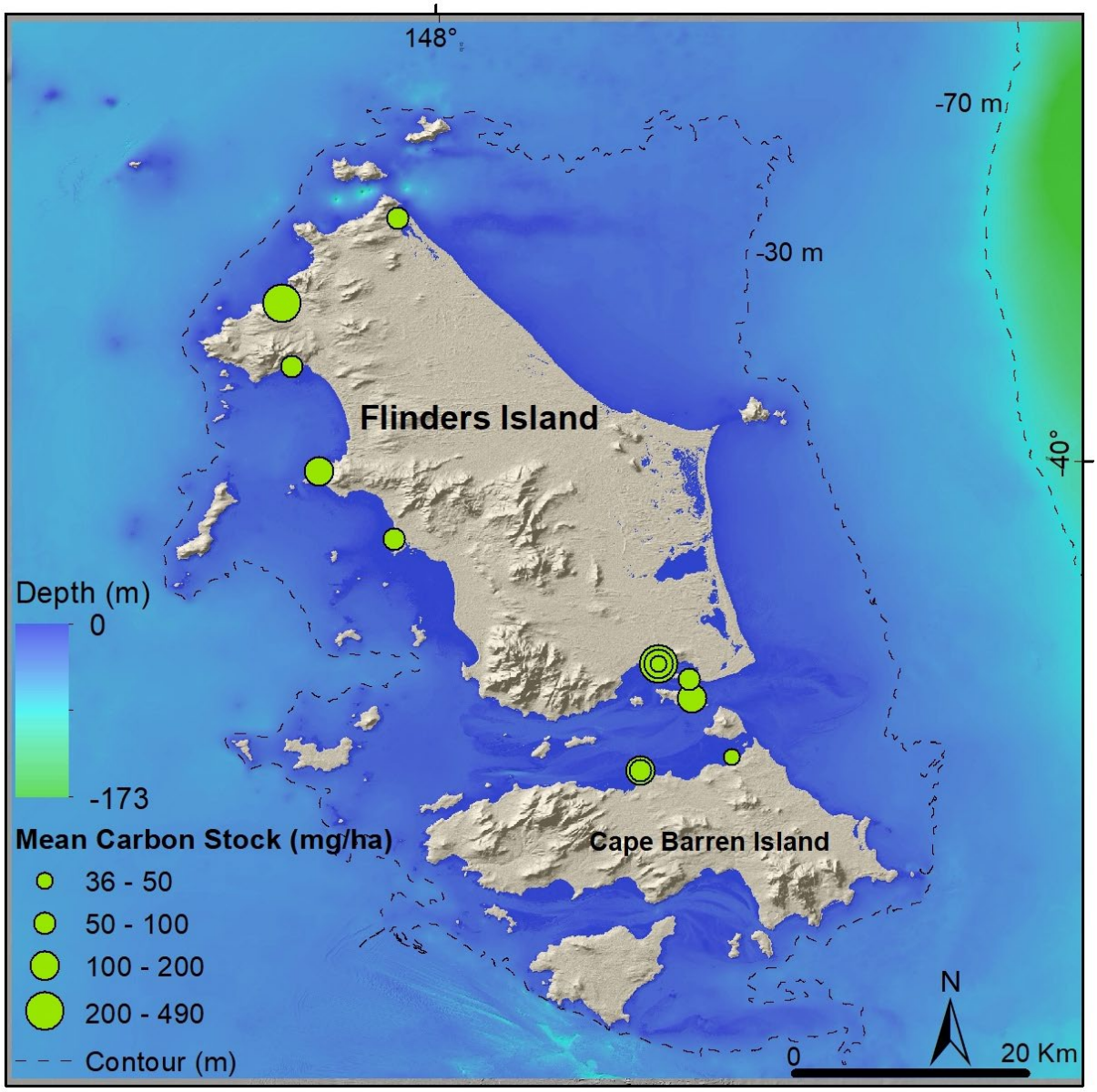


Figure 16. Belowground carbon stocks to 50 cm in the seagrass beds around Tayaritja.

3.3 Carbon stocks in Tayaritja

Based on the new seagrass maps produced for Tayaritja Sea Country as part of this project (see section 6 below), we can use this to estimate the total seagrass carbon stocks for Tayaritja Sea Country. Given the substantial area of *Posidonia*, we estimate the belowground seagrass carbon stocks to be 9.66 ± 8.19 million t OC.

Table 5. Mean belowground organic carbon stock (to 50 cm depth) and total carbon stock estimates for *Posidonia* and *Zostera* seagrass habitats. Mean carbon stock values (t OC ha^{-1}) are presented with standard deviations, and total carbon stock is expressed in millions of tonnes (t OC) based on mapped area estimates.

Species	Mean belowground carbon stock to 50 cm (t OC ha-1)	Area (ha)	Total 50 cm carbon stock (t OC)
Posidonia	130 ± 111	72587	9.43 ± 8.05 million
Zostera	76 ± 47	2990	0.23 ± 0.14 million
Total			9.66 ± 8.19 million

3.4 Summary

Sampling of the seagrass beds around Tayaritja Sea Country showed high sediment carbon stocks, but significant spatial variation. While there were no differences between species at three of the four sites, at Badger Corner, *Posidonia* had the highest carbon stocks, followed by *Heterozostera* and then Zostera. This information, combined with estimates of seagrass area in Tayaritja Sea Country, estimates a substantial, and previously unknown carbon stock.

Based on the data collected, along with the estimates of seagrass area of Tayaritja Sea Country, the estimated seagrass carbon stocks are 9.66 million t OC. This is substantially higher than the estimated seagrass carbon stocks of Victoria (1.05 million t OC; Ewers et al. 2018), but less than the 274 million t OC estimated to be stored in the seagrass of the Great Barrier Reef (York et al., 2018, corrected 2020).

However, the significance of this carbon stock is also a potential liability, as any loss of seagrass can lead to emissions of this already stored carbon. There are now a number of examples demonstrating how seagrass loss via marine heatwaves (Arias-Ortiz et al. 2018), physical disturbance from boat moorings (Serrano et al. 2016) or herbivore overgrazing (Carnell et al. 2020), can lead to carbon emissions from this already stored carbon in seagrass. To gain an understand of the size of this carbon stock, if we imagine a scenario where half of the *Posidonia* seagrass was lost from Tayaritja Sea Country, and assuming 50% of this carbon stock was remineralised into CO_2, this would equate to 8.65 million tonnes of CO_2eq. being released into the atmosphere. This is significant in comparison to the 14.4 million t CO_2 estimated to be released annually from land use change in Australia (Haverd et al., 2013). Thus, understanding the potential threats to the seagrass beds of Tayaritja Sea Country, is important to safeguard these nationally significant carbon stocks.

The data showed interesting patterns in spatial variation, which also resulted in differences between carbon stocks for species in some sites (Badger Corner), but not others. Of particular interest was the high carbon stocks in the *Heterozostera* cores at Killiecrankie Bay. In reviewing the location where these cores were taken, *Heterozostera* only tended to occur on the fringes of *Posdonia* beds in Killiecrankie Bay (Figure 17). It is likely that the sites where *Heterozostera* was sampled in Killiecrankie actually represent the significant carbon stocks in old rhizome material from *Posidonia,* rather than carbon stored by *Heterozostera* itself.

However, given that seagrass loss can lead to remineralisation of already stored carbon discussed above, it may be that *Heterozostera* helps to stablise these carbon stocks in areas where *Posidonia* may have been disturbed or lost.

There are a number of caveats with the estimates of seagrass belowground carbon stocks across Tayaritja Sea Country. Firstly, all the sites sampled were within the intertidal and shallow sub-tidal zones, and a number of studies have showed that seagrass sediment carbon stocks vary with depth (Lavery et al., 2013). Secondly, while we did our best to sample across the geographic distribution of seagrass in Tayaritja Sea Country, we were unable to obtain samples from the southern side of Truwana/Cape Barren and Lungutlanna/Clarke Island, which represent a significant portion of the seagrass area in the region. Future efforts would aim to fill these spatial gaps, and to collect more cores on carbon accumulation rates.

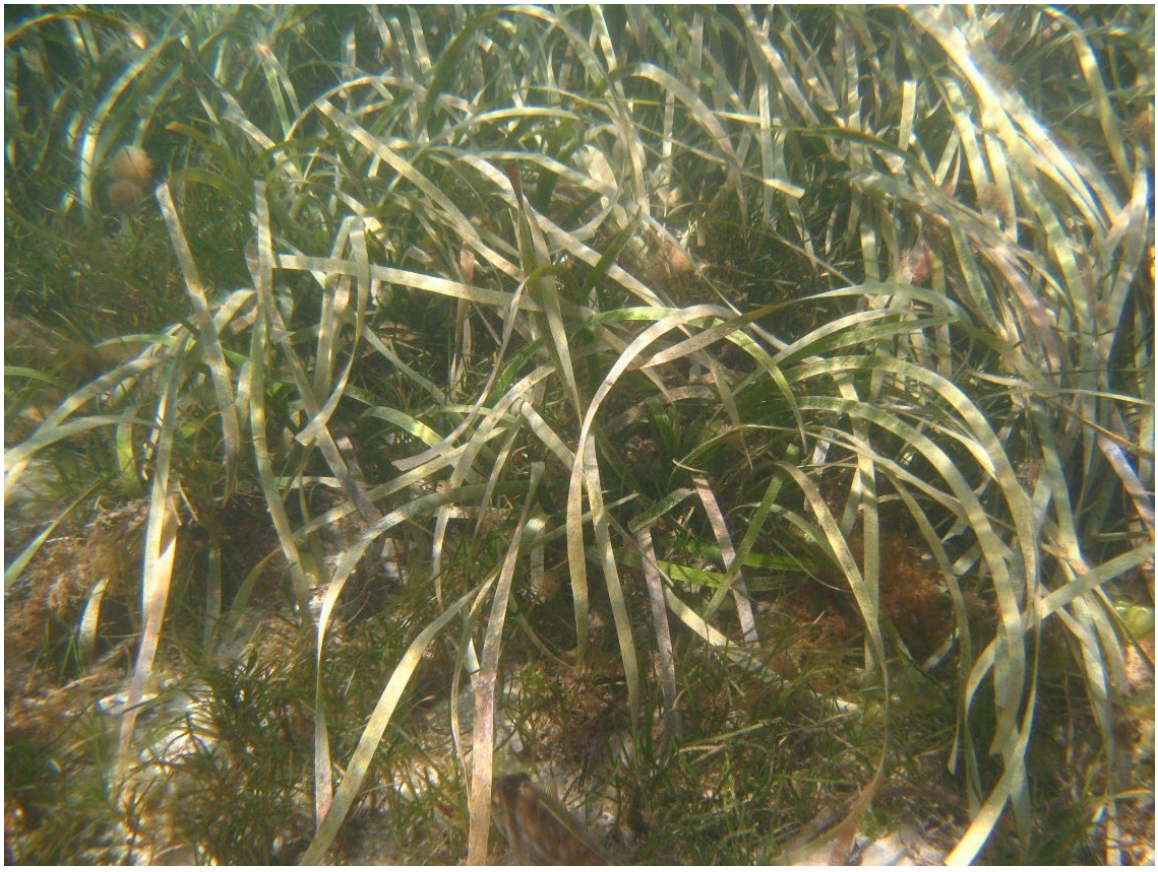


Figure 17. Picture showing *Heterozostera* growing alongside *Posidonia* at Killiecrankie Bay

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4. Genetic connectivity of seagrass in the Tayaritja Sea Country

Seagrass are critical marine ecosystems that provide a range of important ecosystem services. These include providing key habitat to a range of species, including important cultural, recreational and commercial fisheries species. They also play an important role in stabilising sediments and preventing coastal erosion, recycling nutrients and are important carbon sinks. Given this important role in providing habitat and a range of ecosystem services, there has been increasing concern about the rapid decline's seagrass populations are now experiencing globally (Waycott et al., 2009). While a range of natural and anthropogenic factors have been identified as the potential causes of these declines, there is increasing evidence that the ability of seagrass beds to mitigate and cope with increasing levels of disturbance is to a large extent influenced by the levels of genetic variation and its spatial distribution within and among populations (Hughes & Stachowicz, 2004; Reusch et al., 2005, Ehlers et al., 2008). Thus, the resilience and long-term viability of seagrass ecosystems require an understanding of the levels of genetic variation and patterns of connectivity among populations

4.1 Methods

4.1.1 Sample collections

Leaf samples were collected for genetic analysis from beds of four seagrass species (*Amphibolis antarctica, Posidonia australis, Heterozostera nigricaulis,* and *Zostera muelleri*). Samples were collected from between 4-8 sites from around Tayaritja Sea Country for each species, with 15-25 replicate leaf samples collected from each site (Figure 18). Samples were collected from areas approximate 20 x 20 m with minimum distance between samples of 2 m (to avoid collecting samples from the same ramet). Samples were prioritised for DNA extraction and SNP genotyping with a total of 466 samples selected for genotyping (Table 6).

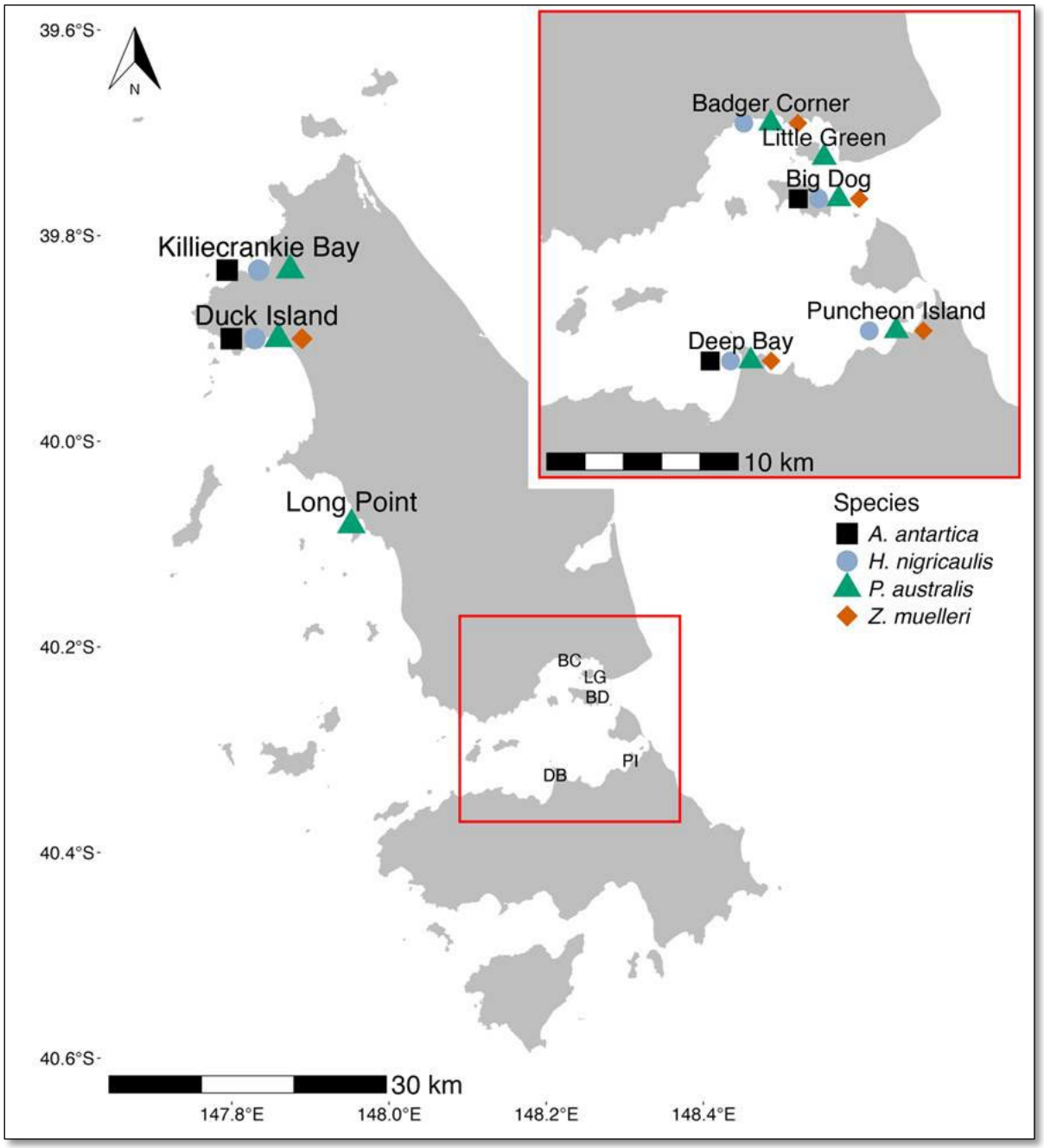


Figure 18. Map of Tayaritja Sea Country with population sites for each species (*Amphibolis antartica:* Black, square; *Heterozostera nigricaulis*: Blue, circle; *Posidonia australis*: Green, triangle; *Zostera muelleri*: orange, diamond).

4.1.2 DNA extractions, sequencing and SNP calling

DNA was extracted from leaf tissue by Diversity Arrays Technologies (DArT Pty Ltd, Canberra, Australia) following the manufactures instructions for the QIAGEN DNeasy plant tissue kits. Sequencing for SNP genotyping was performed using the genome-wide and medium-density DArTseq™ platform. This technology uses a combination of genome complexity reduction

methods, including a DNA digestion and ligation step using proprietary restriction enzymes, followed by PCR and quantification (Kilian et al., 2012; Sansaloni et al., 2010). Samples were standardised and sequenced on an Illumina HiSeq2500. Sequences were filtered and assembled using the DArTseq™ analytical pipeline which uses technical replicates to calculate genotyping reproducibility to filter paralogous regions and identify sequencing errors (Kilian et al., 2012; Sansaloni et al., 2010).

Table 6. Total number of populations and individuals sent for SNP sequencing, and the total number of SNPs called through the DArT pipeline for each species.

Species	Number of Populations	Total samples	Total SNPs Called (%Missing Data)
Amphibolis antartica	4	94	9079 (29.8%)
Heterozostera nigricaulis	6	94	7666 (36.0%)
Posidonia australis	8	184	3692 (38.0%)
Zostera muelleri	5	94	5621 (15.2%)

4.1.3 Population genetic structure analyses

All analysis was conducted with the statistical program 'R' (R Core Team, 2017). Sequences were received from DArT before being filtered by read depth for each species of between 5 and 75, individual call rate per species of > 55%, loci call rate per species of > 60%, minor allele frequencies per species of > 0.02 and hamming distance per species of > 0.2, using the package 'DArTR' (Mijangos et al., 2022). As many seagrass species can reproduce both sexually and asexually, we assessed the importance of sexual/asexual reproduction in maintaining populations by determining the number of unique multilocus genotypes (MGLs) for each species within each site. MLGs were determined by assessing the genetic distance between individuals within a species through a Euclidian distance matrix, with a threshold set to ensure the maximum number of MLGs were identified, while transforming the data to a normal distribution (MLG genetic distance thresholds: *A. antartica* – 22; *H. nigricaulis* – 11; *P. australis* – 16; *Z. muelleri* – 20) using the package 'poppr' (Kamvar et al.. 2014). Clonal richness (r) was calculated as r = (MLG-1)/(Ind-1), where values close to 0 indicate high levels of clonality and values close to 1 indicate all samples represent unique genotypes. Once clonal genotypes were identified, a new data set was established for each species containing only unique MGLs. This is because many population genetic analyses assume sexual reproduction and random mating, and high levels of clonality as cause erroneous estimates of population genetic parameters if not removed.

