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Research article

Lessons learned on the feasibility of coastal wetland restoration for blue carbon and co-benefits in Australia

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A B S T R A C T

Diverse types of saline coastal wetlands contribute significantly to global biodiversity, carbon stocks, and ecosystem functions. Opportunities to incentivise coastal wetland restoration from carbon markets is growing across the world. However, little is known of the economic feasibility of blue carbon restoration across different regions, or the quantities of ecological and social co-benefits that accompany restoration. We explored the opportunities for tidal restoration of coastal wetlands for blue carbon projects in three regions across Australia. We identified biophysically suitable potential restoration sites for mangroves, saltmarshes and supratidal forests, estimated their carbon abatement over 25 years, and undertook a cost-benefit analysis under the carbon market. Potential co-benefits of restoration sites for biodiversity, fisheries, water quality and coastal protection were measured to identify economically feasible sites that maximise the provision of co-benefits. Cultural benefits were identified as the potential for leadership and collaboration by Traditional Custodians at sites. We found that the extent of restoration opportunities varied among regions, with variation in tidal range, extent of agricultural land-use, and the type of hydrological modifications influencing carbon abatement forecasts. The presence of threatened species in hydrologically modified wetlands reduced the amount of land available for restoration, however the restoration of remaining areas could produce rich ecological and cultural benefits. A high carbon price was needed to make blue carbon restoration profitable on land used for beef production. We found sites where carbon credits can be bundled with co-benefits to possibly attain higher carbon prices. Traditional Custodians were interested in leading blue carbon projects, however the opportunity is dependent on Native Title rights. Through comparison of case studies, we developed a regional approach to identify coastal wetland restoration sites for blue carbon and co-benefits that can incorporate local knowledge and data availability, engage with Traditional Custodians, and adapt to the unique characteristics of regions.

1. Introduction

Coastal wetlands, including mangroves, saltmarshes, and tidally influenced supratidal forests store high amounts of carbon, contributing to climate change mitigation (Adame et al., 2020; Serrano et al., 2019). They also support fisheries, nutrient processing and coastal protection (Barbier et al., 2011), and are culturally important to Indigenous people (Clarke et al., 2021). A range of marine and terrestrial fauna utilise coastal wetlands including threatened and migratory species (Rog et al., 2017; Sievers et al., 2019). Yet, large losses and degradation of coastal wetlands have occurred globally (Murray et al., 2022) including in Australia, particularly from drainage, infilling and flood mitigation works as part of agricultural, industrial and urban expansion (Rogers et al., 2016). The hydrology of coastal floodplains is also highly modified from impounding for flooded pastures and aquaculture (Goldberg et al., 2020) and construction of dams and barrages that reduce freshwater flows (Grill et al., 2019). Remaining wetlands are threatened by poor water quality, invasive species such as feral ungulates (e.g. cattle, pigs, and buffalo), and sea-level rise (Mihailou and Massaro, 2021; Ostrowski, Connolly and Sievers, 2021; Schuerch et al., 2018). Hydrologically modified floodplain landscapes may provide opportunities for tidal restoration of coastal wetlands for carbon credits, thereby providing monetary incentives for land holders and managers to undertake restoration. In Australia, carbon credits can be awarded to projects that remove or modify tidal restrictions and reintroduce tidal flows (Lovelock et al., 2023). Funding from carbon projects may provide the sustained funding that are needed to accelerate coastal wetland restoration efforts (Waltham et al., 2020).

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In a high rainfall region of north-east Australia, large areas of low-lying sugarcane and grazing land were found to be economically feasible for coastal wetland restoration for blue carbon (Hagger et al., 2022) while leaving the remaining areas of farms to remain productive (Waltham et al., 2021). However, the factors influencing opportunities for coastal wetland restoration vary across Australia's coastline because of variation in hydrology (Howard et al., 2017; Montalto and Steenhuis, 2004), land-uses (Rogers et al., 2023), and the levels of carbon abatement that could be achieved (Kelleway et al., 2017). Additionally, biodiversity and the provision of other ecosystem services varies regionally and locally (Adame et al., 2015; Ouyang et al., 2018). Therefore, it is difficult to identify where to implement restoration at scale due to variations in regional context and data availability. These factors are likely to apply in other parts of the world where blue carbon projects are planned.

