Prioritising the restoration of marine and coastal ecosystems using ecosystem accounting

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Abstract

Ecosystem accounting is a structured approach to compiling environmental and economic information. While accounts are typically used to compile data on past trends, they have an unrealised capacity to also be used to inform decisions by providing a structured approach to scenario evaluation of potential futures. We used the global standard for ecosystem accounting (System for Environmental Economic Accounting), to examine past trends and potential future restoration options in two large metropolitan bays, where data existed for tidal marshes, mangroves and seagrass. We assessed options for reversing the loss of these ecosystems and although the net benefit varied between sites, we found that if all sites were restored, the overall investment-benefit ratio would be 10.5, resulting from AUD$100 million of ecosystem services from an investment of AUD$8.5 million. This study highlights the advantage of structured approaches to data compilation through ecosystem accounts, and consideration of ecosystem dynamics and response to restoration actions, to inform management decisions.

Introduction

Estuaries and coasts have been the focus of human settlement, industry and resource extraction for millennia (Lotze et al. 2006), largely because of the ecosystem services society derives from the marine and coastal environment, including seafood, timber and transport. These ecosystem goods or services are provided by coastal ecosystems, including mangrove forests, tidal marshes and seagrass meadows (Barbier et al. 2011, de Groot et al. 2012, Yoskowitz and Russell 2014). Despite an understanding of the importance of coastal and marine ecosystems, they continue to decline in both extent and condition, along with many of the services they provide (Davidson 2014, Lucas et al. 2014, Giri et al. 2015, IPBES 2019, de los Santos et al. 2019). This has occurred because government and landowners largely prioritise land use, management and investment based on the financial returns from built or human-produced capital (for example, agricultural production or housing), rather than benefits gained from natural capital (such as coastal and marine ecosystems). In contrast, changes in the services provided by ecosystems are typically not measured or valued in the economy in comparable ways to human-produced capital.

Ecosystem accounting provides a common language for describing and measuring ecosystem contributions to people and the economy so that it can be incorporated into decision making in a similar way to measures of human-produced capital (Obst et al. 2016). The United Nations' System of Environmental-Economic Accounting (SEEA) is an internationally agreed framework that provides a set of accounting principles for compiling information on natural capital (United Nations 2014). Designed to be integrated with the System of National Accounts, the SEEA enables decision-makers to assess the state of the ecosystems that underpin the economy (Eigenraam and Obst 2018). The SEEA Ecosystem Accounting (EA) standard (finalised in 2021) focuses on the state of and trends in ecosystems and the benefits they provide (UNCEEA 2021). Since SEEA's release in 2013, governments have adopted its standardised approach to ecosystem accounting worldwide (Hein et al. 2020).

The motivation of ecosystem accounting is to improve the information used by decision-makers by ensuring it is comparable, consistent and coherent. While much focus is at the national level, applications of ecosystem accounting at local levels can support place-based decision making, at a scale aligned with local management decisions (Keith et al. 2017, Lai et al. 2018, Thornton et al. 2019, Campos et al. 2019). Most examples of application of SEEA so far have focused on compiling data on past trends and current state (Eigenraam et al. 2016, WAVES 2016, Thornton et al. 2019, Dvarskas 2019); while past trends and current state are informative, decision-makers also need information on the potential benefits of different management options, such as which areas to restore or protect, and the most appropriate management options (Adame et al. 2015). Ecosystem accounts have an unrealised capacity to inform decisions about future benefits of ecosystem management, by providing a structured approach to scenario evaluation of management options.

We applied the SEEA EA framework to account for marine and coastal ecosystems at the regional spatial scale in Port Phillip and Western Port in southeast Australia, compiling accounts of present ecosystem state and service flow, and potential future benefits from three management options. While multiple marine and coastal ecosystems exist in the region, we focused on mangroves, tidal marsh and seagrass ecosystems, as several studies have been conducted in the region that assess their ecosystem services (Ewers Lewis et al. 2018, Carnell et al. 2019, Jänes et al. 2020, Huang et al. 2020). Here we present a key advance in the implementation of ecosystem accounting by integrating extent, condition, services and valuation data with estimates of restoration costs to evaluate options for ecosystem restoration. Doing so enabled us to identify priority areas where the net benefit from coastal restoration is high and estimate the total benefits of restoration across all locations.

Methods

Application of the System of Environmental Economic Accounting (SEEA) Ecosystem Accounts (EA) framework

We developed ecosystem accounts for Port Phillip and Western Port using the SEEA EA framework (Eigenraam and Obst 2018, UNCEEA 2021)). Definitions of key components of the accounting framework are provided in the glossary. Our approach comprised five steps: 1) define the geographical scope and ecosystem types being assessed; 2) estimate their current geographic extent; 3) determine ecosystem condition against reference baselines; 4) quantify ecosystem services in physical and monetary terms; and 5) model the benefits of restoring different sites under three hypothetical management scenarios to demonstrate how the accounting information can inform investment.

Define geographical scope and ecosystem types

Geographic scope: Port Phillip (surface area: 1930 km2; mean depth: 12.8 m; Black et al. 1993) and Western Port (surface area: 680 km2; mean depth: 3 m; (CES 2016) are semi-enclosed bays located in Victoria, Australia. They are part of a suite of marine assets under the management of the State of Victoria, including marine ecosystems within the territorial sea baseline (internal waters) and within 3 NM of the territorial sea baseline (coastal waters). Port Phillip and Western Port comprise 25% (18% and 7% respectively) of marine areas under the jurisdiction of the State of Victoria.

Ecosystem types: To identify ecosystem types in Port Phillip and Western Port, we collated existing information and maps of marine and coastal ecosystems and ecological communities in Victoria (Edmunds and Flynn 2018). We aggregated finer-scale classification of our focal ecosystems and matched these with the IUCN Global Ecosystem Typology (Keith et al. 2020), the recommended reference framework for ecosystem accounting (UN Statistical Commission 2021), using expert judgement from knowledge of these ecosystems (Table S1): tidal marshes (MFT1.3 Coastal saltmarshes and reedbeds); mangroves (MFT1.2 Intertidal forests and shrublands); and seagrass (M1.1Seagrass meadows) (Table S1).