All species prior to clone removal had a bimodal distribution indicating clones and unique MLGs, with the exception of *P. australis*, that was trimodal. This trimodal patterns indicates there may have been 'near-clones' within the data set and they were also removed from downstream analyses. Once all MGLs were identified, data sets for each species were

generated which contained only unique MLG at each site, with replicate clonal genotypes removed. For clonal genotypes, a random representative for each MLG was chosen to remain within each data set (SUPP Table 2). Following this, the 'gl.filter.hwe' command within 'DartR' was used to remove any loci that did not conform to Hardy-Wineburg equilibrium (HWE). The above steps were optimised to maintain the maximum number of SNPs while aiming for < 20% missing data to remove noise and avoid erroneous estimates of population genetic parameters (Table 7).

Table 7. Final number of populations, MLGs and SNPs called post filtering for each species.

Species	Number of Populations	Total MLGs	SNPs post filtering (%Missing Data)
Amphibolis antartica	4	57	3652 (6.48%)
Heterozostera nigricaulis	6	66	1348 (18.32%)
Posidonia australis	8	49	1145 (18.07%)
Zostera muelleri	5	48	1954 (9.82%)

Global F-statistics (HO, HS, FST and FIS) were calculated using the 'basic.stats' function, while site level estimates of HO, HE, % polymorphic loci, FIS, and pairwise FST, were calculated using the 'gl.report.hetrozygosity' and 'gl.fst.pop' function. Only sites with >1 MLG were retained for pairwise FST estimates. For each species a Discriminant Analysis of Principal Components (DAPC) was used to assess the number of genetic clusters within the data set for each species using the 'find.clusters' function within the 'adegenet' package (Jombart & Ahmed, 2011) before calculating DAPC with the 'dapc' function and visualised as a DAPC and genotype composition plot. A Bayesian STRUCTURE analysis was also used to assess the potential number of genetic clusters for each species using 'faststructure' (Raj et al., 2014) and visualised with 'gl.plot.faststructure'. Finally, to assess if there was a relationships between the geographical distance (by water) between sites, and levels of genetic differentiation, an isolation by distance analysis was carried out by comparing the correlation between pairwise matrices for geographic distance (log(n) of distance between points) and genetic distance (FST/1-FST), with a mantel test used to account for non-independence of data points within the matrices.

4.2 Results

4.2.1 Patterns of clonality and levels of genetic diversity

Across all sites and species, Killiecrankie Bay had the highest levels of clonality with an r of 0.15, 0.46 and 0 for *A. antartica, H. nigricaulis* and *P. australis* respectively with no *Z. muelleri* found at the site (Table 3). This was followed closely by Big Dog with 0.62, 0.75, 0 and 0.21 for *A. antartica*, *H. nigricaulis*, *P. australis* and *Z. muelleri* respectively. Of the four species of seagrass *P. australis* showed the highest levels of clonality with all samples from two sites (Big

Dog and Killiecrankie Bay) consisting of one clonal genotype and two populations (Badger Corner and Little Green) both having very low clonal richness of 0.14 and 0.21 (Table 8). While clonal richness was low at Badger Corner for P. australis it was significantly higher for both H. *nigricaulis* (r=1) and Z. muelleri (r=0.94). This trend is also reflected in the percentage of polymorphic loci, with samples from Killiecrankie Bay ranging from 18-52% polymorphic loci for the three species present, and Big Dog ranging from 7-88% polymorphic loci (Table 8). While *Z. muelleri* had an overall lower average number of polymorphic loci (44.9%), this was closely followed by P. australis (54.0%) with *P. australis* from Big Dog having only 76 polymorphic loci out of a total of 1112 (6.8%) (Supp. table 1).

For sites where there was more than one MLG, F_{IS} was positive at all sites for all four seagrass species, indicating low levels of heterozygosity within sites, with the exception of *H. nigricaulis* at Killiecrankie Bay, that was mildly negative (-0.13) indicating a slight heterozygous excess (Table 8). Most sites had a high (>0.2) FIS, with only four sites (A. *antartica* – Killiecrankie Bay; *H. nigricaulis* – Killiecrankie Bay; and *Z. muelleri* – Big Dog, Puncheon Island) having moderate to low FIS indicating a heterozygous deficit (Table 8). FIS was very high at two sites, with both *H. nigricaulis* at Duck Island, and *P. australis* at Puncheon Island having an FIS of >0.5 indicating very high levels of heterozygous deficits (Table 8). Similar to these trends, there was a lower-than-expected heterozygosity (HO) at most sites for the four species with the exception of one population for *H. nigricaulis* (Killiecrankie Bay), two populations for *P. australis* (Big dog and Killiecrankie bay), and one population for *Z. muelleri* (Duck Island) (Table 8). Care should be taken when interpreting these results as some populations are only represented by one or a small number of unique MLG (Table 8).

Table 8. Population level statistics for each species with numbers of individuals (Ind), multilocus genotypes (MLG), clonal richness (r), loci, polymorphic loci (PolyLoci), percentage of polymorphic loci (%PolyLoci), observed heterozygosity (HO) ± standard deviation (SD), expected heterozygosity (HE) ± SD and inbreeding coefficient (FIS) ± SD

Species	Population	Ind	MLG	*r*	Loci	PolyLoci	% PolyLocI	H_O (±SD)	H_E (±SD)	F_{IS} (±SD)
Amphibolis antartica	Big Dog	24	16	0.65	3652	3222	88.23	0.189 (±0.158)	0.253 (±0.165)	0.243 (±0.356)
	Deep Bay	21	16	0.75	3652	3317	90.83	0.174 (±0.143)	0.260 (±0.159)	0.310 (±0.361)
	Duck Island	22	21	0.95	3652	3335	91.32	0.201 (±0.156)	0.258 (±0.161)	0.216 (±0.33)
	Killiecrankie Bay	21	4	0.15	3640	1896	52.09	0.171 (±0.239)	0.190 (±0.199)	0.191 (±0.474)
Heterozostera nigricaulis	Badger Corner	13	13	1.00	1348	966	71.66	0.106 (±0.127)	0.199 (±0.169)	0.428 (±0.412)
	Big Dog	13	10	0.75	1348	694	51.48	0.078 (±0.126)	0.150 (±0.174)	0.422 (±0.441)
	Deep Bay	13	13	1.00	1347	972	72.16	0.108 (±0.127)	0.201 (±0.169)	0.419 (±0.413)
	Duck Island	10	10	1.00	1348	640	47.48	0.068 (±0.122)	0.154 (±0.185)	0.522 (±0.437)
	Killiecrankie Bay	12	6	0.46	1342	241	17.96	0.088 (±0.227)	0.068 (±0.157)	-0.134 (±0.436)
	Puncheon Island	14	14	1.00	1348	1052	78.04	0.129 (±0.135)	0.211 (±0.164)	0.356 (±0.395)
Posidonia australis	Badger Corner	22	4	0.14	1144	555	48.51	0.112 (±0.198)	0.189 (±0.207)	0.437 (±0.507)
	Big Dog	22	1	0.00	1112	76	6.83	0.039 (±0.193)	0.019 (±0.096)	0.000 (±0.000)
	Deep Bay	21	14	0.65	1145	970	84.72	0.173 (±0.152)	0.264 (±0.171)	0.337 (±0.373)
	Duck Island	14	10	0.69	1145	833	72.75	0.127 (±0.152)	0.251 (±0.186)	0.482 (±0.421)
	Killiecrankie Bay	8	1	0.00	912	315	34.54	0.089 (±0.286)	0.045 (±0.143)	0 .000 (±0.000)
	Little Green	15	4	0.21	1145	633	55.28	0.133 (±0.200)	0.213 (±0.205)	0.412 (±0.485)
	Long Point	3	3	1.00	1128	479	42.46	0.134 (±0.242)	0.168 (±0.208)	0.330 (±0.519)
	Puncheon Island	12	12	1.00	1145	877	76.59	0.106 (±0.121)	0.252 (±0.179)	0.550 (±0.387)
Zostera muelleri	Badger Corner	19	18	0.94	1954	1431	73.23	0.152 (±0.158)	0.205 (±0.175)	0.261 (±0.352)
	Big Dog	20	5	0.21	1949	770	39.51	0.151 (±0.229)	0.143 (±0.191)	0.047 (±0.381)
	Deep Bay	20	15	0.74	1954	1048	53.63	0.087 (±0.121)	0.135 (±0.161)	0.314 (±0.372)
	Duck Island	17	1	0.00	1772	273	15.41	0.051 (±0.221)	0.026 (±0.110)	0 .000(±0.000)
	Puncheon Island	17	9	0.50	1954	955	48.87	0.113 (±0.157)	0.135 (±0.168)	0.167 (±0.352)

4.2.2 Population genetic structure patterns and differentiation

Analysis of levels of genetic differentiation among populations for each species (global F_{ST}) indicates significant genetic structure is present for all species (Supp. table 1). However, for populations of *A. antartica* and *P. australis* global F_{ST} were moderate (A. *antartica* F_{ST} = 0.057 and P. australis F_{ST} = 0.079), suggesting low levels of connectivity and gene flow between at least some sites (Supp. table 1). In contrast, *H. nigricaulis* and *Z. muelleri* both show much higher levels of genetic differentiation, indicating much more restricted gene flow among sampled populations (H. *nigricaulis* F_{ST} = 0.205 and Z. muelleri FST = 0.209) (Supp. Table 1). Pairwise estimates of F_{ST} among all sites show that for *A. antartica* the highest levels genetic differentiation was between Killiecrankie Bay and all other sites (F_{ST} ranged from 0.100 to 0.126) (Supp. table 2). For *P. australis* Duck Island showed the greatest level of differentiation from other sites (F_{ST} ranging from 0.104 to 0.123) (Supp. table 2). For H. *nigricaulis*, Killiecrankie Bay was again the most genetically differentiated from other sites with F_{ST} ranging from 0.388 to 0.449 (Supp. table 2). All populations of *Z. muelleri* have relatively similar levels of genetic differentiation with pairwise FST ranging between 0.103 and 0.271. Across all four species Killiecrankie Bay and Duck Island tend to show the highest genetic differentiation indicating low gene flow between these two sites and other sampled populations.

4.2.3 Population discriminant and STRUCTURE analysis

Amphibolis antartica

The DAPC showed three distinct clusters, with samples from Deep Bay, Duck Island and Big Dog forming one cluster, samples from Big Dog formed a second cluster and samples for Killiecrankie Bay formed a separate third cluster (Figure 19). These genetic clusters were largely supported by the structure analysis, however, Killiecrankie Bay did show some overlap of genotypes with Duck Island. Deep Bay, Duck Island and Big Dog did share genotypes from the same cluster, suggesting some connectivity between these sites, however, Big Dog did support individuals that formed a distinct cluster of genotypes (blue) that was not shared with other sites (Figure 20).

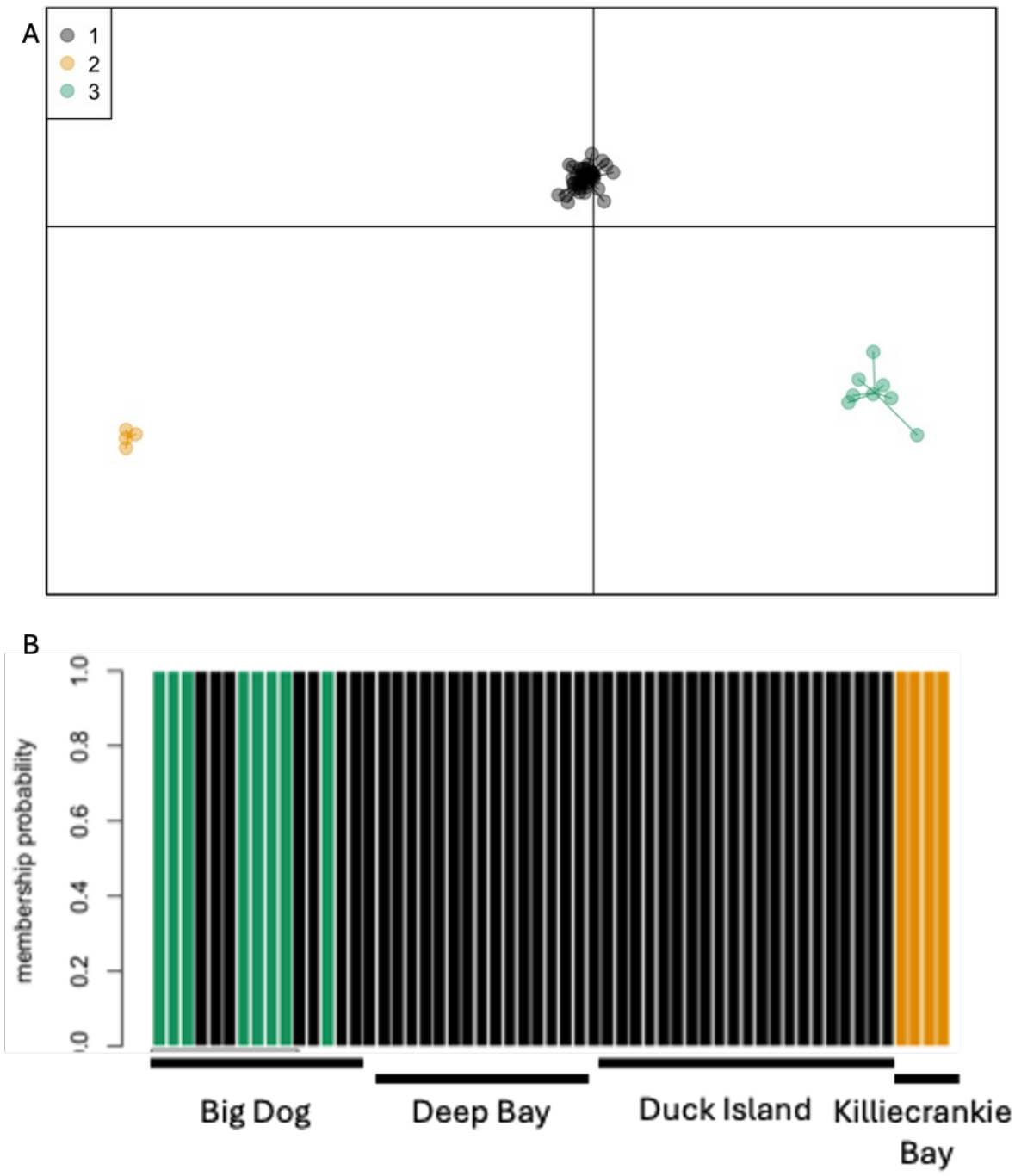


Figure 19. DAPC for *Amphibolis antarctica* showing a) scatter plot of clusters and b) membership of each cluster per sample.

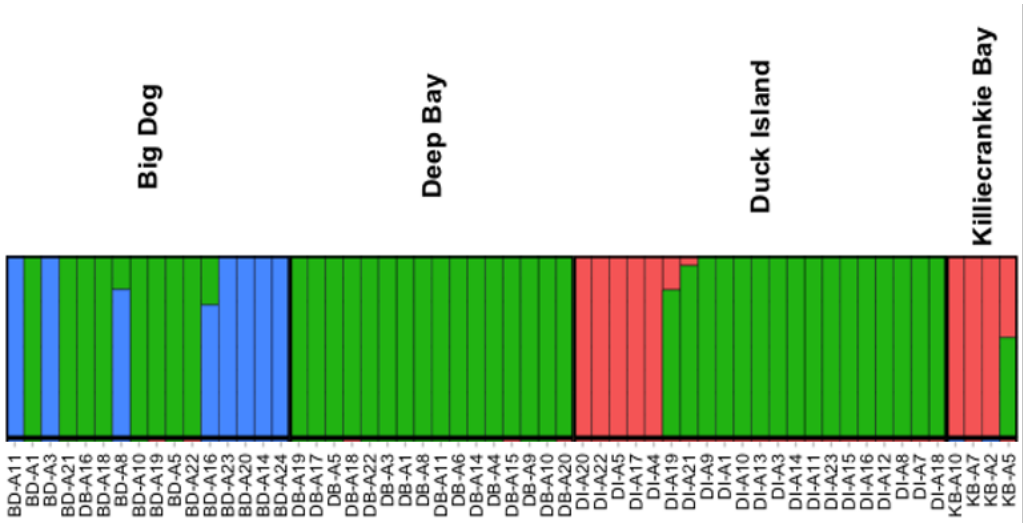


Figure 20. Structure plot of Amphibolis antarctica for K=3

Heterozsotera nigricaulis

The DAPC for *H. nigricaulis* showed three distinct clusters, with samples from Badger Corner, Big Dog, Deep Bay and Puncheon Island forming one dominant cluster, with Duck Island and Killiecrankie Bay both belonging to distinct clusters (Figure 21). This was supported by the STRUCTURE analysis, Duck Island consisting largely of a genotype not commonly associated with the other populations (blue), and Killiecrankie Bay consisted of a second unique set of genotypes (green) with limited admixture with the remaining populations (Figure 22).

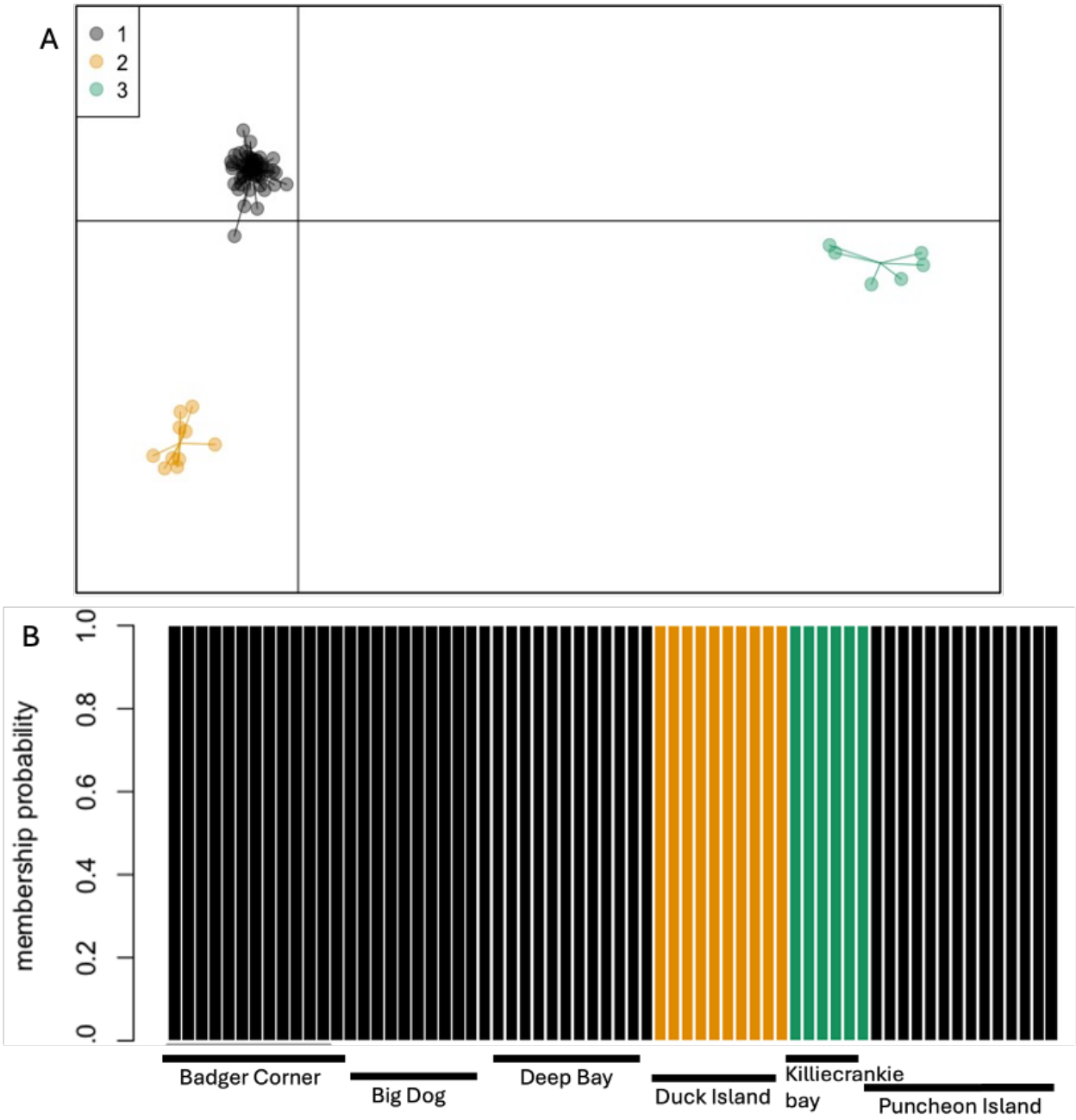


Figure 21. DAPC for *Heterozostera nigricaulis* showing a) scatter plots of clusters and b) membership of each cluster per sample.