Spatial approaches for prioritisation of mangrove and saltmarsh restoration have focused on environmental suitability and landscape connectivity (Shao et al., 2021; Su et al., 2022), bird species conservation (Klingbeil et al., 2018), or cost-effectiveness given the provision of benefits (Adame et al., 2015), however these do not account for the financial benefit from carbon credits. Spatial planning for coastal and marine restoration and accounting for ecosystem service outcomes can help support more effective restoration (Lester et al., 2020).

Here, we evaluated the opportunity for coastal wetland restoration in three regions in Australia that vary in their climatic and hydrological characteristics, agricultural land-uses, wetland types, potential carbon abatement and co-benefits, and which are in different jurisdictions, giving rise to variation in data availability. Our aims were to (1) identify the extent of biophysically suitable areas on agricultural land for tidal restoration of coastal wetlands through removing or modifying tidal restriction structures to reinstate tidal flows. This allows for natural recovery of saline mangroves, saltmarshes, and/or supratidal forests across the intertidal and supratidal zones (Lewis, 2005), (2) assess the economic feasibility for landholders to undertake tidal restoration for carbon abatement, and (3) develop an approach to select sites for blue carbon restoration projects that are profitable and maximise potential co-benefits for biodiversity, fisheries, water quality, and coastal protection.

2. Material and methods

2.1. Case study regions

We applied a regionally-specific approach (Fig. 1) to identify restoration opportunities in (1) Fitzroy Basin in central-eastern Queensland, the largest case study region with multiple catchments draining into the World Heritage listed southern Great Barrier Reef through several open estuaries, (2) Peel-Harvey catchment and the northern part of the south-west catchment in south-west Western Australia characterised by closed and semi-enclosed estuaries, and (3) the Ord region in east Kimberly, north-east Western Australia which includes the Ord River floodplain (Fig. 2). The climate zones range from subtropical with mostly summer dominant rainfall between 650 and 1200 mm median annual in Fitzroy Basin, temperate with winter dominant rainfall of >800 mm median annually in Peel-Harvey, to tropical savanna with summer dominant rainfall of 650–1200 mm median annually in the Ord (Bureau of Meterology, 2022). In Peel-Harvey, the tidal ranges are microtidal (0.5–1 m) compared to 2.2–6.6 m for Fitzroy Basin, and macrotidal (7–9 m) for the Ord region. The main agricultural land-uses are beef cattle in Fitzroy Basin (Merrin et al., 2018), beef cattle and cropping in Peel-Harvey (Kelsey et al., 2011), and grazing and irrigated agriculture in the Ord region (CRCNA, 2020). All three case study regions have experienced hydrological modification. In Fitzroy Basin 83% of the mapped wetlands have been modified, such as by construction of bund