The mangroves in Victoria occur at the highest latitude anywhere in the world and are comprised of one species *Avicenia marina subsp. australasica*. Here, *A. marina* only reaches a maximum height of 3.9 m, compared to 13 m in more tropical climates (Navarro et al. 2020). Over the last 50 years mangroves have been expanding shoreward into tidal marsh ecosystems and is correlated with increasing temperatures (Whitt et al. 2020). In comparison, the tidal marshes are floristically and structurally diverse with over 150 species, making up over 10 Ecological Vegetation Communities, and includes small herbs (eg. *Sarcocornia quinqueflora*), sedge tussock grasses (eg. *Austrostipa stipoides*) and large shrubby dicots (eg. *Tectocornia arbuscula*) (1.5 - 3.0 m tall) (Boon et al. 2011).The seagrasses in Port Phillip and Western Port on soft sediment are comprised of *Heterozostera nigracaulis* in the sub-tidal zone (and in smaller abundances by *Halophila* ovalis) and *Zostera muelleri* in the intertidal (Blake and Ball 2001). The seagrass *Amphibolis antartica* inhabits more wave exposed sub-tidal reef and is therefore classified as part of the sub-tidal reef ecosystem (Blake and Ball 2001). The seagrass *Lepilaena* spp. exists in shallow pools among saltmarsh ecosystems and is classified as part of the saltmarsh ecosystem (Boon et al. 2011).

Extent of ecosystems

We collated the most recent information on the current extent of the different ecosystem types (Edmunds and Flynn 2018, Sinclair and Boon 2012) of marine and coastal ecosystems (Sinclair and Boon 2012, Wilkinson et al. 2016).

Condition of ecosystems

To determine the condition of the identified ecosystems, we used principles from the IUCN Red List of Ecosystems (RLE) (Keith et al. 2013), the global standard for assessing the risk of ecosystem collapse. In the Red List of Ecosystems, a conceptual model of the key ecosystem features, processes and threats is required for each ecosystem type to inform the assessment (Bland et al. 2017). We created conceptual models for seagrass, and tidal marsh and mangrove ecosystems to guide the variables chosen to assess condition, and to underpin the modelled effect of ecosystem management (Figure S2). Through the conceptual models we developed, we identified altered hydrology through artificial levees and grazing of saltmarsh and mangrove areas as key threats to these ecosystems driving their degradation or collapse (Sinclair and Boon 2012). Coastal levees dramatically alter hydrology, species abundance and diversity, and thus ecosystem processes and services, and are generally used to convert mangroves and tidal marshes to other land uses such as aquaculture or agriculture (McFalls et al. 2010, Richards and Friess 2016, Adame et al. 2019, Abbott et al. 2020). Cattle and sheep grazing and trampling directly impact plant structure and diversity in mangroves and tidal marshes, which also has flow on effects to ecosystem services (Andresen et al. 1990, Hoppe-Speer and Adams 2015, Minchinton et al. 2019).

Field based data on condition were unavailable for mangroves and tidal marshes in Port Phillip and Western Port, so we used a threat-based approach to assess condition, based on causal relationships between threats and condition in our conceptual model (see Table S2)(Prahalad 2014a, Grantham et al. 2020). We used information about where these coastal levee walls are present and where agricultural grazing occurs via land-use data. To incorporate the full range of condition (ie. collapsed ecosystem state), we combined the threat data with information about the historical extent (pre-1750) of tidal marsh and mangroves (Boon et al. 2011). Sites were considered collapsed if they were historically mangrove or tidal marsh but are currently another ecosystem type. Sites that are tidal marsh or mangroves were then classified by combinations of two key threats: presence of agriculture (primarily grazing; Low disturbance); and the presence of levees (restricting hydrology; Medium disturbance) or both threats present (High disturbance) (Figure 1, Table S2). Low-Natural ecosystems exist in areas designated for conservation or other low-impact land uses, where the natural assemblages and processes of the ecosystems remain largely intact (Boon et al. 2011, Prahalad 2014b, McNellie et al. 2020).

For seagrasses, our condition assessment was based on knowledge of drivers of temporal variation in the extent of the two dominant seagrasses *Zostera* muelleri and *Heterozostera nigracaulis*, which leads to some seagrass meadows being stable, while others can be relatively ephemeral (Ball et al. 2014, Connolly et al. 2018). In Port Phillip and Western Port, these patterns of persistence of seagrass meadows are driven by local conditions (ephemeral sites are generally more exposed to waves, and in sandy, nutrient-limited environments), with temporal variation driven by climatic variability, poor water quality and nutrient inputs (Bulthuis 1983, Ball et al. 2014, Hirst et al. 2016). In addition to this understanding of seagrass dynamics, the age of seagrass meadows impacts both the abundance and biodiversity of associated species in seagrass meadows, as well as the ecosystem services provided by seagrass meadows (Duarte et al. 2013, Marbà et al. 2015, Orth et al. 2020). We combined this information with an understanding of the resilience and recovery of these seagrass meadows to disturbance and change (Jenkins et al. 2015, Smith et al. 2016) to develop our condition assessment. This led us to focusing on using historical (1960s -2010s) seagrass extent maps to classify the age, stability and last presence of seagrass patches, as opposed to metrics like seagrass length or density, to define current condition of seagrass meadows across Port Phillip and Western Port (Table S3). Sites where seagrass has not been present for more than two surveys (10-20 years) are classified as collapsed, while seagrass that was not present in the most recent mapping survey (0-10 years) has been classified as High disturbance. Where seagrass is present, the age of the seagrass meadows 1-10 years (Medium-High), 10-20 years (Low-Medium) and >20 years (Low-Natural) was then used to classify seagrass condition. Data was available for seagrass extent in Port Phillip from 1966 (Lynch 1966), 1981 (Blake and Ball 2001), 2000 (Blake and Ball 2001) and 2011 (Jenkins et al. 2015), and data for Western Port which covered the period from 1973 - 2014 (Wilkinson et al. 2016).

Physical and monetary valuation of ecosystem services

We collated and synthesized existing data for ecosystem services from published papers and reports. Due to data limitations, this component of the study focuses on five ecosystem services: one provisioning service (fish), one cultural service (recreational fishing), and three regulating services (carbon and nitrogen sequestration, and coastal protection). This represents only a subset of all possible services (e.g. recreational activities such as nature watching, amenity services, education, scientific and research services, spiritual, symbolic and artistic services, and ecosystem and species appreciation services); as a result, the combined value of these ecosystems is underestimated).

The data for ecosystem services comes from studies conducted as part of the Mapping Ocean Wealth Australia program (Carnell et al. 2019) and other relevant studies. Estimates of physical ecosystem services were compiled by attributing physical quantities (e.g tonnes, number of visitors) of ecosystem services for each ecosystem type within the study area. Attribution required either disaggregation or aggregation of input data, depending on the resolution and type of the data; note these data are independent of the data in the extent and condition accounts.