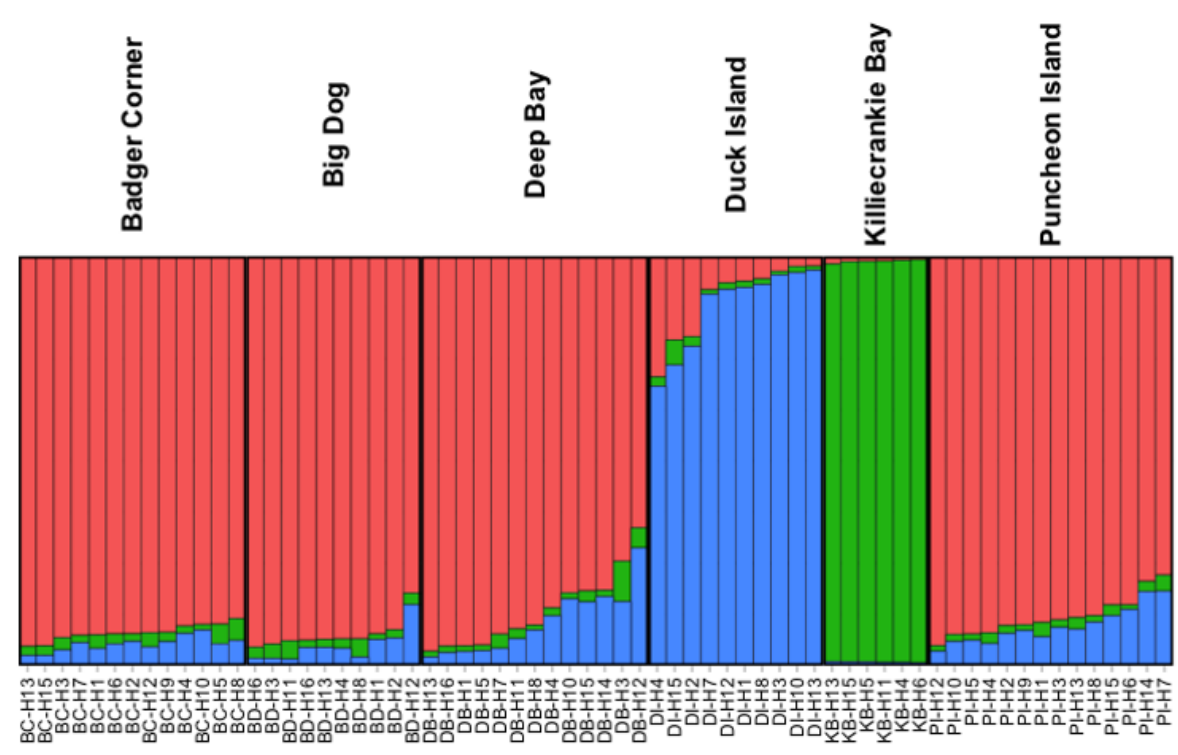


Figure 22. Structure plot of *Heterozostera nigricaulis* for K=3

Posidonia australis

The DAPC for P. *australis* showed two distinct clusters, with Duck Island forming a separate cluster to the remaining populations (Figure 23). This was largely supported by the STRUCTURE analysis with Duck Island primarily composed of a cluster of distinct genotypes (blue), with limited mixing with the other cluster of genotypes (red) (Figure 24). There is also slight mixing of the Duck Island genotype found at Little Green and Long Point.

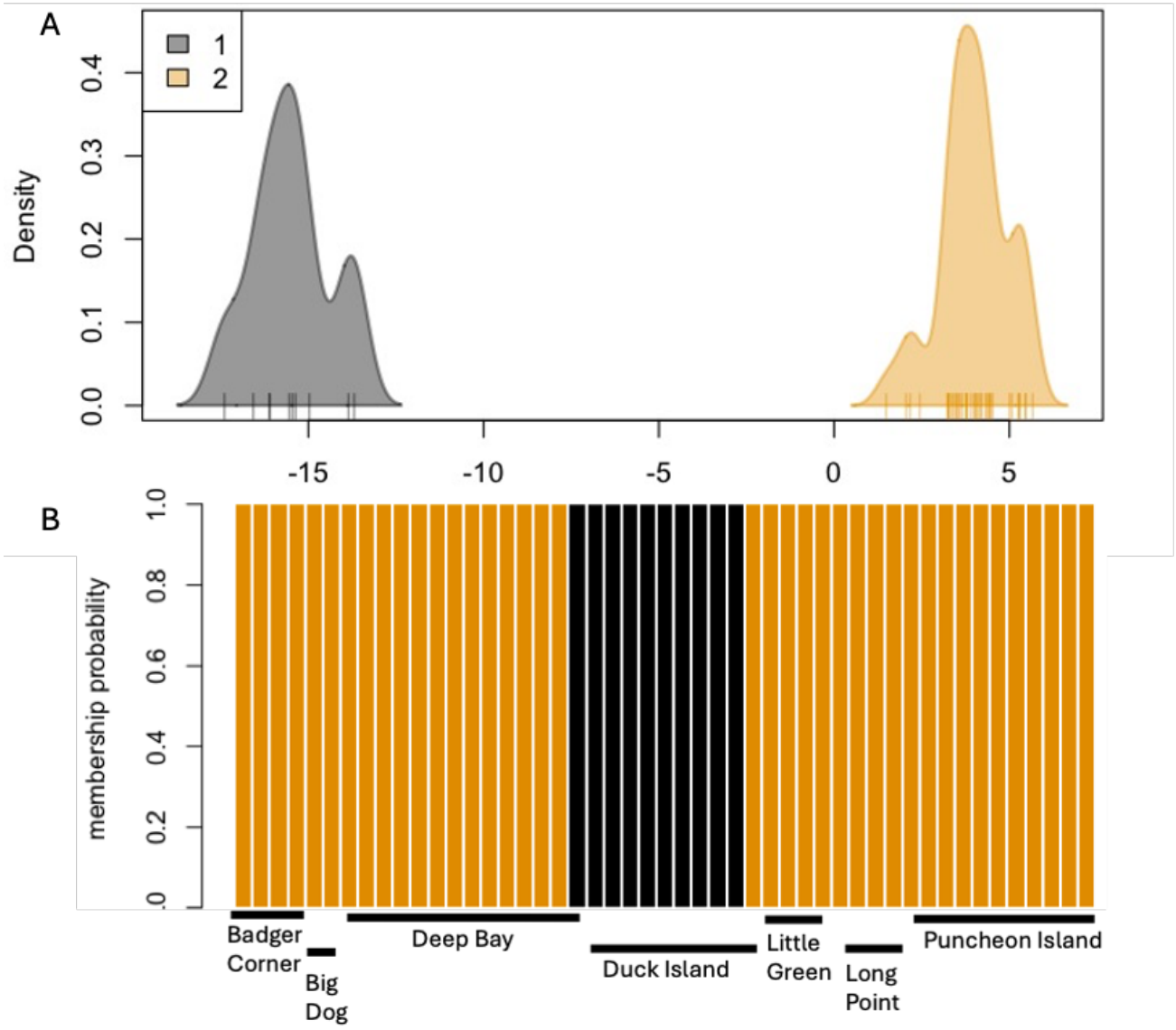


Figure 23. DAPC for *Posidonia australis* showing a) distribution of clusters and b) membership of each cluster per sample.

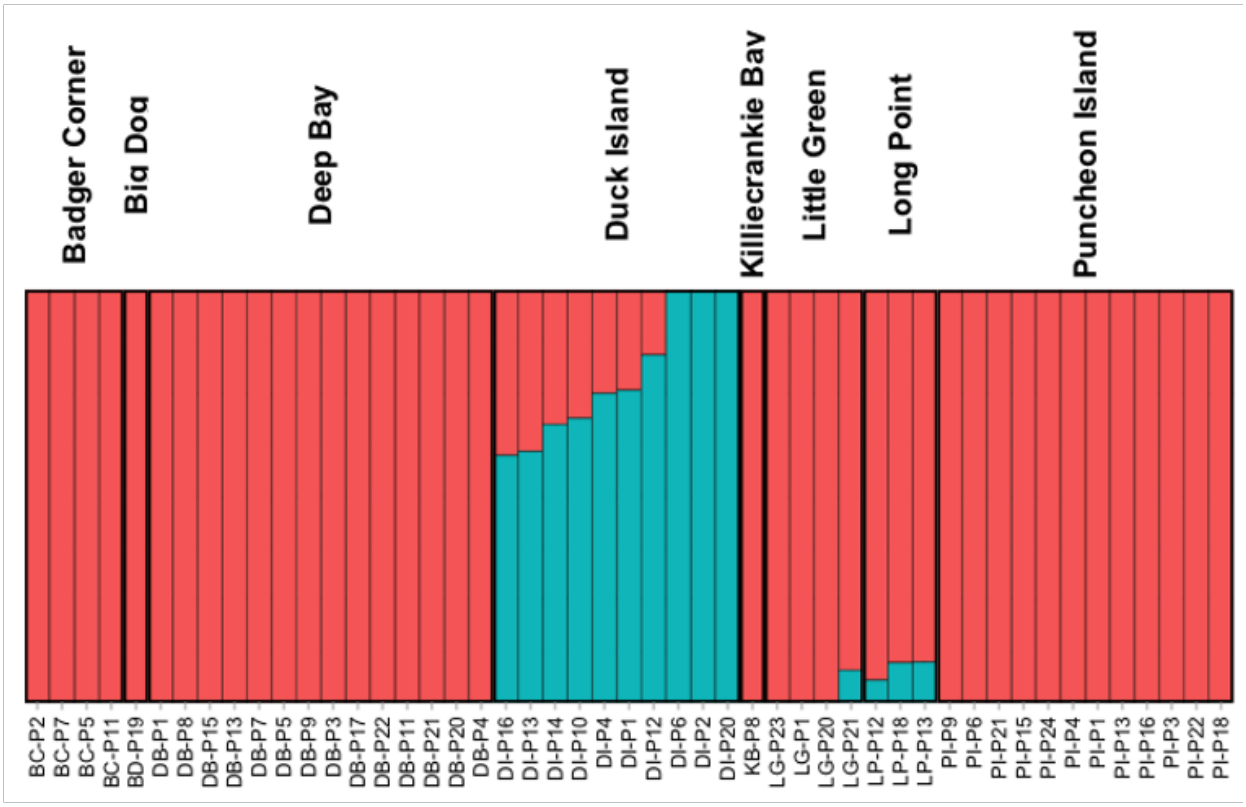


Figure 24. Structure plot of *Posidonia australis* for K=2

Zostera mulleri

The DAPC for Z. muelleri showed three distinct clusters, with samples from Deep Bay. Duck Island and Puncheon Island forming a distinct cluster, Big Dog and a number of samples from Badger Corner formed a second cluster and the remain samples from Badger Corner formed a third cluster (Figure 25). This was supported by the STRUCTURE analysis, with Badger Corner and Big Dog showing some admixture not found at the three other sites, and one genotype (green) found almost exclusively at Badger Corner (Figure 26).

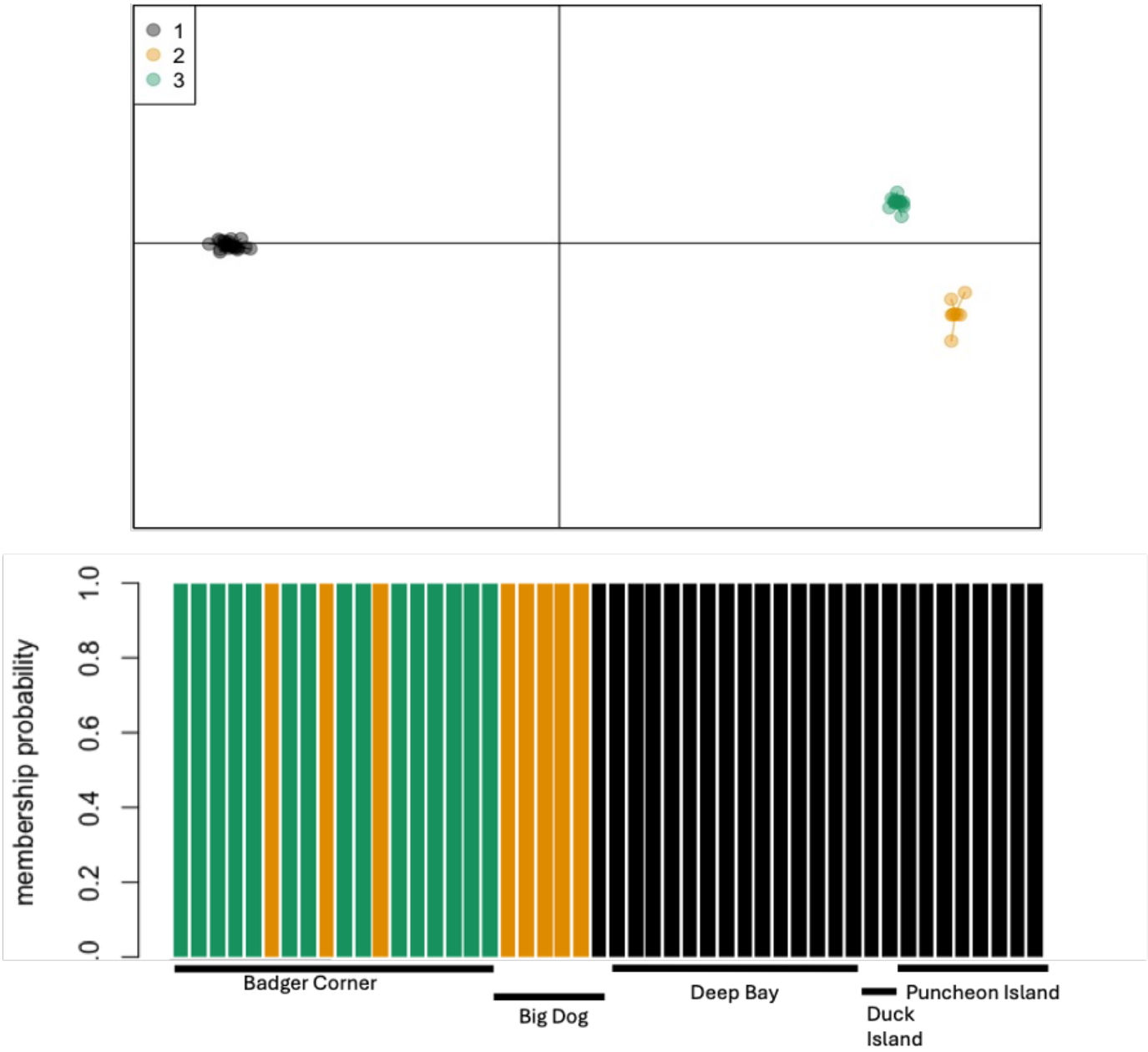


Figure 25. DAPC for *Zostera muelleri* showing a) scatter plot of clusters and b) membership of each cluster per sample.

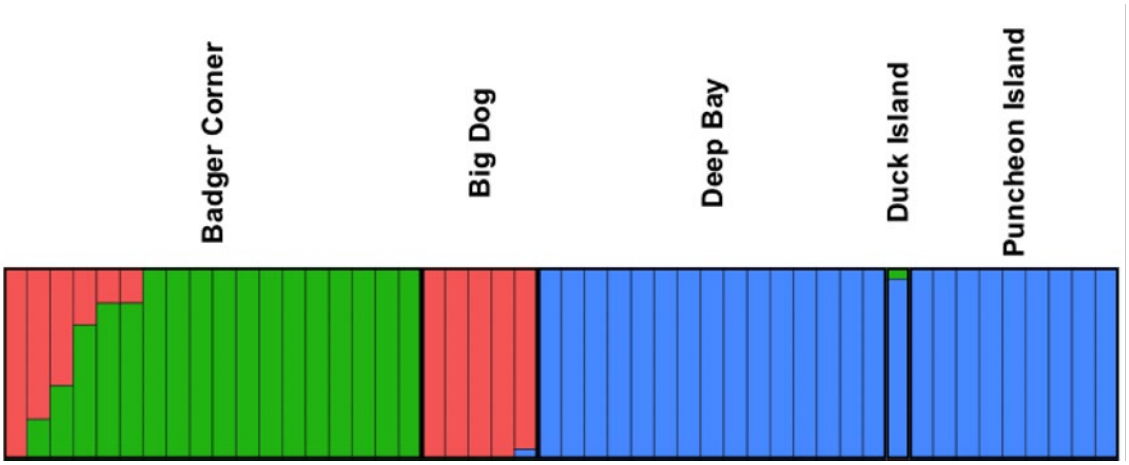


Figure 26. Structure plot of *Zostera muelleri* for K=3

4.2.4 Isolation by distance

The mantel test for all four species indicated there was no significant relationships between genetic differentiation and geographical distance (*A. antarctica*: p=-0.38 ; *H. nigricaulis*: p=0.42 , *P. australis*: p=0.36 , *Z. muelleri*: p=0.92 (Figure 27).