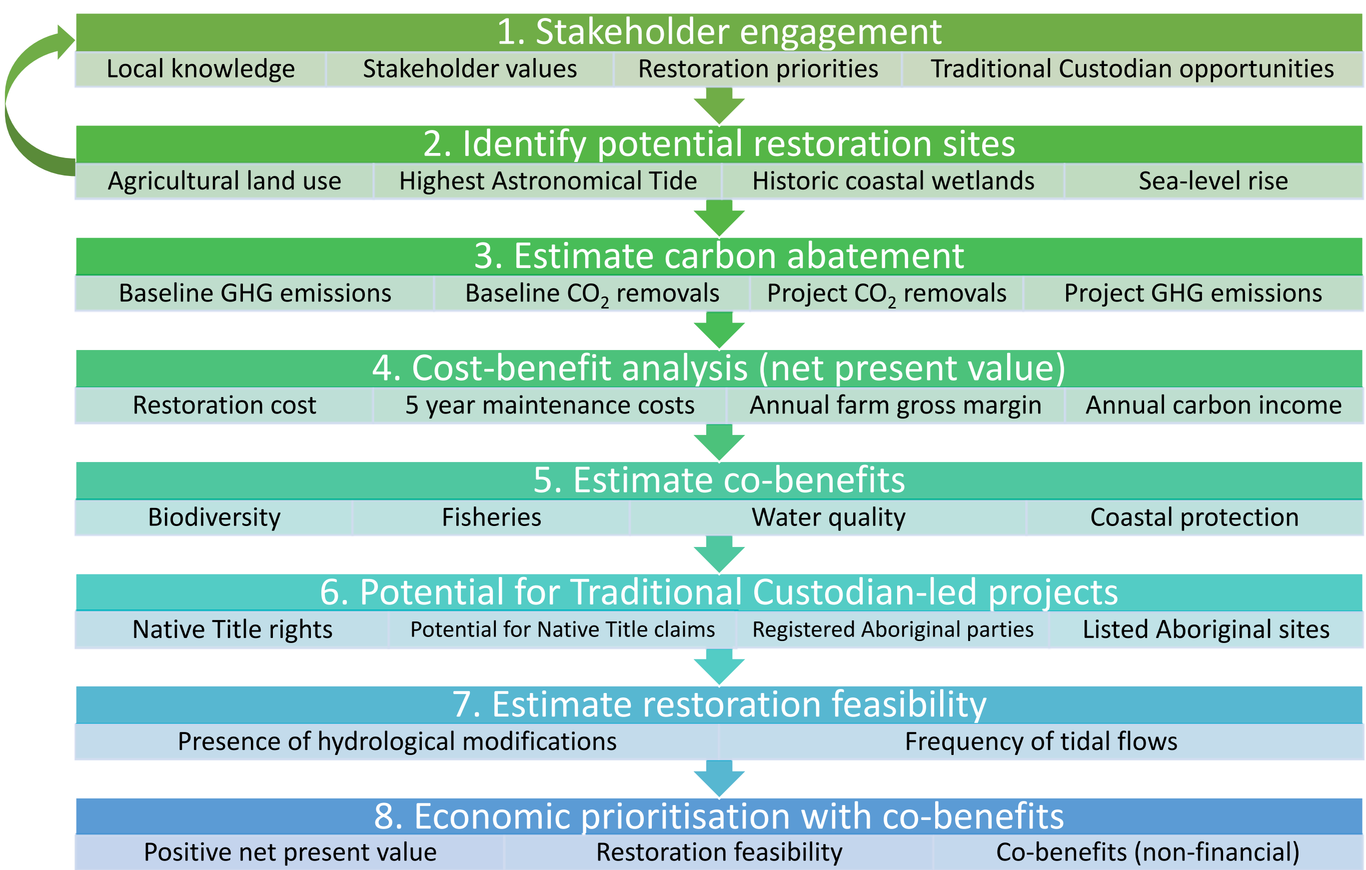

Fig. 1. Approach to select coastal wetland restoration sites for blue carbon and co-benefits.