Provisioning services

Fish provisioning services

Fish provisioning services are the ecosystem contributions to the survival and growth (via their diet) of fish that are subsequently harvested by people; here we focussed on the harvest or wild-capture of fish and used the approach of (Taylor et al. 2018) for the valuation methodology, based on stable isotopes. Fish and invertebrates use coastal and marine ecosystems as both refuge/habitat and for food. Stable isotopes can be used to estimate the contributions of different primary producers to the diet of a species (Taylor et al. 2018). This approach recognises the fact that these species operate in an ecosystem, and that even if a fish doesn't directly consume a primary producer (eg. seagrass), consuming a herbivore makes the predator indirectly reliant on the primary producer for its diet.

We assessed fish provisioning services in three steps (adapted from (Taylor et al. 2018): 1) We estimated the proportion of fish diet originating from coastal ecosystems with data from (Jänes et al. 2020); 2). This was then combined with annual catch values of commercially relevant species to determine how much of the commercial catch relied on the different ecosystems for their diet. 3) Data on recreational fish catch came from estimates in (Henry and Lyle 2003). Here, this data was for recreational catch from select species for the entire state of Victoria. To downscale this for use in this assessment, we combined the catch with the proportional contributions from (Jänes et al. 2020) and then calculated a value (tonnes of fish) per hectare based on the area of these ecosystems across Victoria.

To value the commercial catch, we combined the estimated catch per year resulting from the different ecosystems with the price per kg that commercial fishers receive. The most recent 3 years of commercial catch data (DPI 2012) was extracted and corrected with Consumer Price Inflation (CPI) index to calculate the price per kg of the commercial species (Reserve Bank of Australia, n.d.). We did not estimate the contribution of the ecosystem to the benefit of commercial and recreational catch due to missing information on production costs (for example, asset depreciation, fuel).

Cultural services

Recreational related services (fishing)

Recreation-related services are the ecosystem contributions, particularly through the biophysical characteristics and qualities of ecosystems that enable people to use and enjoy the environment through physical and experiential interactions with the environment ((UNCEEA 2021). In the case of recreational fishing, coastal and marine ecosystems provide fish and invertebrates refuge or places to feed, and thereby increasing the catch of fisheries species by recreational fishers in areas with more seagrass (Huang et al. 2020).

Estimates of the physical services provided by seagrass ecosystems to recreational fishing were based on the number of trips undertaken by fishers each year in Port Phillip and Western Port (Ernst & Young 2015), combined with information from (Huang et al. 2020), on where people choose to fish. To determine the monetary services derived while recreational fishing, our analysis was based on the study by (Huang et al. 2020). Here, the recreational value is based on how willing people were to travel further from home to fish, which is a measure of how much they value a location. The economic value associated with seagrass can be estimated through travel costs and the time it takes to get there. The costs are used to estimate recreational value in dollars. This is a non-market measure of value, based on people's preferences about how and where they spend their time. It is not, for example, how much they spend on fishing gear, though other studies (Ernst & Young 2015) show that equipment expenditure is also substantial.

Regulating services

Global climate regulation services

Global climate regulation services are the ecosystem contributions to regulating concentrations of gasses in the atmosphere, primarily through the retention of carbon in ecosystems (UNCEEA 2021). Most of the carbon captured by tidal marshes, mangroves and seagrass is stored in the sediments trapped by their complex root systems and to a smaller degree in their plant biomass. We used soil carbon sequestration data collected in Port Phillip and Western Port from seagrass meadows, mangroves and tidal marshes (*Carnell unpublished data*). Sediment cores were collected in tidalmarsh, mangrove and seagrass ecosystems from three sites in each of Port Phillip and Western Port. Samples were then sliced every 1 cm for ^{210}Pb age dating (by alpha spectrometry following (Sanchez-Cabeza et al. 1998)) to calculate sediment mass accumulation rates (Arias-Ortiz et al. 2018).These cores were then matched with cores for which percentage of organic carbon was determined, to estimate carbon sequestration rate (Howard et al. 2014). The values used per ecosystem types are the averages of samples for that ecosystem type across the sites sampled in each bay. The Australian Carbon Credit Unit price of AUD$13.87 (average over the years 2018-2020) was used to value carbon sequestration.

Water purification service *(Nitrogen)*

Water purification services are the ecosystem contributions to the restoration and maintenance of the chemical condition of surface water and groundwater bodies through the breakdown and storage of pollutants by ecosystem components that mitigates the harmful effects of the pollutants on human use or health. Coastal wetlands are particularly efficient at processing and storing nitrogen (a pollutant which can cause harmful algal blooms) form the water column into plant materials and into the soils, in a similar way to carbon sequestration. We utilised the above-described data on soil carbon sequestration and used the matching % Nitrogen values to calculate soil Nitrogen sequestration. We used the nitrogen sequestration value from Liekens et al. (2012) where nitrogen was valued at Euro32/kg. The conversion to AUD$ 2020 value is AUD$56/kg.

Coastal protection services

Coastal protection services are the ecosystem contributions of physical structures in the seascape, for instance coral reefs, sand banks, dunes or mangrove trees along the shore, in attenuating waves and preventing erosion, and thus mitigating the impacts of tidal surges or storms on local communities (WAVES 2016, UNCEEA 2021). We used data produced on the coastal protection value of mangroves, tidal marsh and seagrass (Carnell et al. 2019) calculated utilising the InVEST Coastal protection toolbox (Natural Capital Project) and coastal protection models (Arkema et al. 2013). The InVEST toolbox provides a user-friendly first-pass assessment of the coastal protection potential by coastal wetlands at regional to national scales. Here, the models require inputs to rank seven coastal exposure metrics and then combine this with a ranking of the coastal protection provided by each ecosystem. Using the InVEST model, the number of people protected was calculated based on a one in ten-year coastal inundation event. The value of protected property was calculated based on the cost savings associated with a one in ten-year coastal inundation event.

Assessing future management options

We assessed three management actions that could restore tidal marsh and mangrove ecosystems where they historically existed but have been lost (‘Collapsed’ state in step 3), to rank options based on restoration costs and the resultant benefits (increase in ecosystem services' physical and monetary value) from restoring the ecosystem. We did not consider options for seagrass restoration, as both the methods for seagrass restoration in the region are still being developed and the cause of seagrass decline is still being addressed, the success of restoration efforts are likely to be low (Tan et al. 2020). We also did not include restoration of degraded tidal marsh and mangrove areas, because data and models are currently insufficient for estimating return-on-investment for ecosystem condition and ecosystem service provision for these ecosystems (Carnell et al. 2019).