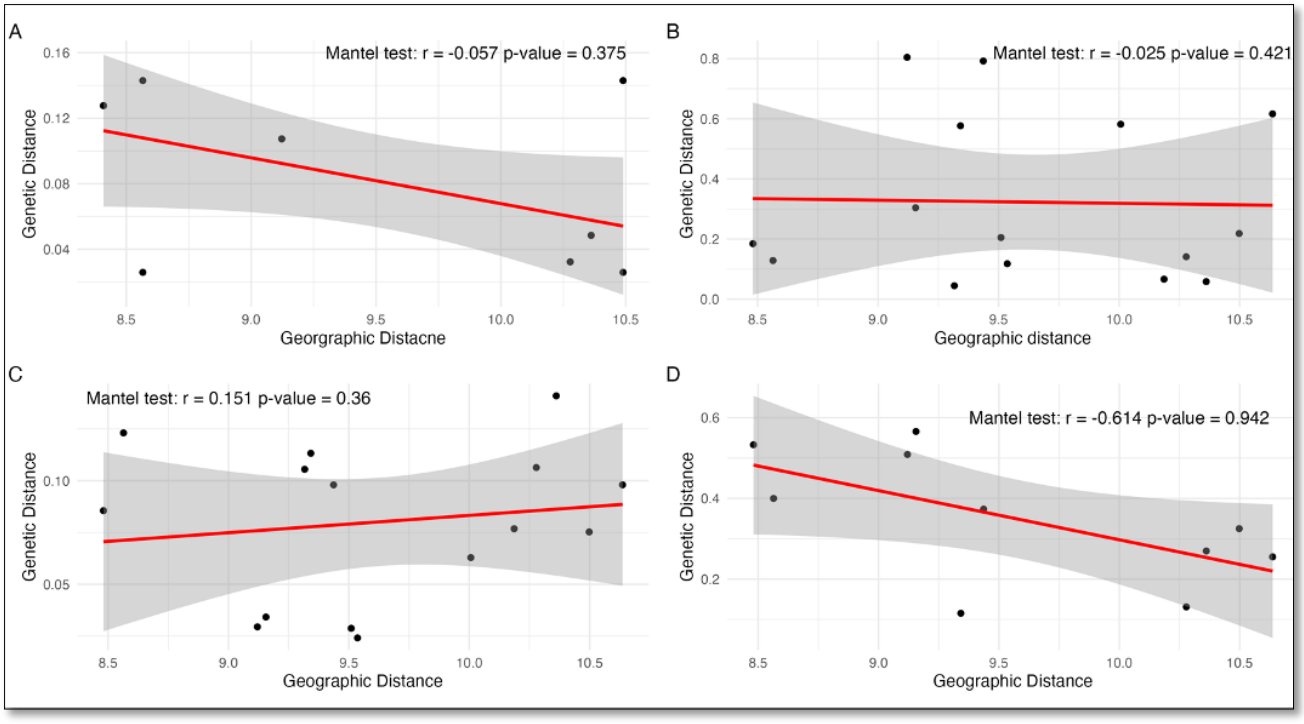


Figure 27. Isolation by distance model calculated as Genetic distance (FST/(1-FST) by log(n) geographic distance with corresponding mantel test and p value for A) *Amphibolis antartica*, B) *Heterozsotera nigricaulis*, C) *Posidonia australis*, and D) *Zostera muelleri.*

4.3 Summary

Overall, the four species of seagrass showed variable levels of genotypic and genetic diversity among sites, and significant levels of genetic structure. Sites varied from being highly clonal (all samples belonging to a single clone), to other sites being highly genotypic diverse with all samples representing unique genotypes. This was practically evident for *P. australis* and *Z. muelleri* with several sites consisting of only one unique genotype indicating the most, if not all of the plants in the population are clonal. These patterns are supported by other studies that have shown levels of clonality can vary widely in both *P. australis* and *Z. muelleri* (Edgeloe et al., 2022; Connolly et al., 2018; Sherman et al. 2016). *A. antartica* and *H. nigricaulis* tended to show higher levels of genotypic diversity with only a single site for each species showing a significant number of clonal genotypes. Further to this, *P. australis* showed a trimodal distribution of genetic distances indicating 'true clones' and 'near clones'. This second 'near clone' group may have occurred through genetic mosaicism, where stomatic mutations within an individual plant led to multiple, closely related genotypes in one plant. This has been observed in *P. australis* in Western Australia (Sinclair et al., 2016), and *Z. marina* in Wales (Reusch & Boström, 2011). To determine if this is the case, further analysis involving sub-sectioning individual leaves to determine any genetic differences within the same individual plant is needed.

Patterns of genetic structure and connectivity showed that *H. nigricaulis* and *Z. muelleri* showed high levels of genetic structure and limited connectivity among some sites. For *H. nigricaulis*, this was driven primarily by the isolation and high levels of genetic differentiation of Killiecrankie Bay, while *Z. muelleri* showed similar levels of genetic differentiation among all sites. *A. antartica* and *P. australis* showed only moderate genetic structure, indicating high levels of connectivity among sites for these species, although there were some sites that were more isolated compared to others. Populations with relatively high levels of genetic differentiation and high levels of clonality, may be at risk from future impacts as their ability to respond or recover from disturbances may be more limited compared to populations that are well connected and have high levels of genetic diversity.

Considering that seagrass beds in Tayaritja Sea Country are in the middle of a climate change hotspot, this may have potential implications for the resilience and persistence of these beds. Monitoring and threat mitigation are required to ensure these valuable sites can continue to support seagrass populations into the future. Interestingly the mantel tests for all four species indicate that geographic distances are not significantly responsible for the genetic differentiation observed, and that other factors including oceanographic currents and other barriers may be responsible for lack of connectivity between sites. This is particularly evident in Duck Island, where, despite being the second northernmost site, it is structurally close to the cluster of sites found in the south of Tayaritja Sea Country in both *A. antartica* and *Z. muelleri.* Further to this, the three sites also have low levels of polymorphic loci, ranging from 7–52% potential limiting the ability for all four species of seagrass across these sites to adapt to external stress including changes in temperature and disease (Duarte et al., 2018; Pazzaglia et al., 2021). Prioritising management and threat mitigation at vulnerable sites that are at most risk would be beneficial. Further understanding barriers relating to dispersal and isolation of populations and understanding whether *P. australis* has been undergoing genetic mosaicism

are needed to ensure populations remain viable into the future in the face of rapid environmental change.

5. Satellite mapping of seagrasses of Tayaritja

5.1 Overview of seagrass mapping activities

The seagrass mapping activities involved the development of a comprehensive Standard Operating Procedure (SOP) for using optical remote sensing to map seagrasses, which was applied to improve data on the distribution of temperate subtidal seagrass in Tayaritja Sea Country (Furneaux Group of Islands), Lutruwita (Tasmania). This two-pronged approach both established standardised methodologies for consistent, repeatable mapping practices across various initiatives and demonstrated their practical effectiveness through Tayaritja as a temperate Australian case study. The project filled a crucial knowledge gap regarding the distribution and ecological importance of temperate subtidal seagrasses in eastern Bass Strait, as identified in the NESP Marine and Coastal Hub national wetlands scoping study. The mapping incorporated satellite imagery analysis, field validation and environmental data, producing detailed seagrass distribution maps that contribute to marine conservation planning, blue carbon assessment, and Aboriginal Sea Country management. These data have been published on the Seamap Australia mapping portal (see section 5.4) and the metadata that references all of the data download points is published here on the IMAS data portal: https://doi.org/10.25959/E4S6-GE74.

5.2 Field data design and acquisition

High-quality training and validation data are fundamental to successful seagrass mapping, as they directly influence classification accuracy and model reliability. In the Tayaritja case study, field data collected using the BOSS camera system served dual critical purposes: providing robust training data for the machine learning classification models and enabling independent validation of the mapping outputs. The publicly accessible nature of the imagery through SQUIDLE+ further enhances transparency and reproducibility of the mapping methodology, allowing future researchers to build upon this foundation.

5.2.1 Sampling design

Field validation points were generated for 398 sites across a polygon defining a maximum depth of 50 m and limited to the western and southern waters of the Tayaritya region using a spatially balanced sampling design using the MBHDesign package (V2.3.14) in R V4.3.1 (Foster, 2021). A spatially balanced approach was employed to ensure that the sampling design did not preference or exclude any region within the mapping zone and that the sampling intensity was as uniform as possible (Kermorvant et al., 2019). In particular, the MBHDesign package was chosen as the package can generate a design that considers the spatial arrangement of legacy sites, reducing the probability that additional points would be placed in close proximity (Foster et al., 2017). Further, the MBHDesign package also allows a weighting variable to increase the probability of sampling in specific areas (Foster, 2021). This can be favourable as increased density can be placed in areas that may contain more heterogeneous habitats which may be more difficult to predict (Foster, 2021).

In this work the design was biased towards areas that have previously contained seagrass or are predicted to currently contain seagrass (the probability of placing a point in a potential seagrass area was 10% higher than that associated with non-seagrass areas – 60% and 40% respectively).

Potential seagrass areas were identified using either polygons of seagrass extent provided from historical mapping efforts (Jordan et al., 1998) or by utilising the outputs of initial desk-based classifications undertaken as part of the initial stages of this project, and review of available satellite imagery and aerial imagery. The sample locations included in the design represented the combination of expert judgement, best practice methods for sample design generation, and available information on seagrass extent within the study locale (Figure 28).

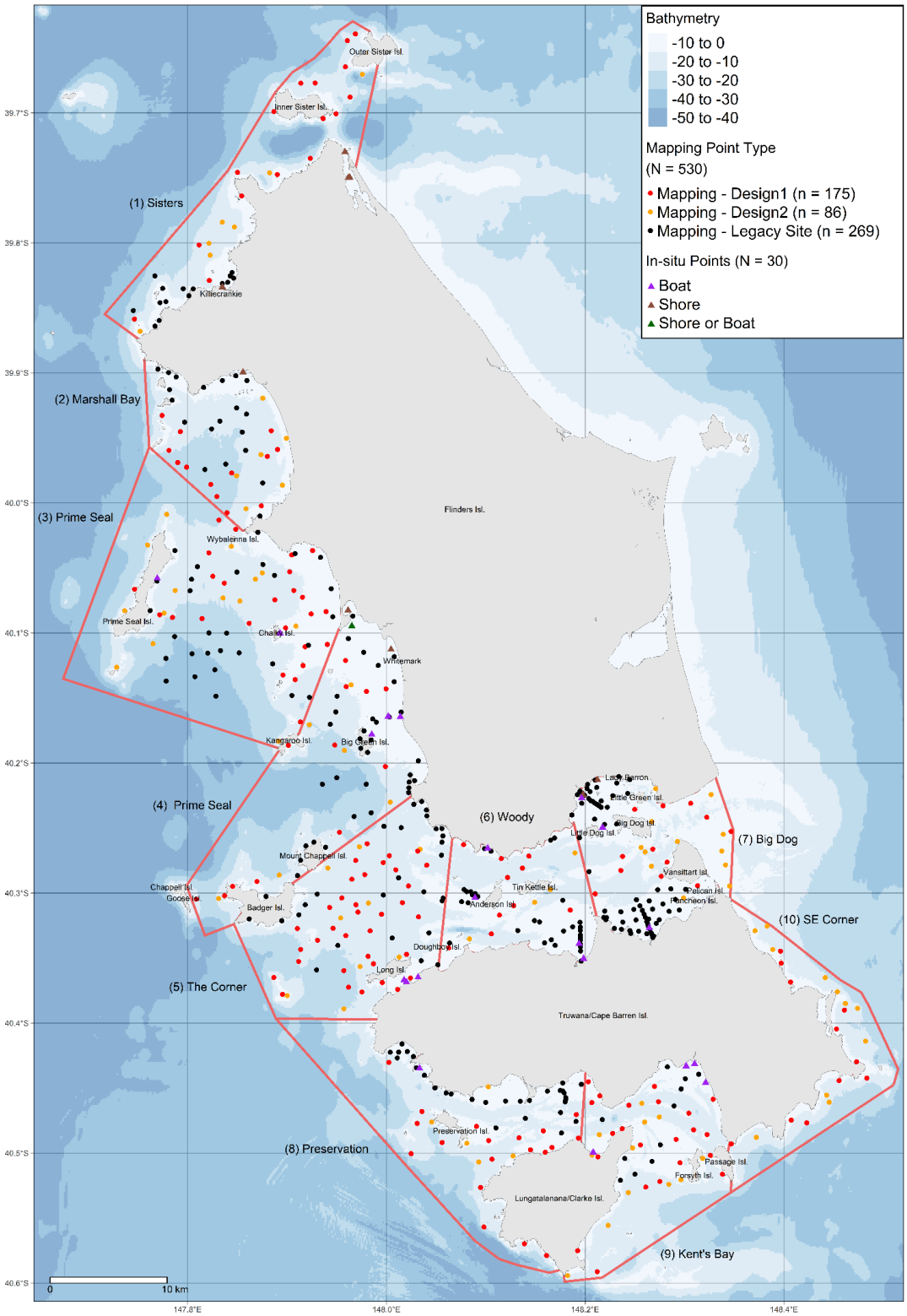


Figure 28. Map depicting field sampling plan for seagrass mapping in Tayaritja Sea Country. Circular coloured points indicate areas were mapping work will be undertaken (camera drops/photoquadrats). Triangular points indicate sites were in-situ sampling (coring, genetics, blue carbon sampling, etc) will be undertaken. Numbers in parentheses indicate ten zones which each contain approximately one day's worth of mapping effort (as estimated by Pakana Rangers).

5.2.3 Field validation using the underwater camera systems

The BOSS (Benthic Observation Survey System) is a new platform designed for rapid assessment of benthic communities (see Figure 29). It features four horizontally facing single or stereo-pair cameras mounted at 90-degree angles within an upright metal frame Langlois et al. Deployed for 3–5 minutes, the BOSS captures a ~270° view of the seafloor.

Metadata such as deployment location, water depth, date, and time were recorded. All deployments followed standard operating procedures outlined in the National Marine sampling field manuals (marine-sampling-field-manual.github.io/). All imagery is publicly accessible at SQUIDLE+ (https://squidle.org).



Figure 29. a) BOSS (Benthic Observation Survey System) *image credit: Jacquomo Monk* b) Pakana Ranger preparing BOSS for deployment *image credit: The Tasmanian Aboriginal Centre.*

5.2.4 Image annotation in Squidle+

The annotation of benthic habitat in each still image was carried out in SQUIDLE+, and followed standard operating procedures (marine-sampling-field-manual.github.io/). For each image, 20 randomly positioned points were allocated across the lower 50% of each image, and the benthic composition was identified under each point based on the Australian Morphospecies Catalogue which is an extension of CATAMI Classification Scheme (Althaus et al., 2015). Morphospecies were schematically translated into Seamap Australia vocabulary (Lucieer et al., 2019) within SQUIDLE+.

5.3 Seagrass and benthic mapping methods

The mapping methodology followed a multi-stage approach as outlined in the published SOP, beginning with the collection of field data using a statistically balanced sampling design to ensure uniform coverage without spatial bias. The Benthic Observation Survey System (BOSS)

was employed to collect high-resolution panoramic imagery of the seafloor, with annotations using the CATAMI classification scheme. For image processing, Sentinel-2 multispectral imagery was compiled into multi-temporal stacks using Google Earth Engine, and cloud masking was applied. Environmental variables including bathymetric LiDAR derived products (bathymetry, slope and rugosity), and modelled wave climate data were incorporated as predictors. Classification was performed using gradient boosting machines, generating three key products: occurrence probability maps (continuous probability surfaces), habitat percentage cover maps (quantitative estimates), and derived binary extent maps. Validation employed a hold-out method with 33% of field data, with accuracy metrics calculated through resampling-based approaches. A comprehensive explanation, along with all required source code (Google Earth Engine and R) and data are provided in the NESP 3.6 case study appendix within the SOP.

5.3.1 Standard Operating Procedure for Seagrass Mapping using Optical Remote Sensing

The Standard Operating Procedure was developed in parallel with the NESP 3.6 project to standardize seagrass mapping methodologies using optical remote sensing, incorporating best practices evolved from decades of remote sensing advances. It primarily recommends Sentinel-2 imagery processed through Google Earth Engine or Python-based Open Data Cube implementations due to optimal spatial resolution, spectral capabilities, and accessibility, while maintaining flexibility for alternative platforms. The SOP emphasises multi-temporal image analysis, integration of environmental variables, hierarchical classification approaches, robust validation procedures, and contextual post-processing. It includes detailed code examples, parameter recommendations, and validation metrics to ensure consistent implementation. The document is designed to be regularly updated, with a formal review schedule to incorporate technological advancements and methodological improvements. The seagrass and habitat mapping in this NESP 3.6 project was used as a case study to demonstrate its application in the real world (it's anticipated more case studies will be added over time).

5.4 Seagrass and benthic mapping results & validation

The mapping results revealed extensive seagrass beds throughout the Tayaritja region. Seagrass probability, extent, percent cover, and fractional cover maps were successfully produced and made available through Seamap Australia. Validation metrics demonstrated high accuracy, with overall accuracy of 91% for seagrass presence/absence classification (95% confidence interval: 0.87-0.92) and root mean square error of 22% for percent cover estimates. Macroalgae and sand habitats were also mapped with high accuracy (82% and 91%, respectively). Analysis of variable importance across the classification models identified depth as the most consistently important predictor of seagrass distribution, followed by Sentinel-2 blue and green bands, and wave exposure metrics. These mapping products served as foundational inputs for the ecological and blue carbon assessments conducted in other sections of the project, enabling quantification of seagrass area by species, supporting the ecological significance assessment of different seagrass communities, and providing spatial context for the mollusc biodiversity studies.

The remote sensing results indicate that seagrass covers 33% of the subtidal area down to a depth of 30 m across the entire island group (see image below). However, if the east coast is excluded, as indicated by the red polygon in the image, which was outside the scope of the

study, seagrass coverage increases to 47% of the subtidal area. The total area of seagrass mapped within the zone was 755.80 km^2 (Table 9).

Table 9 Seagrass area calculations for mapped area.

Calculation	Total in km^2
Seagrass area total	755.80
Seagrass area (subtidal)	725.87
Seagrass area (intertidal)	29.93
Seagrass area (<20% cover)	67.63
Seagrass area (20-40% cover)	13.45
Seagrass area (40-60% cover)	177.12
Seagrass area (60-80% cover)	181.21
Seagrass area (>80% cover)	316.38

The seagrass occurrence probability map (Figure 30) provides a comprehensive assessment of the likelihood of seagrass presence across the study area, offering valuable insights into areas where seagrass is most likely to be found based on remote sensing data. This map is a crucial tool for understanding seagrass distribution patterns.

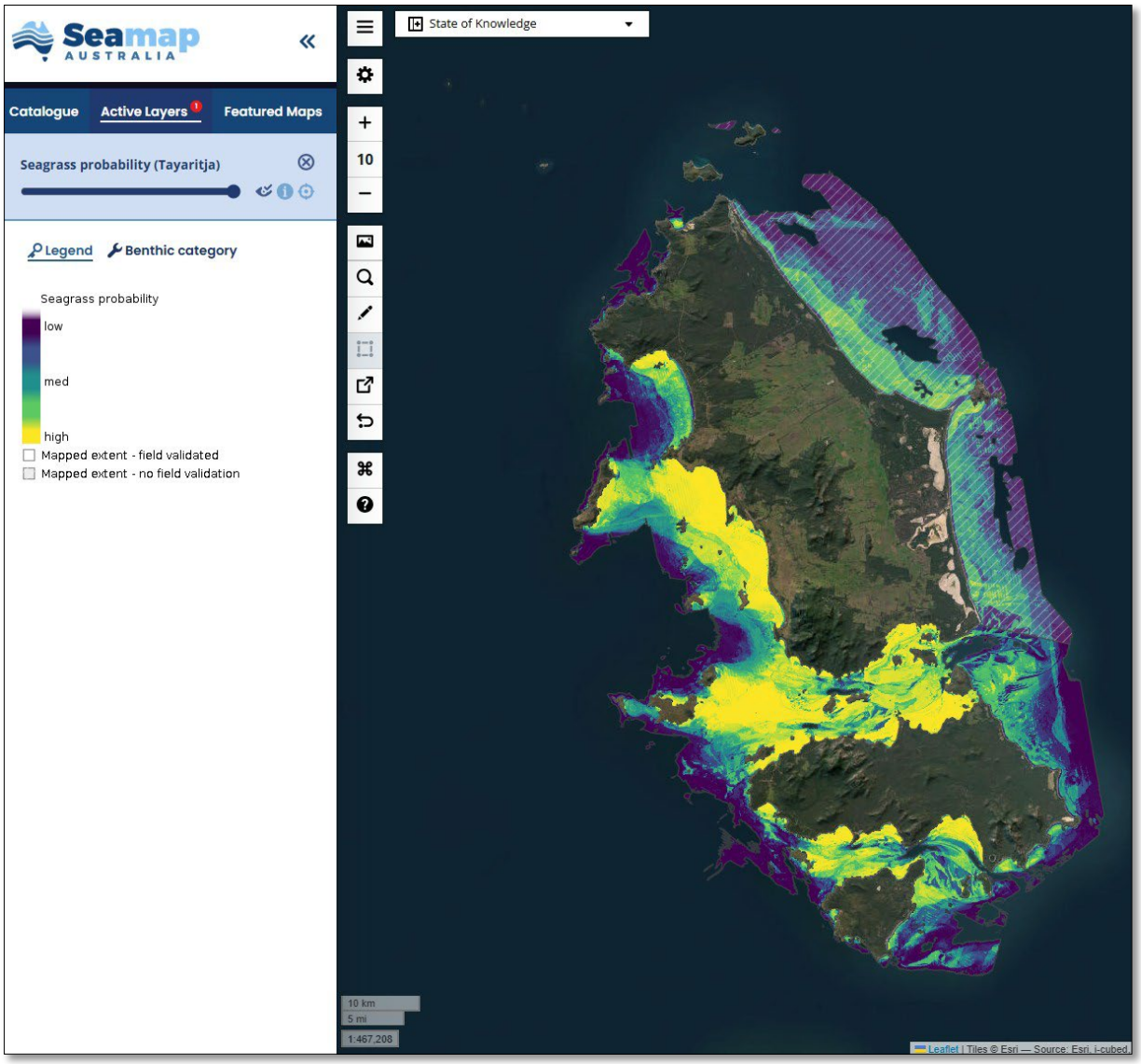


Figure 30. Seagrass occurrence probability map, showing the likelihood of seagrass presence across the study area based on remote sensing data. https://seamapaustralia.org/map/#f432bcab-ad48-45ba-ba30-c0f590272c16

Complementing this, the macroalgae occurrence probability map (Figure 31) highlights the areas where macroalgae are most likely to occur. This map provides a comparison to the seagrass data, offering a broader view of the benthic habitat structure in the region.