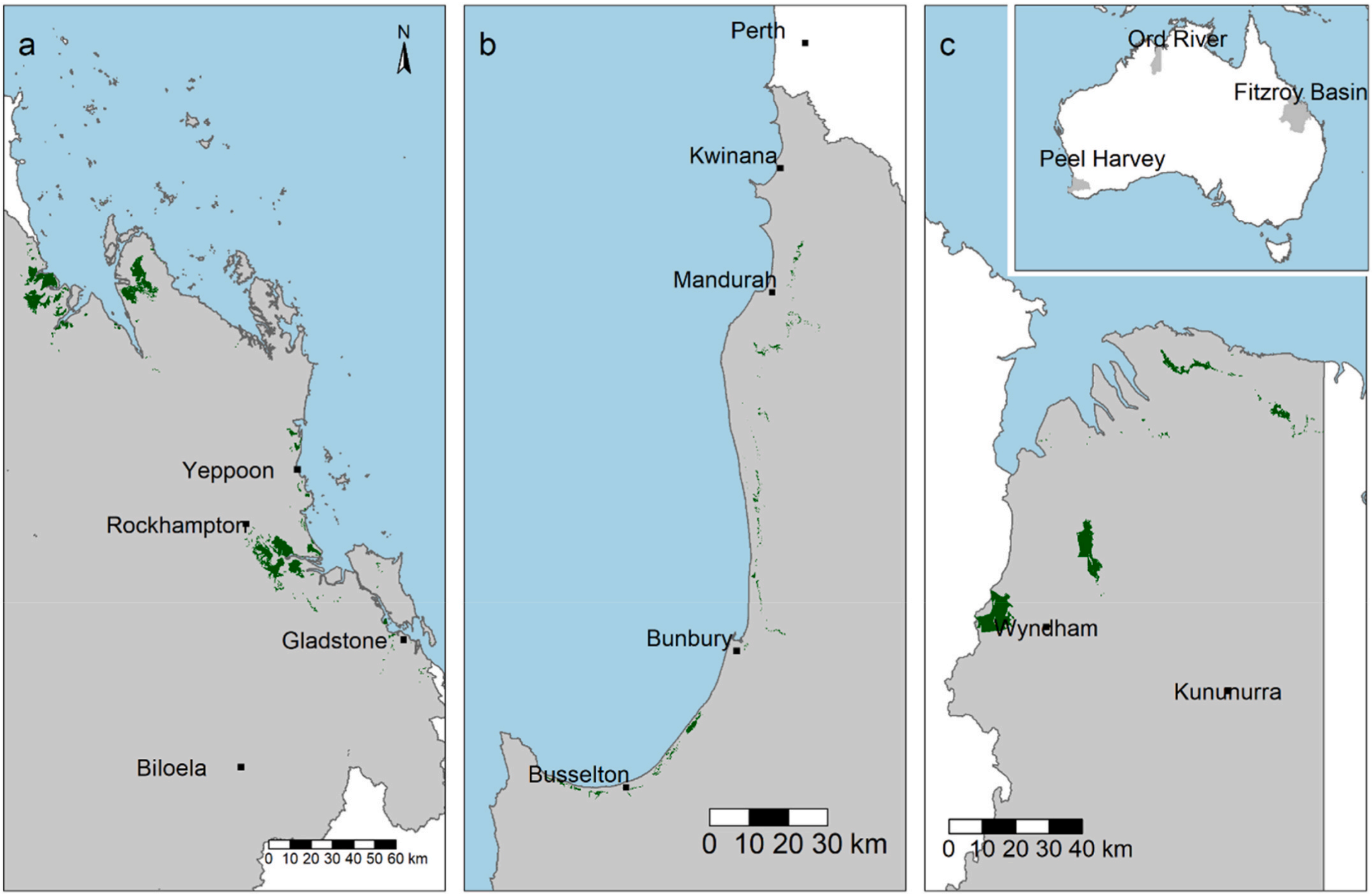

Fig. 2. Location of Fitzroy Basin (a), Peel-Harvey (b) and Ord River (c) case study regions in Australia with potential coastal wetland restoration areas in green and study regions in grey. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

walls to exclude tidal flows and create flooded freshwater pastures (Department of Environment and Science, 2022). The Ord region encompasses the Ord River Irrigation Area, 22,000 ha of irrigated agriculture with water fed from the Ord River diversion dam and Argyle dam (CRCNA, 2020). The coastal plains of the Peel-Harvey estuary have been extensively drained since 1930s–40s with artificial openings to the ocean created to address eutrophication (Hennig et al., 2021).

2.2. Stakeholder engagement

We met with state government agencies and non-government organisations to consider the different perceptions and goals for restoration held by organisations involved in natural resource management (Hagger, Dwyer and Wilson, 2017). We did this in order to refine the methods for identifying restorable areas and for measuring co-benefits, based on local knowledge and data availability and perceptions on coastal wetland values. We also engaged with Traditional Custodians (Indigenous people who have responsibilities in caring for their Country) and Indigenous groups to explore their interest in undertaking blue carbon projects and opportunities for Traditional Custodian-led blue carbon projects. A workshop was held with Traditional Custodian representatives of the Fitzroy Basin coast, including Darumbal Enterprises, Darumbal People Aboriginal Corporation, Port Curtis Coral Coast Trust and Koinmerburra Aboriginal Corporation. An information session was held with Indigenous groups who are members of the Indigenous Carbon Industry Network including representatives from Kimberley Land Council, Northern Land Council, Indigenous Land and Sea Corporation, and the Arnhem Land Progress Aboriginal Corporation.

2.3. Identification of potential restoration sites

Potential restoration sites were agricultural land-use parcels that were $\geq$1 ha within the Highest Astronomical Tide level that had the potential to be inundated with tidal waters, and historically had wetland vegetation. In coastal wetlands, hydrological regimes are influenced by water flows from groundwater, surface water, rivers, and the ocean (Twomey et al., 2024). We have modelled the likely extent of tidal inundation using catchment tide heights and elevation. However, hydrological modelling and assessment of hydrological connectivity between freshwater and marine environments may be needed in selecting coastal wetland restoration sites that facilitate exchange of materials, flora and fauna (Li et al., 2021; Twomey et al., 2024).