As discussed in section 3, two of the primary threats to mangrove and saltmarsh are 1) agricultural grazing, which slowly converted coastal wetlands to pasture, and 2) installation of coastal levees leading to the loss of tidal flow, to convert to another land use such as housing. Often these two actions were done simultaneously to aid conversion to agricultural use. Thus, we considered two restoration activities common in the region, which can be applied on their own or together to restore tidal marsh or mangroves: fencing to protect areas from grazing or other anthropogenic impacts; and removal of levees to restore tidal flow (Figure 1) (Boon et al. 2011).

We selected potential areas for restoration action by:

1) Fencing: we identified areas of collapsed condition tidal marsh and mangroves currently being used for agriculture or grazing according to the Victorian Land Use Information System (a system for spatial data on land use in Victoria). We then assumed that installing a fence around the perimeter of this area would remove grazing and allow the plant community to recover (Boon et al. 2011).

2) Levee removal: we identified areas of collapsed tidal marsh and mangroves where the original ecosystem extent/condition (in 1750 terms) had been modified by erecting a levee and restricting tidal flow. We assumed that the original ecosystem had only been modified in this way if a levee feature intersected it. Therefore, we assumed that removing the levee would return tidal flow to the ecosystems, after which the characteristic species and ecological processes would also return (Table S4) (Abbott et al. 2020).

3) Fencing and levee removal: We identified locations of collapsed tidal marsh or mangrove that are currently used for agriculture or grazing (scenario 1), and have a levee present (scenario 2), and thus require both fencing and levee removal to restore the ecological processes and the ecosystem.

We calculated net benefit as (Net benefit \$AUD = $\sum(F, C, N)$ - Costs), where F is the fisheries value, C is the carbon sequestration and N is the nitrogen sequestration value (in dollars). We calculated these values from the three management scenarios in three steps: first, we defined the potential restoration areas based on historical extent of tidal marsh and mangroves and current land use parcels, second we estimated the capital expenditure associated with restoring the area (Table S5), and third we estimated the return associated with restoring the area based on a subset of ecosystem services defined in the account (steps 4 & 5 in Fig. 1) (Table S6).

Fencing installation costs were obtained from tidal marsh fencing projects undertaken in the region, and the costs associated with levee removal were obtained from different government management agencies in the region (Table S5) This approach is preferred to global numbers (e.g. (Bayraktarov et al. 2016)) as it allows for local wages and material costs to be included. Costs were a function of the restoration action required and site-specific information such as area (which influenced fencing perimeter and length of the levee to be removed) (see Methods in supplementary material). We did not model how the ecosystem extent could be influenced by sea level rise in the future.

We estimated the benefit associated with the investment by projecting the change in the monetary value of ecosystem services derived from step 6 (Table S6). The monetary value of the multiple ecosystem services was summed by restoration area *i*. We applied a 3 per cent discount rate to determine the net present value of 50 years of flows from the restored asset. Sensitivity analysis is provided in Table S7 across 1, 3, 5, 7 and 11 per cent discount rates and 20, 50 and 100 year time horizons. We projected the yearly returns under each time horizon using several linear and nonlinear functions that relate the restoration trajectory with ecosystem service provision. The function used differed depending on the ecosystem service (Figure S1). We do not project changes to ecosystem service quantities that may depend on other exogenous factors, such as population growth and climate change. This demonstration is not a comprehensive benefit-cost analysis as we do not estimate the opportunity cost of restoration; all benefits are relative to a current baseline.

Results

Current accounts

Extent

In 2011, seagrass, tidal marsh, and mangroves make up 6%, 2% and 1% of the marine and coastal ecosystems by area in Port Phillip and Western Port bays (Figure 2, Table S8). There has been an estimated net loss (or collapse) of 194 ha (10%) of mangroves and 2,753 ha (34%) of tidal marshes across Port Phillip and Western Port since European colonisation (Figure 3).

Condition

In addition to the collapse of 10% of mangroves and 34% of tidal marshes, 2 ha (0.10%) of mangroves and 559 ha (7%) of tidal marshes were in the High degraded condition, having both bund walls and agricultural land use where these ecosystems were present. Another 80 ha (4%) of mangroves and 1,011 ha (12%) of tidal marsh were in Medium disturbance condition with bund walls impacting these ecosystems, and 87 ha (4%) and 865 ha (11%) were in Low categories with grazing/agriculture present. In comparison, 1,585 ha (81%) of mangroves, but only 2,788 ha (35%) of tidal marshes are in Natural condition.

Utilising the seagrass mapping data available, we found that only 7031 ha or 15% of seagrass meadows have been stable since the 1960's and classified as Low-Natural disturbance (Figure 3). Conversely, 27,080 ha (or 58%) of seagrass meadows have collapsed over the same time period. There is 2178 ha (or 4.7%) of seagrass meadows that have only died back recently (becoming MF1.7 Subtidal sand beds or MF1.8 Subtidal mud plains) and classified as High disturbance. The remaining 10,283 ha (or 22%) are meadows aged between 1-20 years old and classified as Low-Medium and Medium-High.

Physical and monetary value of ecosystem services

The combined physical and monetary value across all categories from these ecosystem services from tidal marsh, mangrove and seagrass ecosystems in Port Phillip and Western Port were calculated at AUD$89.7 million per year (Table 1).

Provisioning

Fish provisioning services

Seagrass, mangroves and tidal marsh ecosystems produce fish that are commercially or recreationally caught at 91 tonnes and 287 tonnes each year respectively. The catch of commercial and recreational fish across the two bays is valued at AUD$2.35 million per year across seagrass meadows, mangroves and tidal marshes.

Cultural services

Recreational related services (fishing)

Seagrass provides enhanced recreational fishing opportunities that see fishers travel specifically to particular areas, culminating in an estimated 2.2 million trips per year.

The monetary value provided by seagrass to recreational fishing, based on the expected benefits of catch and the time and travel costs to get to that location (Table 1). With an estimated 1.42 million fishing trips in Port Phillip Bay and 0.79 million in Western Port, this resulted in a total of AUD$33.1 million each year across the two bays.

Regulating services

Global climate regulation services

Seagrass, mangrove and tidal marsh ecosystems store an estimated 52,339 tonnes of carbon and 836 tonnes of nitrogen each year across Port Phillip and Western Port (Table 1). Carbon sequestration is valued at a combined AUD$0.72 million per year, with the highest value coming from seagrass storing 28,879 tonnes of CO2e per year, valued at AUD$0.4 million per year.

Water purification service *(Nitrogen)*

Seagrass, mangrove and tidal marsh ecosystems store an estimated 836 tonnes of nitrogen each year across Port Phillip and Western Port (Table 1). Nitrogen sequestration had the highest value of all the services (combined value of AUD$46.8 million), with seagrass and tidal marsh contributing AUD$23.7 million and AUD$21.7 million respectively.