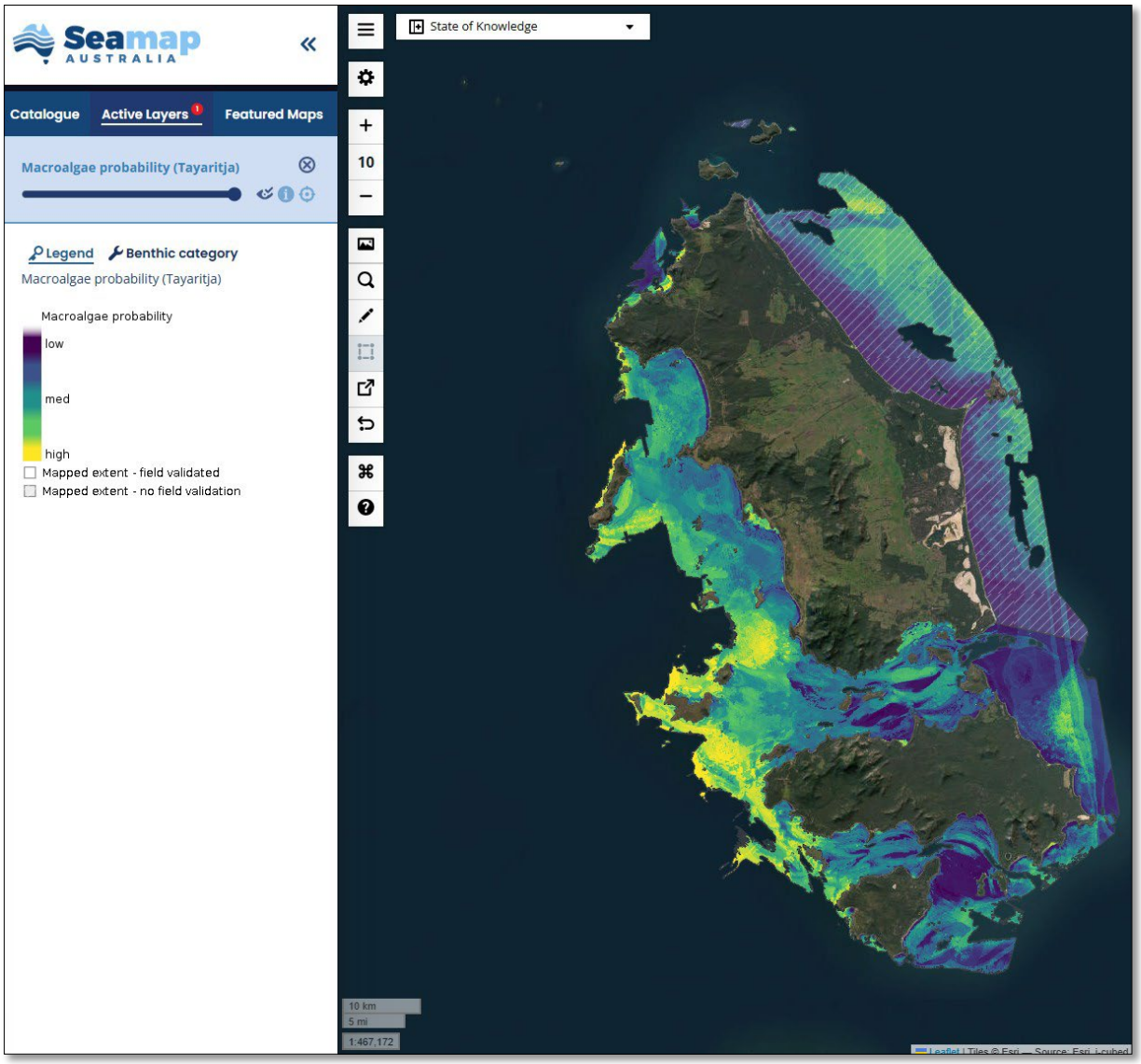


Figure 31. Macroalgae occurrence probability map, highlighting the areas most likely to support macroalgae, complementing the seagrass occurrence data. https://seamapaustralia.org/map/#f4028dcd-4d74-4c36-845d-ea1cd2b733d5

Additionally, the sand occurrence probability map (Figure 32) shows the distribution of sandy areas, a key factor in determining habitat suitability for various marine organisms, especially those in seagrass and macroalgae communities.

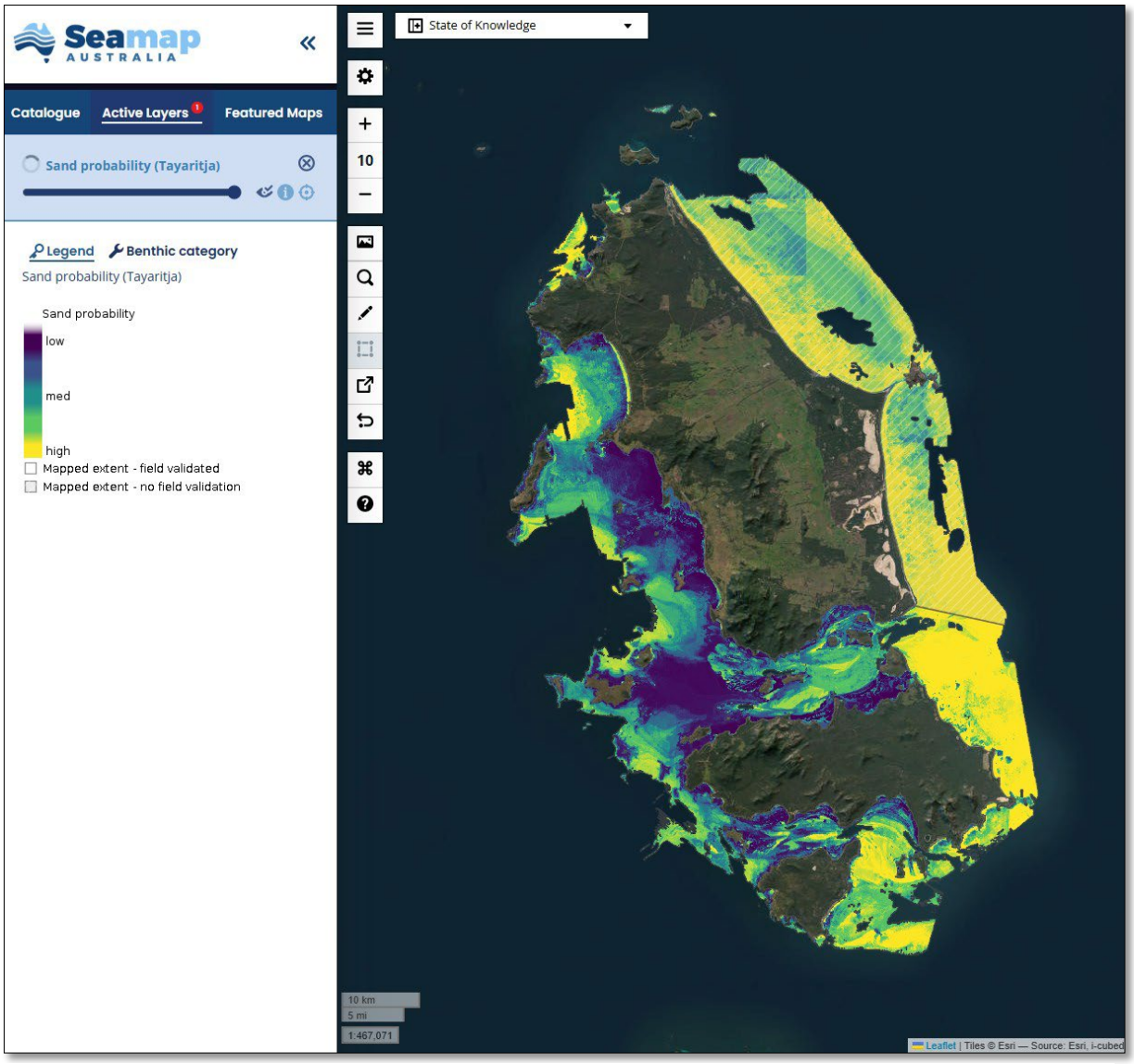


Figure 32. Sand occurrence probability map, illustrating the distribution of sandy substrates across the study area. https://seamapaustralia.org/map/#77b6785a-c726-402e-a6f5-c40990ea6cb2

Further, the seagrass percentage cover map (Figure 33) provides a clear depiction of the proportion of seafloor covered by seagrass, allowing for a quantitative evaluation of seagrass abundance. Similarly, the macroalgae percentage cover map (Figure 34) presents the extent of macroalgae coverage, providing a complementary measure to the seagrass cover data. The sand percentage cover map (Figure 35) details the proportion of seafloor covered by sand, a fundamental aspect for understanding the substrate dynamics and habitat composition within the study area.

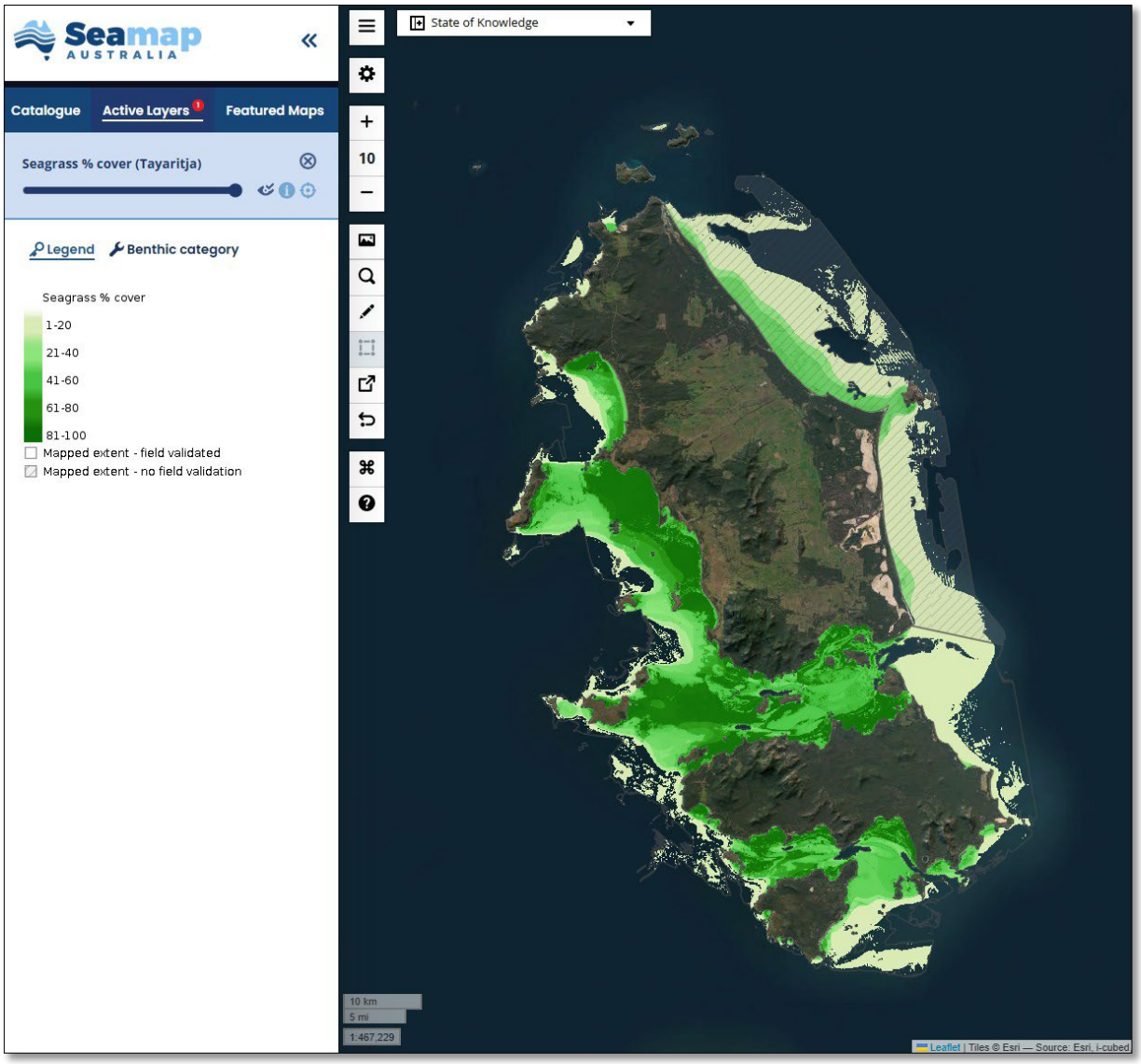


Figure 33. Seagrass percentage cover map, displaying the proportion of seafloor covered by seagrass, aiding in the assessment of seagrass abundance. https://seamapaustralia.org/map/#aecb60fd-80bf-40a9-b0a2-4bc6cffbefdb

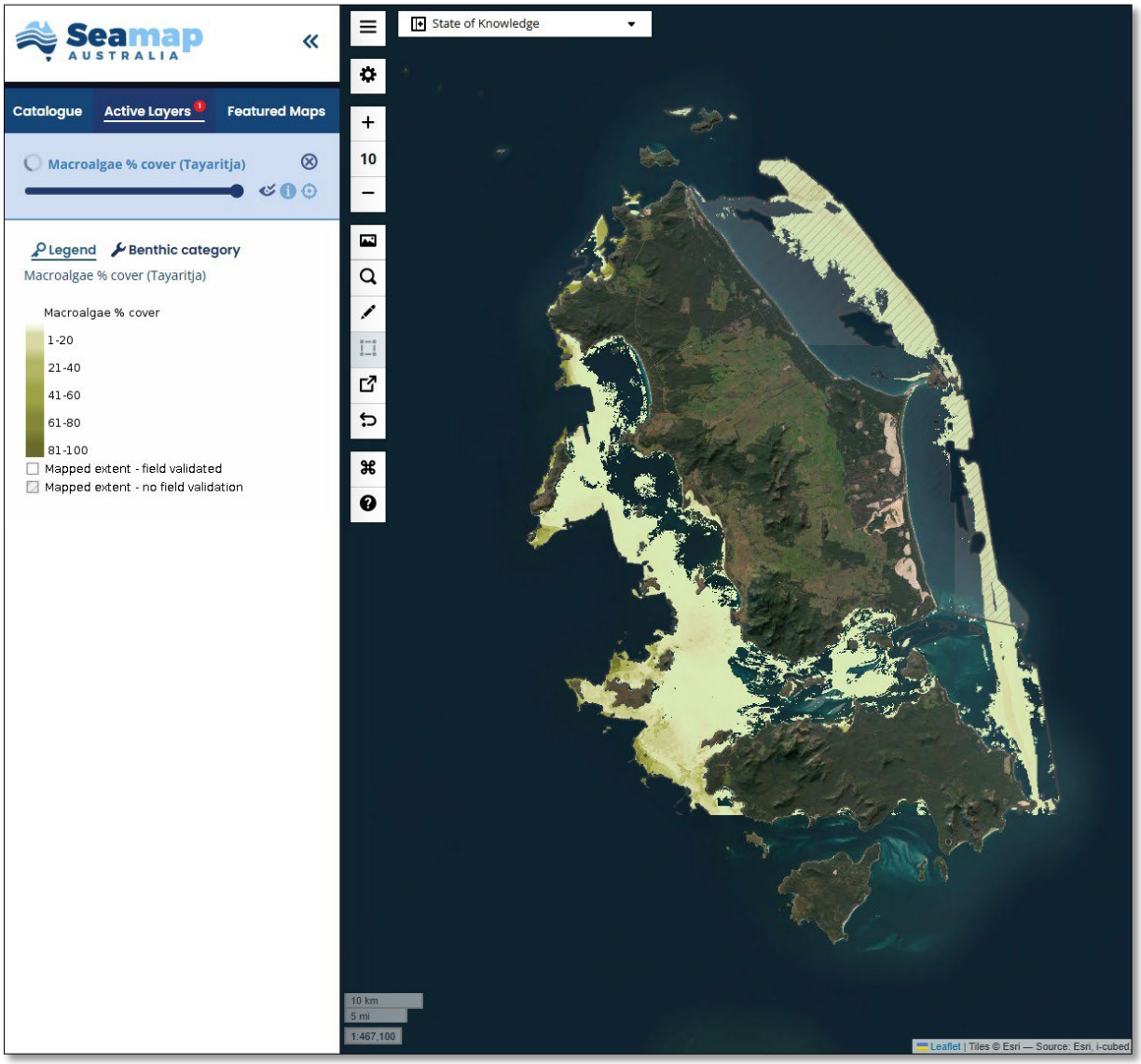


Figure 34. Macroalgae percentage cover map, presenting the extent of macroalgae coverage, providing a measure of macroalgal abundance in the region. https://seamapaustralia.org/map/#1c133d7d-8f4d-4e96-8713-06279833603a

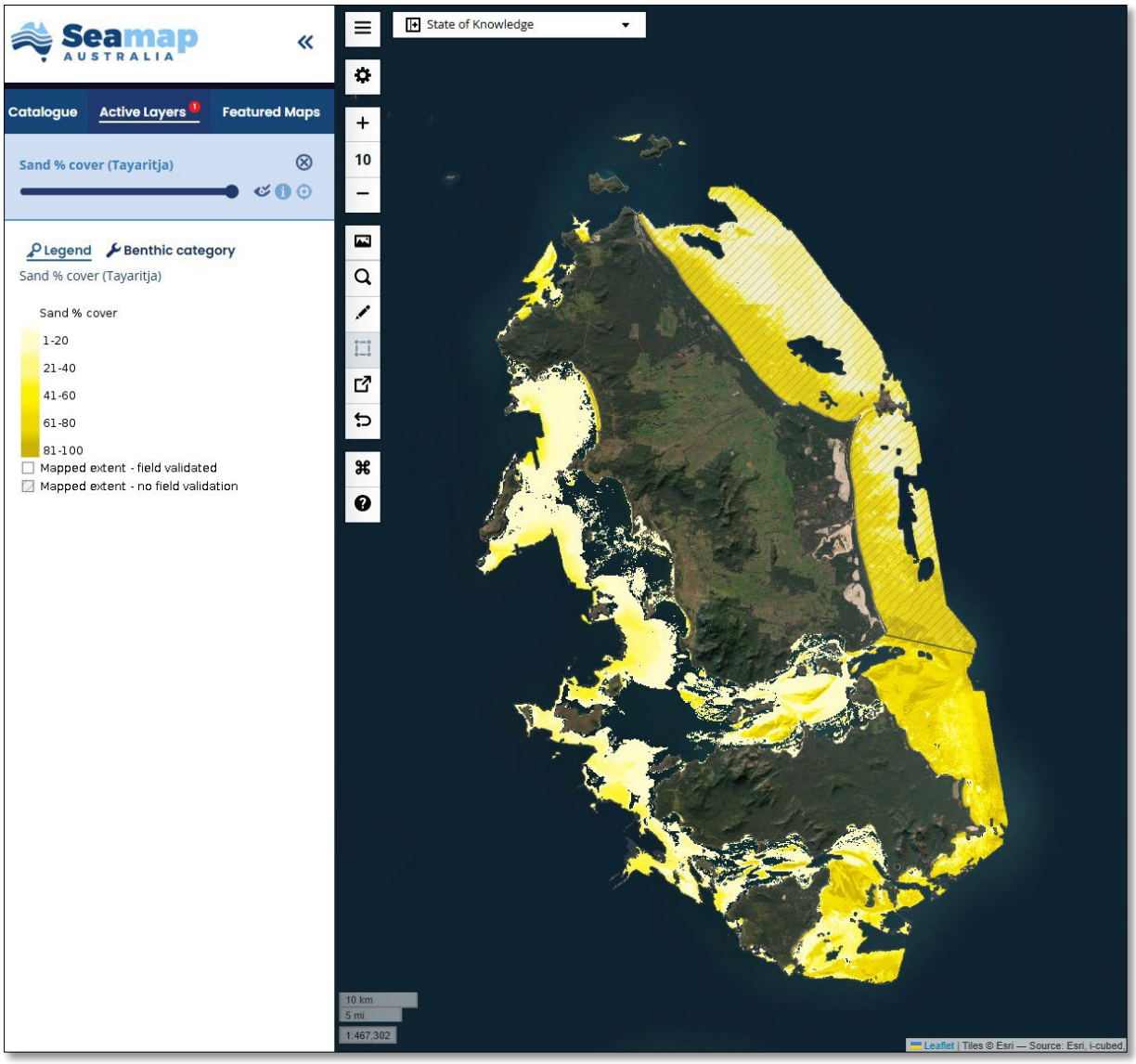


Figure 35. Sand percentage cover map, detailing the proportion of seafloor covered by sand, a key substrate type in benthic habitat composition. https://seamapaustralia.org/map/#b707ac08-e307-4c1e-a9e6-5e303cdb3904

To visualise the spatial distribution of benthic habitats, the fractional cover visualisation in RGB view (Figure 36) presents a colour-coded map, illustrating the distribution of seagrass, macroalgae, and sand. This visualisation helps in quickly identifying the areas dominated by each substrate type. Moreover, the fractional cover visualisation in the occurrence probability view (Figure 37) provides a predictive map of the likelihood of different benthic habitats across the region, while the fractional cover visualisation in the percentage cover view (Figure 38) gives a more precise view of substrate coverage based on percentage data.