Agricultural land-use codes were identified from the regional land-use mapping programs (Table S1) (Australian Bureau of Agriculture and Resource Economics and Sciences, 2018; Department of Environment and Science, 2019b). We included grazing as the dominant agricultural land-use for all three regions. For Fitzroy Basin we also included Defence land in non-remnant areas (Department of Environment and Science, 2019a), as well as areas of wetlands used for agricultural production on freehold or leasehold land (Department of Resources, 2021), and hydrologically modified wetlands (codes in Table S1) (Department of Environment and Science, 2020). There was no data to indicate the extent of hydrological modified wetlands in Western Australia, although drainage works are present (Department of Water, 2008). Small areas of cropping (land-use codes 3.3 and 4.3) and abandoned intensive animal production (land-use code 5.2.8) were excluded from the analysis (Table S2).

Land with historic coastal wetland vegetation (mangrove, saltmarsh, and supratidal forest comprising *Melaleuca*, *Eucalyptus*, *Casuarina* or *Acacia* spp) or freshwater wetland (riverine forest, swamps, vine forest, sedgelands, and open water; Table S3) was determined from pre-clear vegetation mapping (Department of Environment and Science, 2019a; Department of Primary Industries and Regional Development, 2017). Our rationale for including freshwater wetlands was that agriculture can result in subsidence and compaction of organic soils (White and Kaplan, 2017), thus these areas may transition to some type of coastal wetlands. We assumed natural recovery of vegetation would follow reintroduction of tidal flows given the large areas of natural coastal wetland vegetation in the regions and potential for dispersal (Lovelock et al., 2023). Pre-clear mangrove, saltmarsh and supratidal forest vegetation types were assumed to transition to the mangrove, saltmarsh and supratidal forest categories used in the *Tidal restoration of blue carbon ecosystems method*, respectively (Table S4). Pre-clear floodplain forests and sedgeland were assumed to transition to the supratidal forest category (e.g. *Melaleuca*, *Casuarina*) and/or to saltmarsh or sparsely vegetated saltmarsh, depending on climatic region (Lovelock et al., 2023).

A Highest Astronomical Tide (HAT) inundation area (based on Australian Height Datum, AHD) was developed for each region using the mean sea level (MSL) and HAT tide predictions for standard ports and locations (Keysers, Quadros and Collier, 2012; Maritime Safety Queensland, 2021) and a Digital Elevation Model (DEM; 5 m resolution) (Geoscience Australia, 2015). Note an allowance of 2.5 mm per year for sea level change has been made in the MSL estimate for Queensland tidal planes. For Fitzroy Basin, the intertidal zones were defined according to tide levels: low (from MSL to mean high water neaps, MHWN), mid (MHWN – mean high water springs, MHWS) and high (MHWS – HAT). Average tide levels were calculated from predictions located in each catchment (Table S5). For Peel-Harvey and the Ord region, the data

Table 1

Greenhouse gas emissions and removals estimated for baseline land uses and restored coastal wetlands. Default values (mean or median) with upper and lower 95% confidence intervals in brackets, where relevant.