Coastal protection services

These ecosystems protect an estimated 7830 people from coastal inundation from one in ten-year storm events. Tidal marshes were valued at AUD$3.38 million from their coastal protection value in Port Phillip, accounting for 50% of the total value of this service across the two bays and ecosystems (Table 1). This is followed by seagrasses at AUD$1.42 million and mangroves at AUD$0.88 million per year in Port Phillip. In Western Port, the three ecosystems combined add up to a value of AUD$1.01 million per year to coastal protection or 15% of the value across the two bays.

Table 1. The ecosystem services and values assessed on a per year basis for Port Phillip and Western Port. *associated with a one in ten-year storm surge event

		Port Phillip			Western Port		
Ecosystem type		Seagrass	Mangrove	Tidal marsh	Seagrass	Mangrove	Tidal marsh
	hectares	6,788	4	2,492	10,505	1,726	2,672
Soil carbon sequestration	tonnes yr^{-1}	11,336	26	5,975	17,543	11,046	6,413
	$\$yr^{-1}$	157,230	355	82,873	243,326	153,214	88,946
rogen sequestration	tonnes yr^{-1}	193	0	72	230	25	316
	$\$yr^{-1}$	10,835,423	3229	4,012,702	12,905,063	1,393,461	17,690,222
oastal protection*	People	1464	125	3634	266	1,269	1,072
	$E(\$yr^{-1})$	1,425,140	876,764	3,375,904	75,213	515,552	415,195
mmercial fisheries (catch)	tonnes yr^{-1}	30	0	2	46	11	2
	$\$yr^{-1}$	264,732	218	17,693	409,695	93,894	18,971
creational fisheries (catch)	tonnes yr^{-1}	97	0	4	160	19	7
	$\$yr^{-1}$	522,523	240	20,901	860,161	103,680	40,394
ecreational fishing time and travel	trips yr^{-1}	1,429,864	-	-	794340	-	-
	$\$yr^{-1}$	7,617,361	-	-	25,509,069	-	-
Total	**$\$yr^{-1}$**	20,822,409	880,806	7,510,073	40,002,527	2,259,801	18,253,728
Average	**$\$ha^{-1}\ yr^{-1}$**	3,068	220,202	3,014	3,808	1,309	6,831

Future management options & impacts on ecosystem services

We assessed the benefit of restoration activities by assessing the change in the value of ecosystem services and costs to restore collapsed tidal marsh and mangrove ecosystems via installing fencing and/or levee removal across 114 locations (or 1,689 ha) (Figure 4, Table 2). This analysis showed that if all areas where these ecosystems previously existed were restored (where current land use allows) for a total cost of AUD$8.5 million, a combined net benefit of AUD$137.5 million (at 3% discount rate) would be achieved (Table 2). The net benefit varied between sites and restoration methods (Figure 4). Levee removal alone provided the highest net benefit, with values ranging between AUD$0.025 million to AUD$33.5 million per site, with the highest value sites along the western shoreline of Port Phillip Bay (Figure 4). These areas are predominantly used as sewage treatment ponds or abandoned salt production ponds awaiting restoration or management.

If areas which currently experience cattle grazing, but were previously tidal marshes or mangroves, were fenced, net benefits would range from AUD$0.011 million to AUD$3.3 million per site. While the overall costs for fencing were relatively cheaper than for removing levees, the patches available for this type of restoration were smaller, meaning the net benefit generally wasn’t as high. However, some large areas near Swan Bay in SW Port Phillip Bay show a larger net benefit, compared to smaller sites in Western Port (Figure 6b).

Some sites required both fencing of land and removal of levees to restore them. Needing both restoration methods meant that costs were higher, however some large sites in Port Phillip, returned a net benefit of up to AUD$3.3 million, and in Western Port resulted in a net benefit of between AUD$0.042 million to AUD$8.2 million per site (Figure 6c).

Table 2. Summary of results from analysis of the different restoration options, using a 3% discount rate over a 50-year time horizon, with the 1% and 7% discount rates shown for Net Benefit. The table displays the number of available sites for action across Port Phillip and Western Port, the total area, total investment, total benefit, net benefit (Investment - Benefit), and the overall investment-benefit ratio for the scenario and region. Net benefit does not include a measure of opportunity cost

nario	Bay	# sites	Total Area (ha)	Investment (AUD$million)	Total Benefit (AUD$ million)	Net Benefit (AUD$million)* [1-7% discount]	Investment benefit ratio
ncing	PPB	31	178	1.40	13.3	11.9 [18.7 – 5.77]	8.5
	WPB	5	8	0.18	0.98	0.80 [1.30 – 0.35]	4.3
vee removal	PPB	23	1040	3.50	77.6	74.1 [114 – 38.4]	21.2
	WPB	15	82	0.49	13.9	13.4 [20.6 – 7.04]	27.7
ncing & levee noval	PPB	13	262	1.11	19.5	18.4 [28.4 – 9.43]	16.5
	WPB	27	129	1.79	20.6	18.8 [29.4 – 9.34]	10.5
al	Both	114	1689	8.48	146	137 [212 – 70.4]	16.2

Discussion

Ecosystem Accounting frameworks were designed to enable integration of environmental data into national accounts to help inform management decisions. This study used the recently developed SEEA EA framework to compile ecosystem accounts for marine and coastal ecosystems and estimate the net benefits of investment (monetary value of ecosystem services minus the costs) from different restoration options of coastal wetlands. Based on the services we assessed, we found that seagrass meadows, mangroves and tidal marshes have an annual value of at least AUD$89.7 million per year across Port Phillip Bay and Western Port. By assessing opportunities to restore degraded mangroves and tidal marshes, we estimate that with a cost of $8.5 million, almost 1700 ha could be rehabilitated in the region, resulting in $146 million of benefits after 50 years. While the ecosystem services considered here represent just a small subset of the benefits these ecosystems provide (de Groot et al. 2012, Costanza et al. 2014), this results in a net positive return on investment ratio of 16.2 across all the possible sites in Port Phillip and Western Port bays.

Because we took a spatially explicit approach to determine restoration costs, the substantial variation in net benefit was revealed, ranging from -$11,000 to $33.5 million. This was primarily due to the nature of the costs associated with restoration, as larger return-on-investment occurred where the perimeter (cost) to area ratio was the lowest (as larger area meant larger benefit). This tended to favour larger sites over smaller ones, and this also tended to be at sites where removing levees would rehabilitate relatively large areas. This information is vital as government and non-government organisations seek to maximise return on investment into the environment and is timely as we enter the UN decade on ecosystem restoration (Waltham et al. 2020).