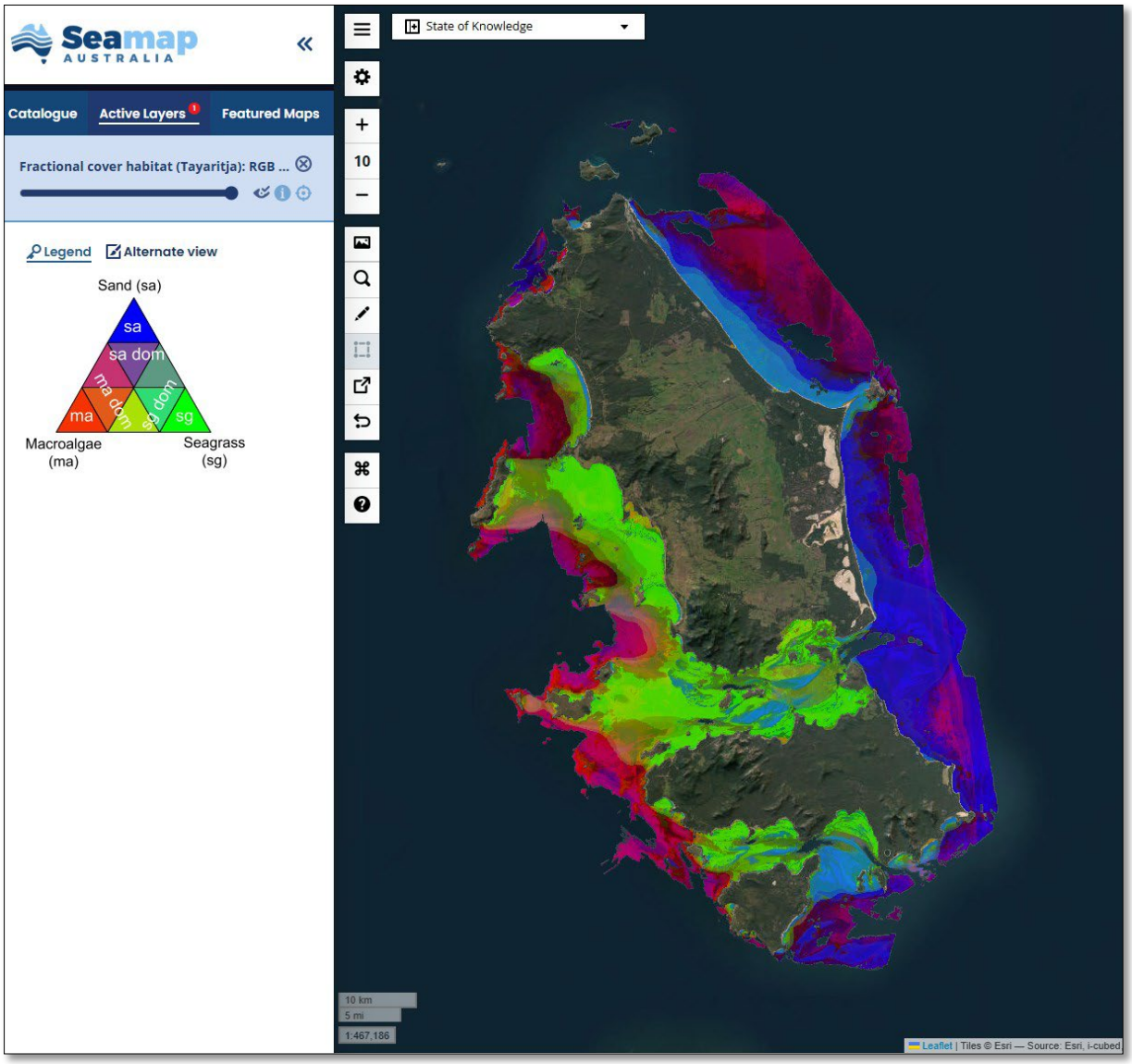


Figure 36. Fractional cover visualisation in RGB view, using a colour-coded approach to show the distribution of seagrass, macroalgae, and sand across the study area. https://seamapaustralia.org/map/#ae6ccd5f-6532-4758-8b3c-5246ebc65840

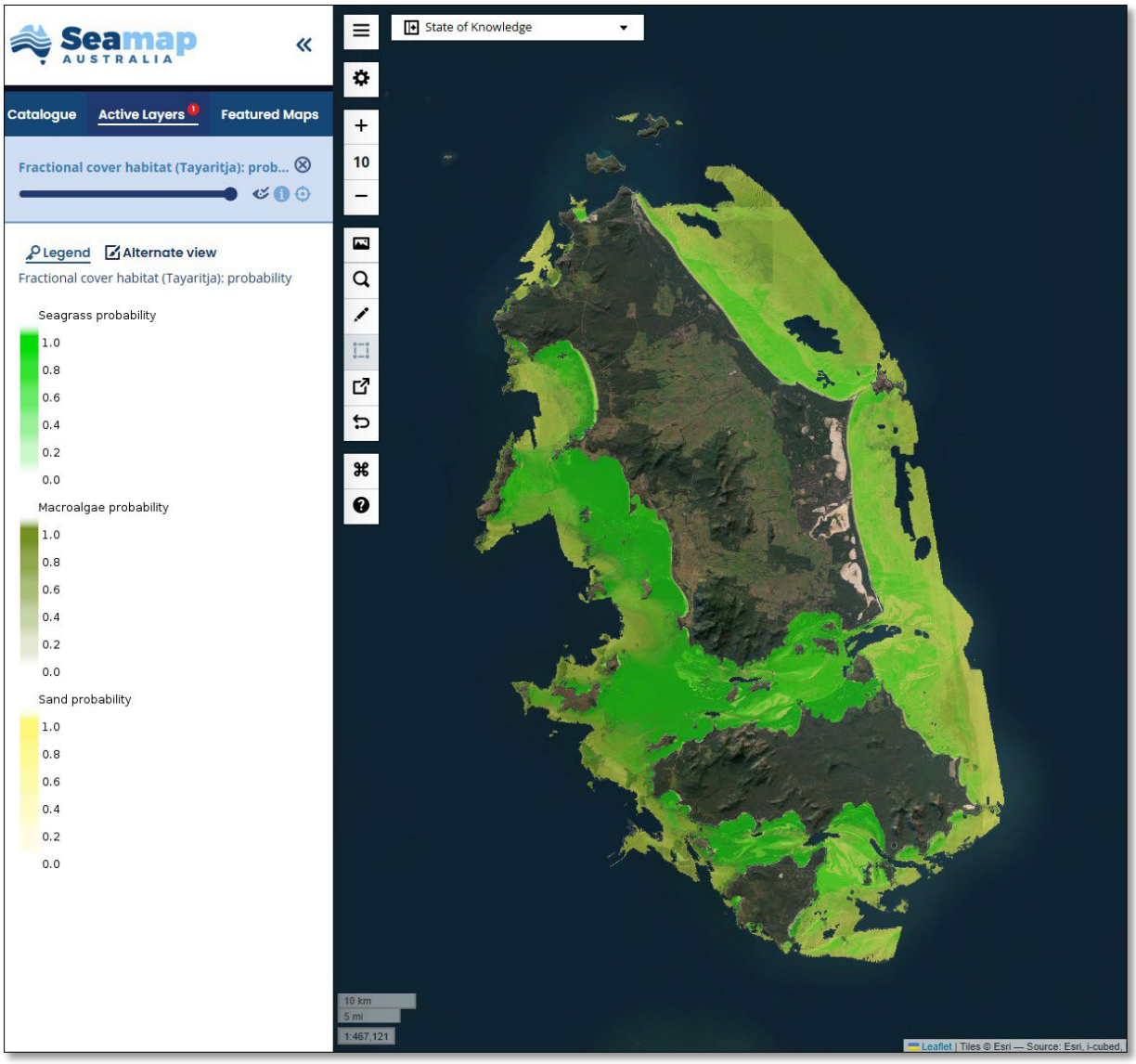


Figure 37. Fractional cover visualisation in occurrence probability view, illustrating the likelihood of different benthic habitats being present across the region. https://seamapaustralia.org/map/#dcb494f6-d0c2-481d-9d3f-92a6de7d0879

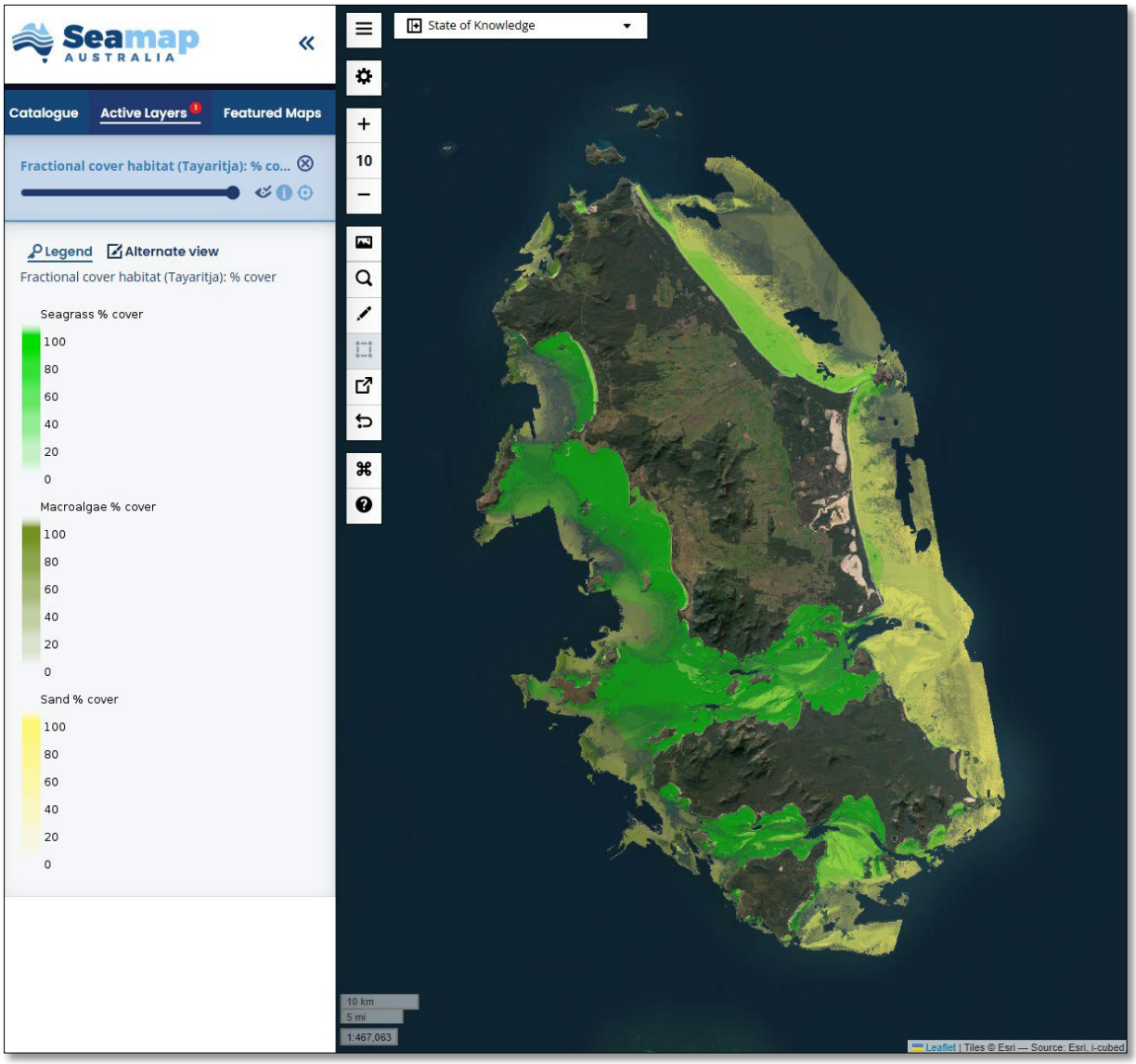


Figure 38. Fractional cover visualisation in percentage cover view, showing the coverage of seagrass, macroalgae, and sand as a percentage of the seafloor. https://seamapaustralia.org/map/#13dc94a6-2b4b-4872-bbbe-ccfe1c5b4122

The baseline seagrass extent map (Figure 39) offers a reference point for the initial distribution of seagrass, facilitating future comparisons and tracking changes in seagrass coverage over time. Similarly, the baseline macroalgae extent map (Figure 40) provides the initial distribution of macroalgae, serving as a baseline for assessing shifts in macroalgal coverage. The baseline sand extent map (Figure 41) shows the initial extent of sandy areas, offering a foundational dataset for understanding the dynamics of the study area's substrates.

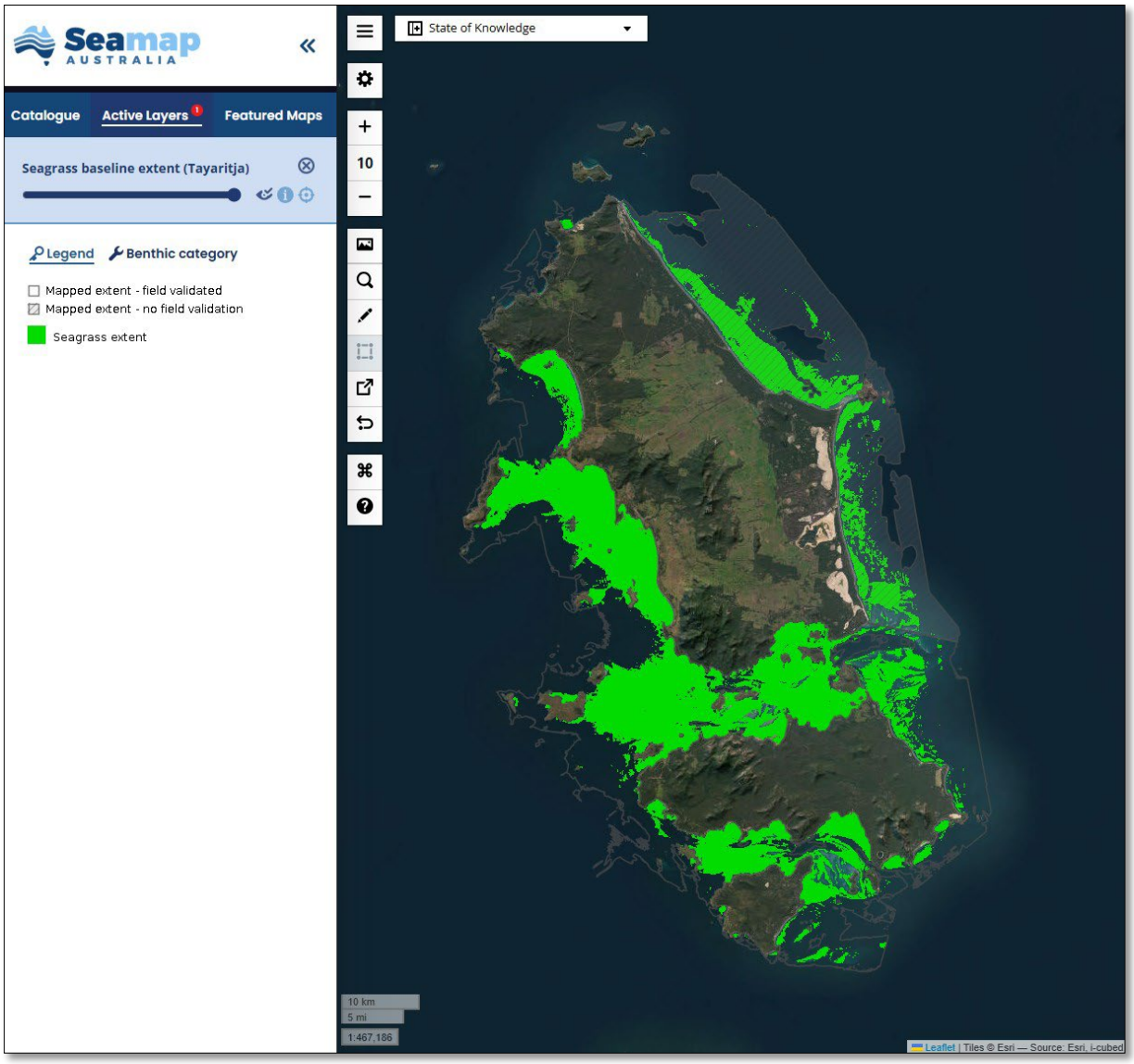


Figure 39. Baseline seagrass extent map, providing a reference for the initial distribution of seagrass in the study area. https://seamapaustralia.org/map/#32f72d20-fca1-4395-8718-8cf1b4507cc9