This study supports the growing realisation of the wealth gained from restoring or protecting natural ecosystems, making a compelling business case for their restoration in addition to biodiversity values (Bullock et al. 2011, Keith et al. 2017, Shimamoto et al. 2018, Orth et al. 2020). Overwhelmingly, case studies that have investigated multiple ecosystem services delivered have found that over reasonable time-frames, the ecosystem service benefits outweigh the costs of management action (de Groot et al. 2013, Cziesielski et al. 2021). The spatial variability in costs and benefits seen here and in other studies also means that this information can be used to prioritise areas for management when funding is limited (Adame et al. 2015, Atkinson et al. 2016). Several approaches exist to combine information on ecosystem services for prioritising decisions (Strassburg et al. 2020, Sala et al. 2021), but there are limitations to prioritising sites for restoration or protection utilising ecosystem services, particularly if only one or few ecosystem services are prioritised (Possingham et al. 2015, Schröter and Remme 2016, Atkinson et al. 2016). The standardised framework of the SEEA EA enables a consistent approach to conduct these analyses that is repeatable in the future.

The five-step approach to Ecosystem Accounting and future scenario analysis developed and implemented here demonstrates an underutilised potential of the Ecosystem Accounting framework in guiding future management of ecosystem assets based on economic and financial analysis (Keith et al. 2017). As we pursue a green recovery, systems that enable the consistent assessment of investment decisions across time and space, both at a local and national level, can contribute to more efficient, effective and transparent decision making (Barbier 2013, Obst et al. 2016, Hein et al. 2020). While there are examples of scenario evaluations of the benefits of management actions (Atkinson et al. 2016, Weijerman et al. 2018, Strassburg et al. 2020), the SEEA EA provides a common framework for comparably assessing site-level investment choices (Keith et al. 2017). It has an added benefit of allowing managers to take stock of current ecosystem accounts, before then making decisions about the future. As more nations begin to implement the SEEA (Hein et al. 2020), we recommend an increasing focus on utilising the structured approach of SEEA EA in informing management and investment decisions.

The need to restore these vital coastal and marine ecosystems is due to their degradation and loss over the last 50–200 years (Murdiyarso et al. 2015, Richards and Friess 2016, de los Santos et al. 2019). The loss of these ecosystems has likely had a large impact on resultant ecosystem services (Millenium Ecosystem Assessment 2005). However, in our case study, it was difficult to estimate this historical loss of ecosystem services for two reasons: 1) we did not have information on what year mangrove and saltmarsh ecosystems were lost, to calculate the loss of ecosystem services since then 2) Services like coastal protection do not change linearly with increasing or decreasing ecosystem extent or condition, so a per hectare value is difficult to calculate without re-running models based on different historical ecosystem distributions. Despite this, we chose to utilise the accounts to focus on producing future management options (Keith et al. 2017). This is most relevant to managers seeking to understand what decisions they can make today to rectify some of the historical losses.

This case study is a first attempt at blending marine and coastal economic and environmental information in the SEEA EA framework for investment decisions. If policymakers find value in this type of analysis, then there is a need to upscale the collection of information to best inform ecosystem accounts. This is because, while information existed to be able to complete this case study, there are still gaps in the information available (e.g. ecosystem service data for other ecosystems), and in some instances, the data quality was not ideal (e.g. Lack of sufficient survey data to inform condition assessments). We used the approach from the IUCN Red List of Ecosystems assessments to develop conceptual models to identify key threats to mangrove and tidal marsh ecosystems on which we based our assessment of condition (Keith et al. 2013, Lee et al. 2021). Similar to the SEEA, Red List assessments also require information on a change in the extent and ecosystem properties (condition). Given the development of both of these frameworks at the international level, finding a way to combine or utilise information between the Red List of Ecosystem assessments and SEEA would be advantageous for both programs and their implementation. As more and more nations sign up to utilising the SEEA framework (Hein et al. 2020), this should encourage the provision of resources for monitoring programs that encompass and align with the framework.

The case study and approach presented here in using SEEA EA to inform future management options, denotes a significant step forward in EA implementation for coastal ecosystems, however, there are a number of ways this can still be improved upon in the future. The data available on ecosystem services were limited to relatively natural ecosystems, with sometimes little spatial resolution. Here, we could make this work as our coarse approach for assessing the condition of tidal marshes, mangroves and seagrass matched the coarse nature of the data relating the condition of ecosystems to ecosystem services. However, in many instances, ecosystem service data with any sort of spatial resolution is still limited, or if it does exist, represents an average across many sites of likely varying condition (Blandon and Zu Ermgassen 2014, Gaylard et al. 2020). Of those that do relate ecosystem services to variables, these are often environmental (e.g. Elevation, temperature) rather than measures of the condition of the ecosystem (Sanderman et al. 2018, Ewers Lewis et al. 2020, Jänes et al. 2021, Young et al. 2021). We also recognise that estimation of the net benefit does not consider an estimate of the opportunity cost of restoration, that is, the benefit of the current land-use foregone by restoring the ecosystem. There is scope to improve this analysis in the future by including a counterfactual and expanding the ecosystem services included in the study.

The United Nations System of Environmental-Economic Accounting provides a standardised approach to environmental measurement and a framework that can reach scale for integrating the environment in decision making. The increasing number of SEEA case studies in such a short space of time highlights this as a growing area of interest to conservation and management of natural ecosystems (Hein et al. 2020). This application allows some of the many ecosystem services to be assessed in the same values-based system as more traditional economic measures of value and productivity that directly contribute to

GDP. This approach aims to turn the tide from the traditional approach of degradation of natural ecosystems for the benefit of humanity, to incorporating the many services and benefits of these ecosystems into national accounting. This enables the tradeoffs of degradation of ecosystems to be assessed against opportunities for restoration and protection to ensure we protect biodiversity and improve human wellbeing at the same time.

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Declarations

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Author Contributions

P.C., E.N., P.M., D.I., C.G., S.R., C.O., M.E., R.M. designed the project. P.M., E.N., D.I., C.G acquired funding for the project, P.C., R.M., M.Y., H.J., J.K., collected and collated data, P.C. and R.M. performed the calculations and spatial analyses. The manuscript was written by P.C. and R.M, with contributions from all the authors

Competing Interests

The authors declare no competing financial interests.

Figures

Figure 1

The two management actions considered here (both alone and in conjunction) to restore tidal marsh and mangrove ecosystems. **a)** Improving hydrology via levee removal (Image: Paul Carnell). This image shows a coastal levee preventing saltmarsh (right) from inhabiting the land to the left and **b)** Reducing impacts from livestock via fencing. This image shows the difference in saltmarsh cover after the site on the right was protected due to the fence installed running down the middle. The left side shows the currently grazed side (Image: Mel Wartman).