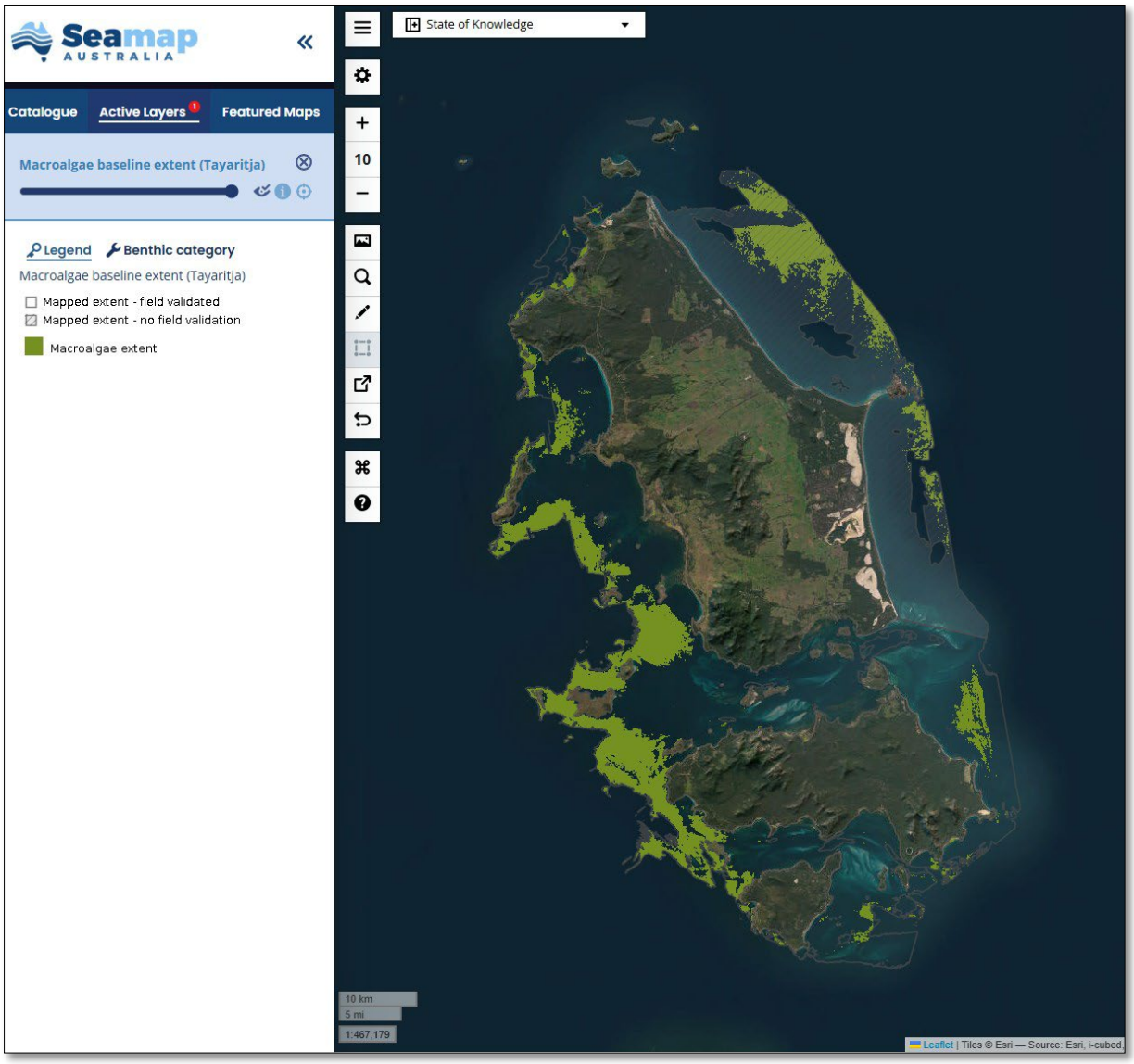


Figure 40. Baseline macroalgae extent map, showing the initial distribution of macroalgae and serving as a baseline for future monitoring. https://seamapaustralia.org/map/#9d7ee933-d54c-474d-a128-c8b2ba9b6b1c

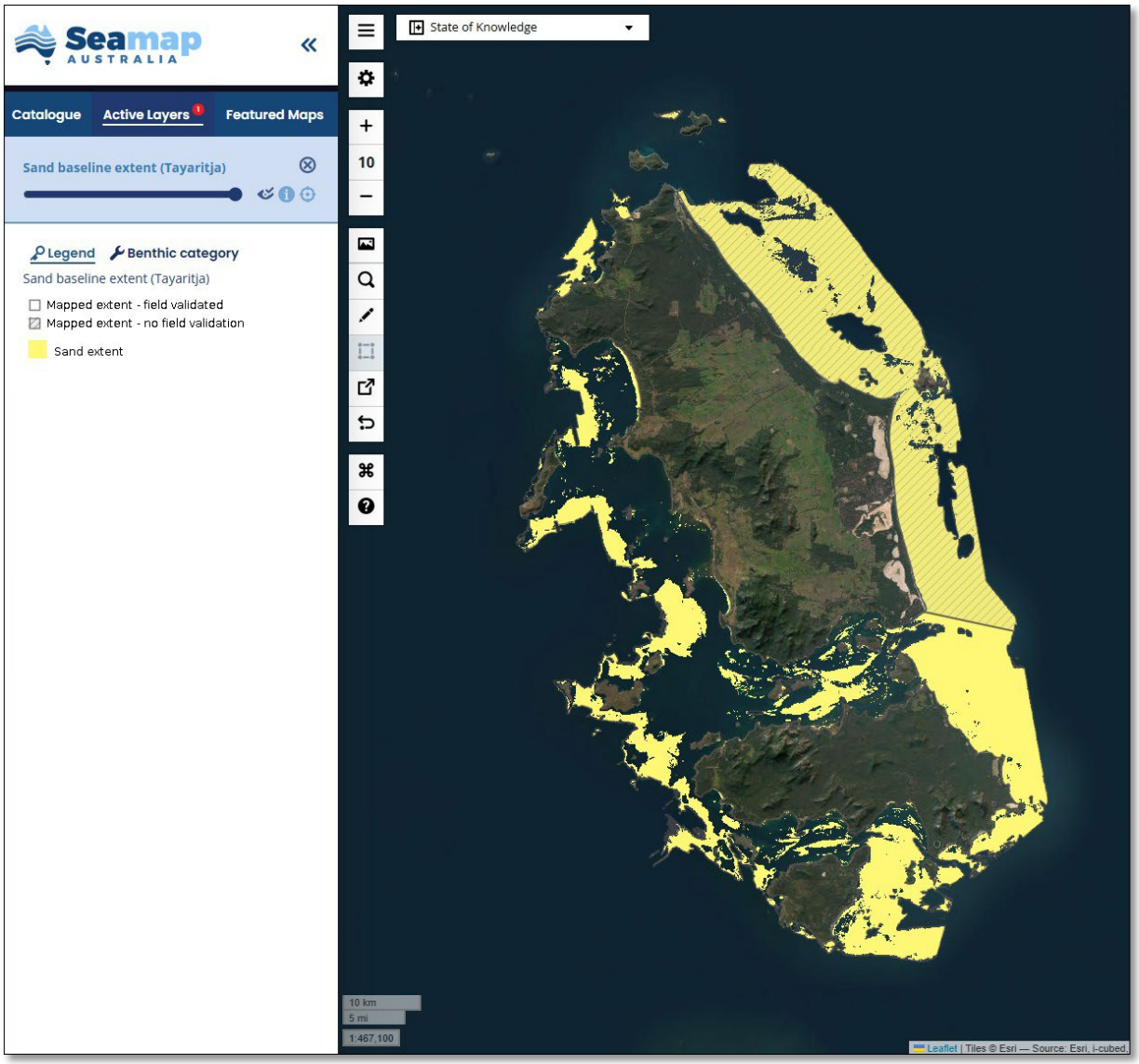


Figure 41. Baseline sand extent map, presenting the initial extent of sandy areas within the study region, aiding in the understanding of substrate dynamics. https://seamapaustralia.org/map/#44009426-b40c-499b-909d-d47e91871c1f

Finally, the combined benthic habitat map (Figure 42) integrates the seagrass, macroalgae, and sand data into a single map, providing a comprehensive overview of the benthic habitat composition in the region. This is complemented by the dominant seagrass species map (Figure 43), which identifies the key species of seagrass in the area, offering valuable information for habitat conservation and management.

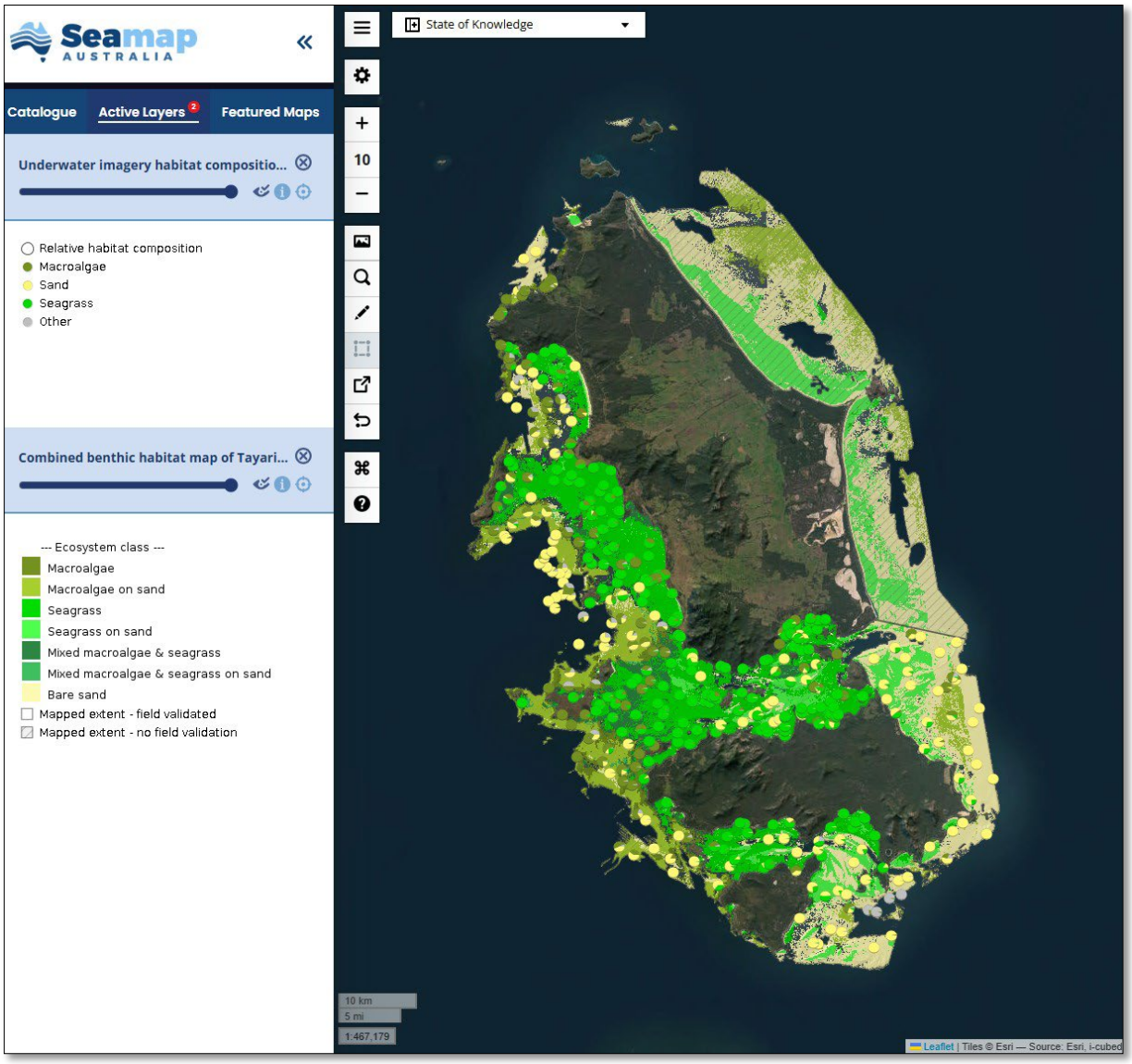


Figure 42. Combined benthic habitat map, integrating seagrass, macroalgae, and sand data into a single map to offer a comprehensive view of the benthic habitat composition. Includes the BOSS observation results as piecharts overlaid on map. https://seamapaustralia.org/map/#fce28240-26d0-475a-a5ab-497b0d33a8b1

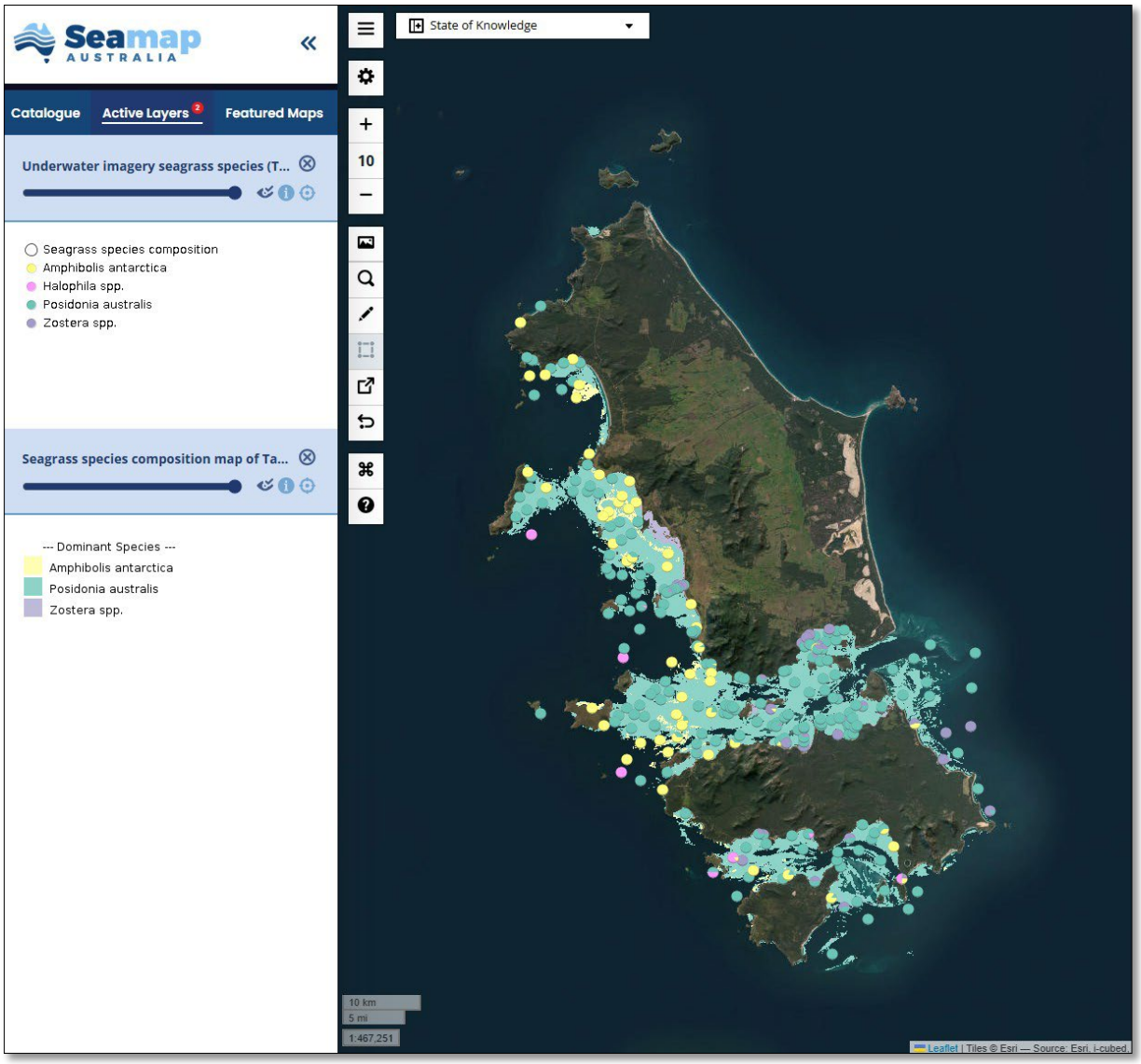


Figure 43. Dominant seagrass species map, identifying the primary species of seagrass present in the study area, providing insights for habitat management and conservation. Includes the BOSS observation results as piecharts overlaid on map. https://seamapaustralia.org/map/#2d51233e-ad30-4e8c-9c56-cde4476097d8

6. Discussion: research impact

This project was a collaborative research initiative aimed at improving the mapping, monitoring, and management of seagrass habitats in Tayaritja (Furneaux Islands) region of Tasmania. The area encompasses culturally significant Sea Country for the Tasmanian Aboriginal community, and records of its subtidal seagrass beds was limited to coarse, decades-old surveys. The project addressed a critical knowledge gap by establishing robust baseline data on seagrass distribution, ecological value, and blue carbon potential. The project was conducted in partnership with the Tasmanian Aboriginal Centre and Pakana Rangers, ensuring Aboriginal knowledge and community priorities guided the research in a culturally sensitive way. By focusing on both ecological science and cultural collaboration, the project set out to support seagrass conservation, blue carbon assessment, and inclusive coastal stewardship in the region.

Key outcomes of the project included:

1. High-resolution habitat maps: Using satellite remote sensing and machine learning, the team mapped approximately 1,539 km² of sea country, identifying 755.8 km² of seagrass beds – one of the most extensive temperate seagrass areas in Australia These maps distinguish seagrass from other habitats (e.g. macroalgae or sand) and greatly improve upon historical coarse-scale mapping.

2. Ecological baseline data: Comprehensive field surveys documented seagrass species composition (*Posidonia, Amphibolis, Heterozostera, Zostera* spp.*)* and the rich communities of invertebrates (e.g. molluscs) they support. The findings confirm that these underwater beds function as biodiversity hotspots, providing food and shelter for a wide array of marine life

3. Blue carbon assessment: Sediment core sampling at ten representative sites quantified substantial below-ground carbon stocks within seagrass beds. This information fills a gap in understanding the carbon storage capacity of Tasmania's seagrasses and is essential for future climate change mitigation strategies.

4. Cultural collaboration and capacity building: The project was co-designed and carried out with Pakana Rangers. Pakana Rangers participated in all fieldwork, received training in marine monitoring techniques, and helped develop culturally appropriate monitoring protocols. This approach empowered the local community and ensured that traditional knowledge and cultural values were integrated throughout.

The sections below discuss these outcomes in detail and highlight their significance for biodiversity assessment and cultural collaboration.

6.1 Seagrass ecosystem diversity, biodiversity and genetic connectivity

The research revealed the high ecological value of Tayaritja's seagrass beds. Four key seagrass species were recorded – *Posidonia australis, Amphibolis antarctica, Heterozostera nigricaulis,* and *Zostera muelleri* – spanning habitats from intertidal shallows to depths of ~25 m. These beds represent some of the largest intact seagrass beds in southeastern Australia. The structural complexity of the seagrass (from dense *Posidonia* beds to mixed-species stands) creates diverse microhabitats that support a variety of marine organisms. This study confirms that seagrass beds act as biodiversity hotspots, underpinning rich communities of fish, crustaceans, molluscs and other fauna.

Through quantitative surveys, the project evaluated the relationships between seagrass habitat attributes and faunal communities. Through field surveying with wading and snorkel diving, data on mollusc abundance, species richness, biomass, and secondary production across different seagrass types was collected. Initial results showed significant variations in these metrics among seagrass species and sites, indicating that each seagrass type contributes uniquely to habitat complexity and productivity. Dense *Posidonia*, for instance, were associated with especially high mollusc diversity and biomass: a sign of healthy, functioning ecosystems. These findings validate that temperate seagrass beds in Tayaritja Sea Country are critical for maintaining marine biodiversity and ecological function. They provide foraging grounds and shelter for numerous species, some of which are culturally and economically important (e.g. finfish and scallops that spend part of their life cycle in seagrass). By documenting species presence and ecological interactions, the project filled knowledge gaps on species diversity and ecosystem function in this remote region.