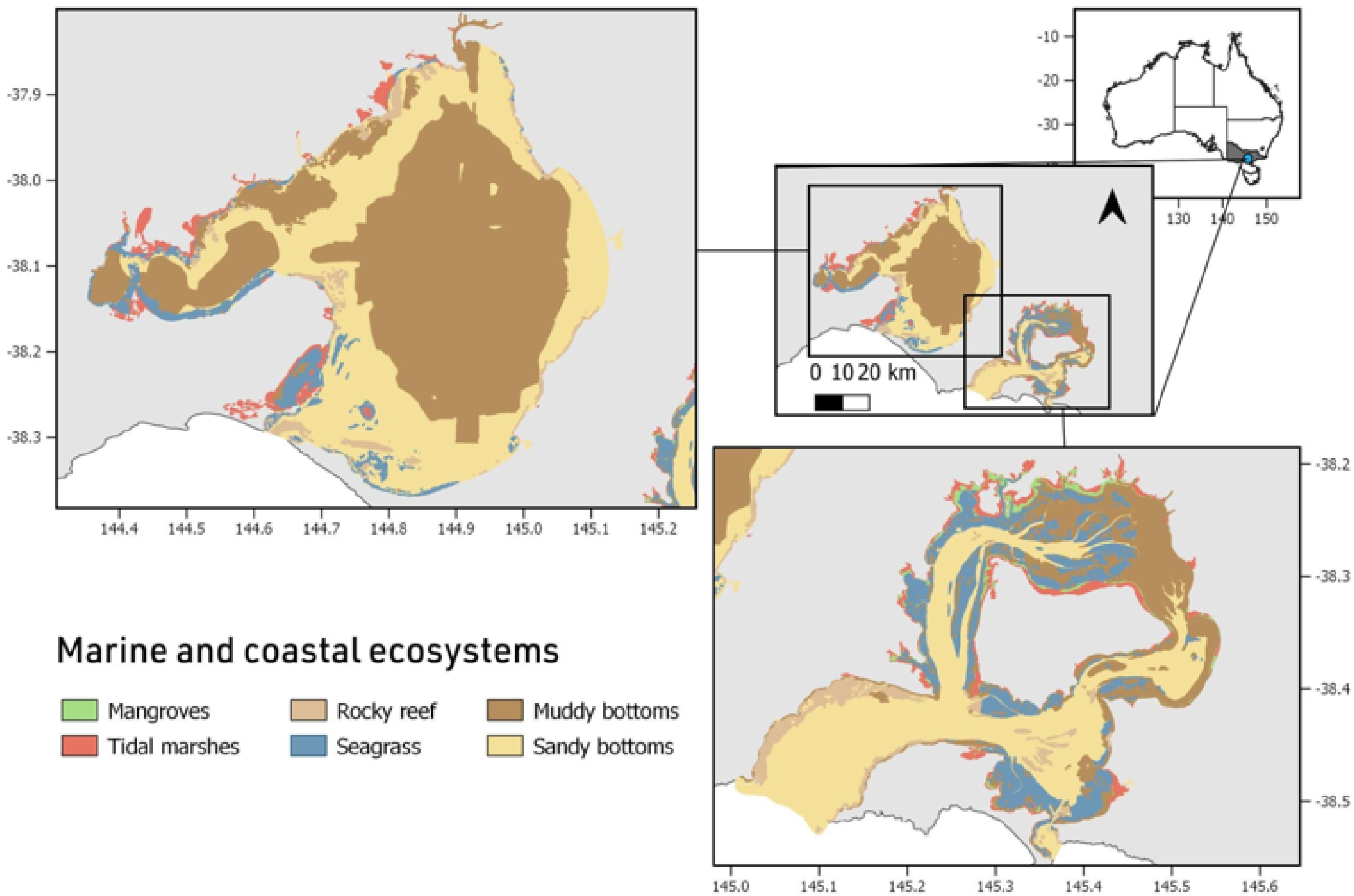

Figure 2

The extent of the coastal and marine ecosystem types of Port Phillip (a) and Western Port (b) bays in south-eastern Australia.

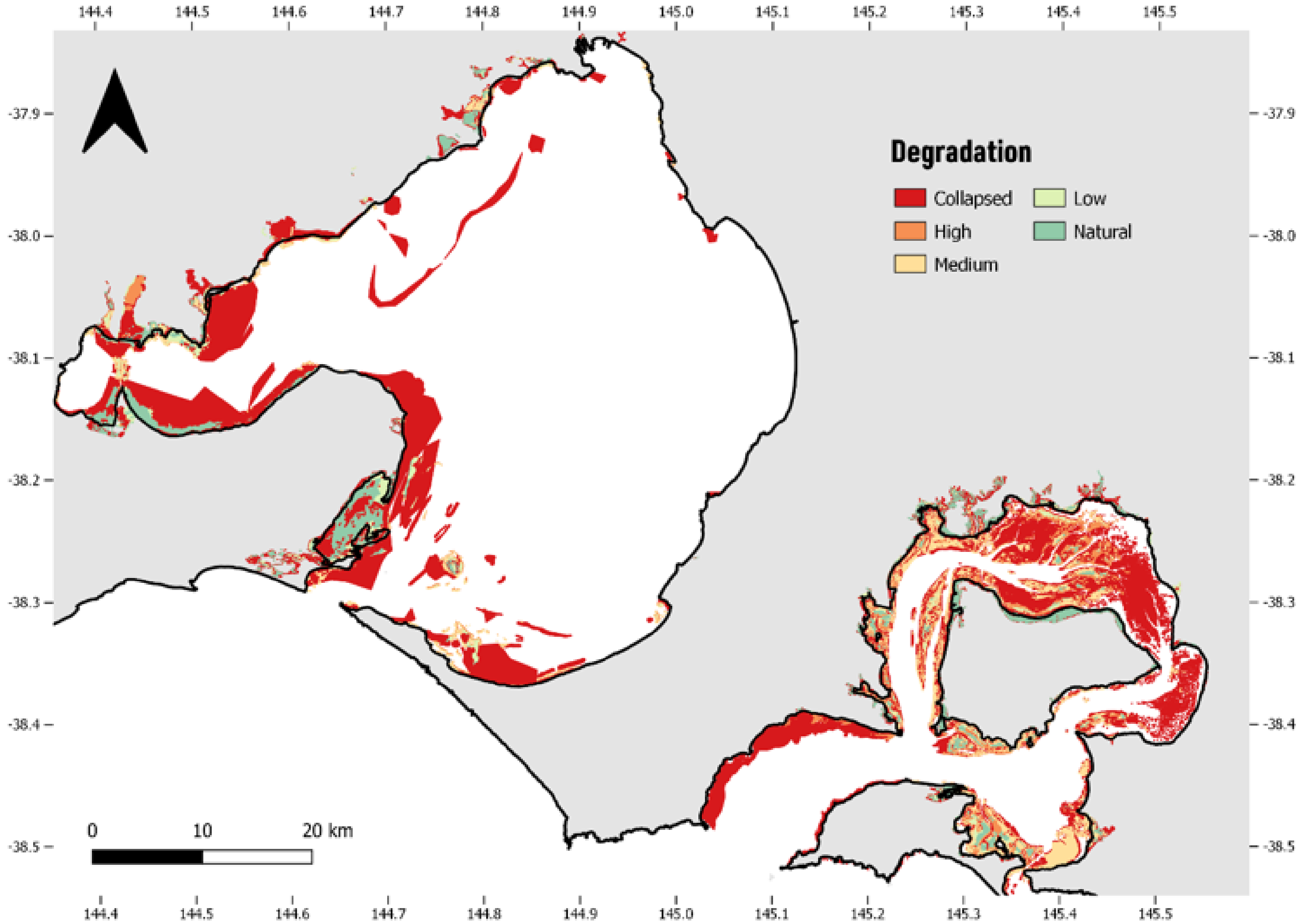

Figure 3

The condition of tidal marsh, mangrove and seagrass ecosystems in Port Phillip and Western Port Bays in southeast Australia, based on the condition framework developed (Table S2 and S3). Collapsed sites are currently not mangrove or saltmarsh ecosystems. Sites that are currently tidal marsh or mangroves but are impacted by reduced hydrology (levees: Medium disturbance) or agriculture (primarily grazing: Low disturbance) or by both threats (High disturbance). Natural ecosystems exist in areas designated for conservation or other non-impact land uses**.** The condition of seagrass ecosystems based on the age of seagrass and last presence at a given site using historical seagrass mapping. At sites that are collapsed, seagrass has not been present for over 10 years. Sites that were last present less than 10 years ago are rated as High disturbed. The age of the seagrass meadows 1-10 years (Medium), 10-20 years (Low) and >20 years (Natural) was then used to classify seagrass condition.

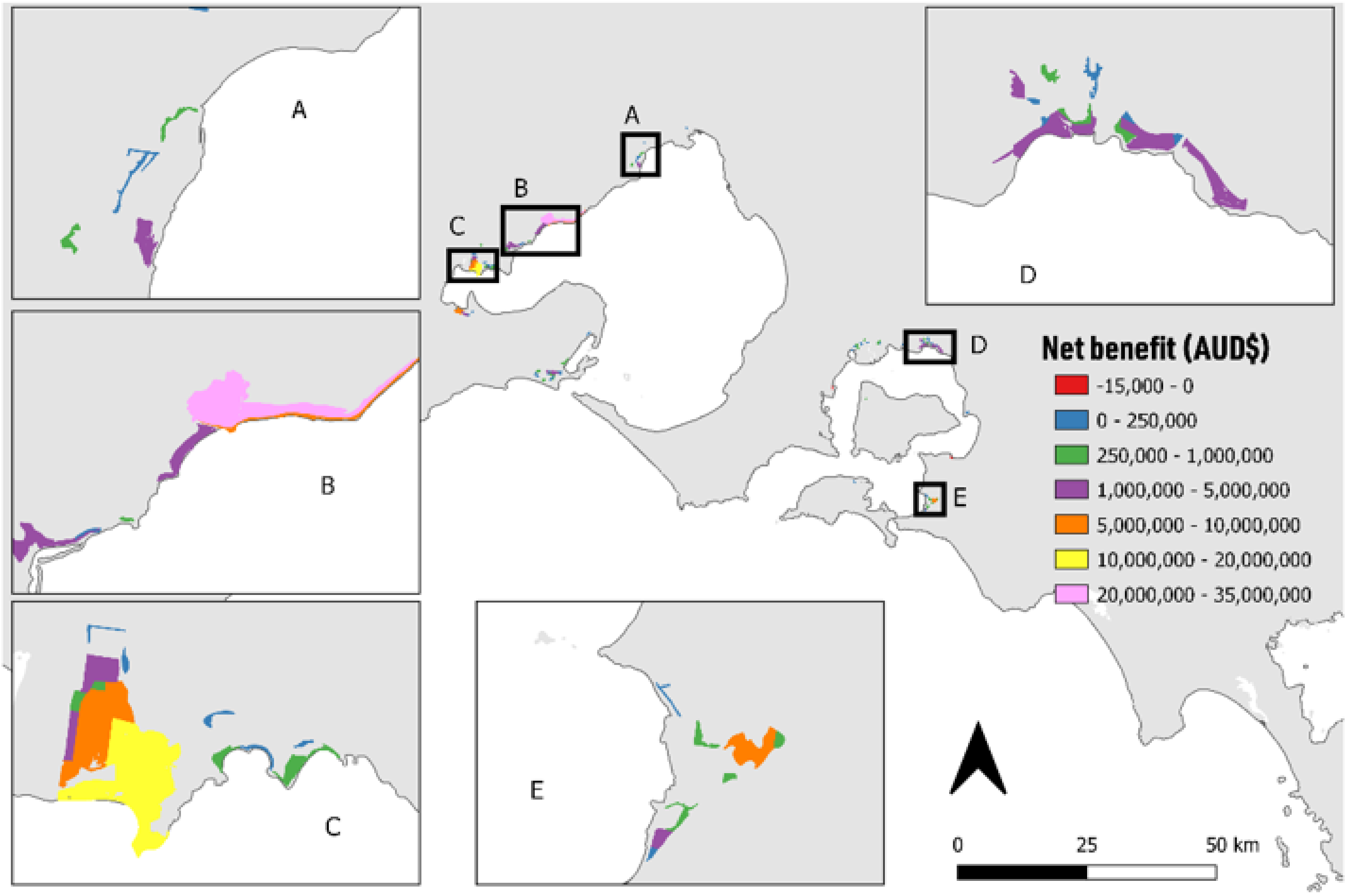

Figure 4

Net benefit (after 50 years) from restoring collapsed saltmarsh and mangrove ecosystems via fencing, levee removal and combinations of both across Port Phillip and Western Port Bays. Values represent the ecosystem service values minus the costs of restoration over a 50-year timeframe. Note net benefit does not include the opportunity cost of investing.

Supplementary Files

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