Genetic connectivity of the seagrass beds was an additional ecological dimension examined in this project. Preliminary genetic analyses (Single Nucleotide Polymorphism markers) were conducted for selected species (e.g. *Posidonia australis, Heterozostera, Zostera*) to understand population structure across the islands. The detailed results in Chapter 4 demonstrate how this work is important for conservation. High genetic diversity and connectivity can indicate greater resilience of seagrass beds to disturbances. Insights into clonality and gene flow among beds help to inform biodiversity conservation strategies, ensuring that distinct genetic populations are managed appropriately (for example, identifying if certain bays host unique genetic variants that merit protection). This genetic component complements the biodiversity assessment by highlighting the underlying resilience and adaptive capacity of the seagrass ecosystem. In summary, the ecological surveys conducted through this project demonstrate that Tayaritja's seagrass beds harbor a rich biodiversity and perform vital ecological roles. This new knowledge provides a scientific basis for protecting these habitats as keystone ecosystems in the region. It also establishes baseline metrics (species lists, abundance, biomass, etc.) against which future changes can be monitored by scientists and rangers alike.

6.2 Blue carbon findings and climate relevance

A major focus of this project was assessing the blue carbon value of the seagrass habitats – their capacity to store organic carbon in biomass and sediments. The team collected sediment cores (to 50 cm depth) from ten seagrass sites distributed around the Furneaux Islands to measure organic carbon content. Analysis of these cores revealed high sedimentary carbon

stocks in the seagrass beds, though with notable spatial variability among sites (influenced by factors like seagrass species and local hydrology). In beds dominated by *Posidonia australis* – a long-lived, large-bodied seagrass – carbon densities were particularly high, reflecting thick mats of roots/rhizomes and associated organic-rich sediments. Sites with finer-leaved seagrass (*Zostera* or *Heterozostera*) still showed substantial carbon storage but generally at lower levels, aligning with differences in plant morphology and biomass. These results mark the first detailed quantification of blue carbon in Tayaritja Sea Country, addressing a key gap in regional climate data.

The implications for climate policy are significant. Seagrass beds are globally recognised as effective natural carbon sinks, capable of trapping carbon for millennia under anaerobic seabed conditions. While per-hectare carbon stock can vary, the sheer extent of the mapped Tayaritja beds makes them nationally important carbon stores. By establishing current carbon stock baselines, this project provides the evidence needed to include these ecosystems in climate change mitigation planning. For instance, protecting these intact beds aligns with strategies to prevent carbon emissions from disturbed coastal sediments. Conservation of seagrass beds ensures that the large carbon reservoirs remain sequestered and are not released by habitat degradation (e.g. due to dredging or poor water quality).

Moreover, the blue carbon data have direct management relevance for Lutruwita/Tasmania and Tayaritja. Knowing where carbon-rich sediments lie can help prioritise areas for protection (for example, designating certain seagrass beds as off-limits to damaging activities). It also sets a benchmark to evaluate the success of future restoration – any new seagrass growth could be monitored for additional carbon accumulation over time. In summary, the project's blue carbon assessment not only enriches scientific understanding of carbon storage in temperate seagrasses but also guides climate-related strategies at local and national levels, emphasising the climate mitigation value of conserving these blue ecosystems.

6.3 Mapping innovation and seagrass distribution

Another impactful outcome of this project has been the development and application of innovative mapping techniques for mapping temperate seagrass distribution in the Tayaritja region for the first time from satellite data. Prior to this project, the marine environment of Tayaritja had not been mapped for more than 30 years, especially in deeper waters, limiting the ability of managers to assess habitat extent and condition. To overcome this, the project has delivered a comprehensive Standard Operating Procedure (SOP) for seagrass mapping using optical remote sensing that has been tailored to temperature marine waters (see Appendix 2).

This method combined high-resolution satellite imagery (Sentinel-2 at 10 m, Planet Dove at ~3 m) with bathymetric LiDAR and extensive field validation. Using machine-learning classification algorithms trained on field data, the team produced detailed benthic habitat maps for over 1,500 km² of Sea Country. Crucially, about 755.8 km² of that area was confirmed as seagrass habitat – a figure that vastly refines previous estimates and illustrates the true scope of the seagrass habitat. The mapping effort did not just delineate seagrass; it also differentiated between habitat types (e.g. macroalgae-dominated areas, unvegetated sands) to provide a holistic picture of the marine environment. The result is a set of high-resolution habitat maps that are invaluable for planning and management. These maps have been made publicly

accessible, including via the Seamap Australia portal and the IMAS data repository, ensuring that stakeholders can readily use the information.

A novel underwater camera system (the Benthic Observation Survey System, BOSS) was deployed at nearly 400 validation points across the islands. This provided georeferenced photographs and videos of the seabed to verify satellite predictions. The integration of community knowledge was also evident here – local rangers and boat skippers helped optimise the field survey design, ensuring even remote and deep locations were sampled. These ground-truth data were used in the classification models, the team achieved high mapping accuracy despite challenges like turbid waters and variable depth.

The methodology accounts for deep-water seagrass, a typically hard-to-map component, by leveraging LiDAR bathymetry, satellite imagery and key image analysis techniques, thereby extending mapping beyond the limits of past studies (which were constrained to ~20 m depth). The mapping innovations from this project fill a crucial knowledge gap in Australia's coastal habitat data. Eastern Bass Strait's temperate seagrass beds were identified as a gap in a national wetland scoping study, and Project 3.6 directly responded to that need. Now, resource managers and the community have an authoritative map of seagrass extent in the Furneaux Islands. This empowers more effective spatial planning – for example, zoning of marine use can be informed by the locations of sensitive seagrass beds. It also allows for monitoring change over time: having a baseline map means future surveys can detect gains or losses in seagrass coverage (important for tracking impacts of climate change or human activities). Finally, by establishing standardised mapping protocols, that have been applied in tropical areas of Australia, the project sets a template that can be replicated in other temperate regions, enhancing national capacity for seagrass monitoring nationwide. In summary, the maps not only show where seagrasses are and how extensive they are but also provides tools and methods for ongoing management and research across Australia's coasts.

6.4 Policy management and relevance

The findings of this project carry strong policy and management significance, providing actionable knowledge for both government environmental bodies and the Tasmanian Aboriginal community. Key ways in which this project may inform policy include:

Guiding Indigenous Protected Area (IPA) planning: The research is timely making the data available to support the establishment of the proposed Tayaritja Wukalina milaythina muka Indigenous Protected Area – the first Sea Country IPA in Tasmania. The data collected on seagrass distribution, species diversity, and ecological function directly feed into the IPA's management plan. The Palawa community had already identified seagrass beds and their associated marine life as a conservation target in their draft IPA plan; this project provided scientific evidence to support this. The mapping of the largest intact seagrass beds in southeastern Australia within the IPA boundaries demonstrates the area's national significance. IPA planners can use the high-resolution habitat maps to delineate zones of high conservation value, foci areas for monitoring, and identify areas for potential community-led restoration. Furthermore, the project's inclusive approach strengthens the IPA ethos: it showed how co-management can work in practice, with Pakana Rangers and scientists jointly producing knowledge. This lays the groundwork for Pakana Rangers monitoring and management once the IPA is formally declared. Project 3.6 has thereby informed the Tasmanian Aboriginal

community's efforts to establish the Tayaritja Wukalina milaythina muka IPA by providing a baseline inventory of natural values and a trained Pakana Rangers in methods that can be used monitor seagrass beds into the future.

Enhancing biodiversity conservation strategies: At the state and national level, the outcomes improve our ability to conserve coastal biodiversity. Marine managers can now incorporate this new seagrass data into broader conservation planning and reporting. For example, areas identified as biodiversity hotspots (with high mollusc or fish diversity) can be flagged for heightened protection under marine conservation policies. The Tasmania Parks and Wildlife Service, or the Commonwealth Government, can use the maps and species data to update seagrass inventories and assess whether existing protected area networks adequately cover these critical habitats. The project also contributes to Australia's obligations under international agreements (such as the Convention on Biological Diversity) by filling information gaps on an important ecosystem. Additionally, knowing the genetic connectivity patterns of seagrass can inform translocation or restoration decisions – if some beds are isolated, managers might prioritize connecting habitats or safeguarding those unique gene pools. Overall, by addressing the lack of mapping and ecological data for temperate seagrass, the project enables evidence-based decisions for biodiversity conservation in the region.

Informing climate change and blue carbon policy: The quantified carbon stocks from this study provide a concrete basis for including seagrass beds in climate mitigation strategies. Policymakers in climate and environmental portfolios can reference this work to advocate for blue carbon initiatives. For instance, the Tasmanian Government could integrate seagrass protection into Tasmania's Climate Action Plan, recognising that protecting these ecosystems helps lock away carbon.

Climate adaptation plans for coastal areas can leverage these findings – conserving seagrass not only mitigates emissions but also bolsters ecosystem resilience (seagrasses stabilise sediment and dampen wave energy, helping protect shorelines). The project's emphasis that blue carbon knowledge is "essential for future climate mitigation strategies" is directly translatable into policy advice. As a result, seagrass ecosystems may begin to feature more prominently in climate-related funding and legislation, similar to mangroves and saltmarshes which are already acknowledged for their carbon value.

Community-led management and co-design models: Project 3.6 demonstrated a successful co-design model that addresses the Indigenous partnership principles of Australia's National Environmental Science Program. and provides a case study for how to effectively engage Aboriginal Rangers and communities into research to achieve dual outcomes. The fact that the project delivered on-ground training, and shared governance of data is a powerful example for future NESP projects and other environmental programs. It supports policies that call for greater Indigenous participation in managing Country and highlights the capacity benefits of doing so. From a community perspective, the project's legacy (trained rangers and an elevated profile of seagrass importance) means the Tasmanian Aboriginal community is better equipped to self-manage their Sea Country. This aligns with the goals of Indigenous Protected Areas and ranger programs, which are to empower Aboriginal people as custodians the custodians of their Country. By clearly documenting how the community was involved and benefited, the project can inform funding bodies that such collaborations yield long-lasting stewardship advantages, not just short-term research outputs.

6.5 Future research and recommendations

Looking ahead, a key limitation of this study was the depth constraint, with data collection limited to approximately 30 m. The use of next-generation multibeam sonar systems for shallow-water mapping could offer significant improvements for future seagrass mapping efforts. These systems provide higher-resolution topographic and textural information compared to the airborne bathymetric LiDAR used in this study. Incorporating multibeam sonar would be particularly valuable for enhancing the mapping of seagrass in estuarine environments and for more accurately validating the extent of deep-water seagrass margins.

In particular, 3D sidescan sonar offers a significant advancement in underwater mapping by combining the strengths of traditional multibeam sonar and interferometric systems. It utilizes beamforming techniques to achieve high-precision bathymetric data, similar to multibeam systems, while also incorporating the wide swath coverage characteristic of interferometric sonar (some systems up to 16 times water depth). This integration allows for efficient and detailed mapping of the seafloor, capturing both accurate depth measurements and high-resolution backscatter and sidescan imagery over large areas. As a result, 3D sidescan sonar enhances the quality and speed of seabed surveys, making it a valuable tool for applications such as seagrass mapping with portability of compact systems with integrated inertia systems allowing for rapid deployment on vessels of opportunity.

Particularly for carbon estimates, bed thickness in species such as *Posidonia* is an important characteristic to help define estimates. Sub bottom profile systems have been shown to be effective in measuring mat thickness of *P. oceanica* in the mediterranean (Iacono et al., 2008) The gradation between plant material (e.g rhizomes) detritus may provide enough contrast of impedance to generate neat seismic reflectors to enable separation from the underlying sediment to provide thickness estimates. Acoustic technologies provide a non-destructive method in the study of seagrasses, bringing new insights on carbon budgets and temporal evolution (Malerba et al., 2023).

To obtain an improved and comprehensive understanding of fish assemblages and association with benthic communities we propose to use a variety of sampling methodologies to achieve this objective, namely 1) baited remote underwater stereo-video systems (stereo-BRUVs); 2) environmental DNA. Stereo-BRUV technique as it is known to be an effective method for surveying fishery target species that are often carnivorous species and therefore attracted to bait (Harvey et al., 2013; Mallet et al., 2014; NESP Operating Manual). Environmental DNA (eDNA) refers to all the genetic material that can be recovered from an environmental sample, such as water, soil, or air. Free-floating DNA is ubiquitous in the marine environment and, in combination with recent advances in DNA sequencing technology, can provide valuable information on the diversity of organisms present in any given area, and has been shown to be a useful tool for baseline biodiversity assessments and long-term monitoring across a range of marine environments (e.g., Thomsen and Willerslec, 2015; Truelove et al., 2019; Hwanhwi et al., 2020).

Reprocessing bathymetric LiDAR data to enable normalized intensity offers significant benefits for seagrass mapping and monitoring. By correcting for variables such as water column attenuation, sensor angle, and depth, normalized intensity provides a more consistent and comparable measure of the seabed reflectance. This enhances the ability to distinguish

seagrass from surrounding substrates, improving classification accuracy and reducing false detections. Additionally, it supports long-term monitoring by enabling consistent comparison across different survey times and locations. Overall, normalised intensity data enhances the ecological value of bathymetric LiDAR, supporting more effective habitat assessments, conservation planning, and environmental management of seagrass ecosystems. While a normalised product was not provided by the Australian Hydrographic Office for this study, data access and processing options should be explored as it may assist in the separation of seagrass species as well as the discrimination of deeper beds beyond the range of optical satellite technologies due to attenuation of signals. However, it is noted that the complexity of the survey area and multiple LIDAR platforms used may limit the normalisation that can be achieved.

The SAMBUCA (Semi-Analytical Model for Bathymetry, Uncertainty, and Classification Algorithm) algorithm is a powerful tool in satellite remote sensing for aquatic environments, particularly shallow coastal and inland waters. Its application is especially valuable for mapping benthic habitats, such as seagrass beds, by separating the water column signal from the substrate reflectance in multispectral or hyperspectral imagery. SAMBUCA uses physics-based modelling to estimate water depth, bottom type, and water quality parameters simultaneously, enabling more accurate and detailed characterization of underwater habitats. One of its key strengths is the ability to work with data from widely available satellite platforms (e.g., Sentinel-2), making it cost-effective and scalable. For seagrass mapping, this means more consistent and reliable detection across broad spatial extents, improved understanding of habitat distribution, and better support for monitoring changes over time due to natural or anthropogenic impacts.

Sampling sediment carbon in deeper seagrass beds is essential for improving the accuracy of blue carbon estimates and understanding the full extent of carbon storage within these ecosystems. Deeper beds are often underrepresented in carbon assessments due to logistical challenges, yet they can hold significant carbon stocks accumulated over long periods. By including these areas in sampling efforts, researchers can capture a more complete picture of spatial variability in sediment carbon, leading to better-informed carbon accounting and climate mitigation strategies. Additionally, understanding carbon dynamics in deeper beds helps assess their resilience and role in long-term carbon sequestration, contributing to more effective conservation and policy development for seagrass ecosystems.

6.6 Conclusion

In conclusion, the impact of MaC Hub Project 3.6 extends well beyond academia – it is directly shaping environmental management and policy in the Tayaritja region and serves as a model nationally. The project filled critical knowledge gaps in seagrass distribution, species diversity, ecological function, and carbon storage, all of which are foundational for effective conservation and climate action. Equally, it exemplified how cultural collaboration can enhance research outcomes: the inclusion of Pakana Rangers and cultural knowledge not only enriched the science but also ensured the results were consistent with the community's rights and interests. For the Commonwealth Government, these findings offer guidance for protecting blue carbon ecosystems and supporting Aboriginal-led conservation initiatives. For Pakana Rangers and the Tasmanian Aboriginal Community, the project has respected and supported their cultural

knowledge and strengthens their ongoing efforts to protect their Sea Country. Ultimately, the project's comprehensive approach – blending ecological assessment with a cultural partnership – has delivered a set of tools and insights that will inform Indigenous Protected Area planning, biodiversity conservation measures, and climate resilience strategies in Tayaritja Sea Country for years to come.

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Appendix 1.

Supp. Table 1. Global stats for each species showing the number of populations (Pop), Individuals (Ind) Multi locus genotypes (MLG), clonal richness (R=(MLG-1)/(Ind-1), number of SNPs and % missing data, observed heterozygosity (H_O) ± standard error of the mean, expected heterozygosity (H_S) ± standard error of the mean, F_{ST}, F_{IS} and the average % of polymorphic loci within each population. One representative per MLG was used for all statistical analyses.

Species	Pop	Ind	MLG	Clonal Richness	SNPs (missing%)	H_O (±SD)	H_S (±SD)	F_{ST}	F_{IS}	Avg %PolyLoc
A. antartica	4	88	57	0.644	2367 (3.73%)	0.206 (0.131)	0.246 (0.139)	0.057	0.162	80.5%
H. nigricaulis	6	75	66	0.878	734 (16.89%)	0.107 (0.079)	0.147 (0.086)	0.205	0.273	56.6%
P. australis	8	117	49	0.414	712 (16.11%)	0.123 (0.088)	0.232 (0.139)	0.079	0.472	54.0%
Z. mulleri	5	93	48	0.511	1539 (9.09%)	0.119 (0.089)	0.149 (0.097)	0.209	0.203	44.9%

Supp. Table 2. Pairwise F_{ST} for populations of a *Amphibolis antartica*

A. antartica	**Big Dog**	**Deep Bay**	**Duck Island**
Deep Bay	0.0312		
Duck Island	0.0463	0.0250	
Killiecrankie Bay	0.1258	0.1004	0.1148

Supp. Table 3. Pairwise F_{ST} for populations of *Heterozostera nigricaulis*

H. nigricaulis	**Badger Corner**	**Big Dog**	**Deep Bay**	**Duck Island**	**Killiecrankie Bay**
Big Dog	0.1253				
Deep Bay	0.0549	0.1143			
Duck Island	0.1812	0.2289	0.1579		
Killiecrankie Bay	0.4048	0.4492	0.3880	0.4486	
Puncheon Island	0.0691	0.1087	0.0422	0.1740	0.3967

Supp. Table 4. Pairwise F_{ST} for populations of *Posidonia australis*

P. australis	**Badger Corner**	**Deep Bay**	**Duck Island**	**Little Green**	**Long Point**
Deep Bay	0.0907				
Duck Island	0.1230	0.1074			
Little Green	0.0685	0.0237	0.0754		
Long Point	0.0879	0.0844	0.1042	0.0324	
Puncheon Island	0.0685	0.0230	0.0943	0.0229	0.0578

Supp. Table 5. Pairwise F_{ST} for populations of *Zostera mulleri*

Z. mulleri	Badger Corner	Big Dog	Deep Bay
Big Dog	0.1263		
Deep Bay	0.2159	0.2711	
Puncheon Island	0.2145	0.2654	0.1025

Appendix 2.

Report on Phasianotrochus spp. presence and abundance in the Tayaritja sea country region, prepared for the Tasmanian Aboriginal Centre, INC

NESP 3.6: Report on *Phasianotrochus* spp. presence and abundance in the Tayaritja sea country region, prepared for the Tasmanian Aboriginal Centre, INC

Based on the honours thesis work of Sophie Sinclair, 2024.

Prepared by Sophie Sinclair & Elisabeth Strain

March, 2025

CONFIDENTIAL TO THE TASMANIAN ABORIGIONAL CENTRE

Appendix 3.

Standard Operating Procedure Report for mapping seagrass habitat from remotely sensed data.

The NESP Standard Operating Procedure for using optical remote sensing for seagrass mapping in temperate waters can be found at the following location:

Standard Operating Procedure for Seagrass Mapping using Optical Remote Sensing | nesp-sop-seagrass-mapping



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