Land use/wetland	Emission/removal	Activity	Emission Factor (kg ha^{-1} yr^{-1})	Removal Factor (Mg C ha^{-1} yr^{-1})	Method
Baseline land-uses - emissions					
Grazing	Methane (CH_4)	Flooded agricultural land, managed wet meadow or pasture	325.0	–	National emission factors (median values of CH_4 and N_2O emissions) from Australian coastal land published and unpublished data (Lovelock et al., 2023).
	Nitrous oxide (N_2O)	Flooded agricultural land, managed wet meadow or pasture	14.0	–	
	CH_4	Ponds and other constructed water bodies	226.3	–	
	CO_2	Soil carbon loss	Table S6	–	Default stock change factors (IPCC, 2006) applied to site-specific soil organic carbon stocks from Australian baseline map of soil organic carbon (Lovelock et al., 2023; Viscarra Rossel et al., 2014).
Baseline land-uses - removals					
Tidally-restricted wetland (freshwater or brackish)	CO_2	Soil carbon accumulation in hydrologically disturbed mangrove, saltmarsh, and herbaceous settings (degraded wetlands)	–	0.47	National default values from Kelleway et al. unpublished data and Jones, Lavery et al. unpublished data (Lovelock et al., 2023).
Supratidal forest	CO_2	Soil carbon accumulation in disturbed supratidal forest	–	0.61 (0.51, 0.74)	National default value (Lovelock et al., 2023).
Restored coastal wetlands – removals					
Mangrove	CO_2	Soil carbon accumulation	–	0.95 (1.07, 1.73)	National default values from Serrano et al., (2019), updated to include recently published and unpublished datasets (Lovelock et al., 2023).
Saltmarsh	CO_2	Soil carbon accumulation	–	0.48 (0.32, 1.21)	
Supratidal forest	CO_2	Soil carbon accumulation	–	0.61 (0.51, 0.74)	
Mangrove	CO_2	Below-ground biomass carbon accumulation		Root shoot ratio: 0.32 (0.17, 0.47)	Proportion of above-ground biomass to below-ground biomass (root shoot ratio) (Lovelock et al., 2023).
Saltmarsh	CO_2	Below-ground biomass carbon accumulation	–	Root shoot ratio: 0	
Melaleuca	CO_2	Below-ground biomass carbon accumulation		Root shoot ratio: 0.27	
Mangrove	CO_2	Above-ground biomass carbon accumulation	–	Stocks Mg C ha^{-1}: Tropical: 167 (115.89, 218.11). Temperate: 70.4	Logistic growth curve model of above-ground biomass carbon accumulation for 25 years from mature mangrove above-ground biomass carbon stock values in tropical humid and temperate Australia (Table S7) (Lovelock et al., 2023).
Saltmarsh	CO_2	Above-ground biomass carbon accumulation ≤1 year	–	Stocks Mg C ha^{-1}: Tropical: 1.36 (0.94, 1.78). Temperate: 7.89	Mature saltmarsh above-ground biomass carbon stock values in subtropical (no data for tropical) and temperate Australia grown in the first year (Lovelock et al., 2023).
Supratidal forest	CO_2	Above-ground biomass carbon accumulation	–	Stocks Mg C ha^{-1}: Tropical: 192 (148.99, 236.01). Temperate: 178 (137, 219)	Logistic growth curve model of above-ground biomass carbon accumulation for 25 years from mature stocks for supratidal forest in tropical humid and temperate Australia (Table S7) (Lovelock et al., 2023).
Restored coastal wetlands - emissions					
Mangrove	CH_4	Flooding of coastal wetlands	2.19	–	National emission factors (median values of CH_4 and N_2O emissions) from tropical humid and temperate climate region (same value) in Australian coastal wetlands from published and unpublished data (Lovelock et al., 2023).
	N_2O	Flooding of coastal wetlands	0.24	–	
Saltmarsh	CH_4	Flooding of coastal wetlands	0.11	–	
	N_2O	Flooding of coastal wetlands	0.13	–	
Supratidal forest	CH_4	Flooding of coastal wetlands	−2.19	–	
	N_2O	Flooding of coastal wetlands	0.25	–	

available did not provide tide levels for MHWN and MHWS, therefore only the HAT inundation area was developed.

Stakeholder engagement identified the presence of the nationally critically endangered Capricorn Yellow Chat (*Epthianura crocea macgregori*) in Fitzroy Basin, on grassy marine plains associated with tidal-exclusion banks (Houston, Black and Elder, 2013). Potential restoration sites intersecting chat populations were removed from the analysis to avoid adverse impacts on this species where a blue carbon project might be recommended.

The influence of sea-level rise on the potential restoration sites was assessed by applying an additional 1 m to the HAT, which approximates higher range of projected sea-level rise by 2100 (Lovelock et al., 2022) and 0.7 m which is the median sea-level rise projected under Representative Concentration Pathway 8.5 by 2100 (Oppenheimer et al., 2019).

2.4. Estimation of carbon abatement

Annual carbon abatement from restoring mangroves, saltmarshes, and supratidal forests at the Fitzroy Basin and Peel-Harvey sites was calculated over 25 years following the Blue Carbon Accounting Model (BlueCAM) from the Australian *Tidal restoration of blue carbon ecosystems* method (Lovelock et al., 2023). Carbon projects in Australia have a 25-year crediting period although projects are encouraged to have a 100-year permanence period (Department of Climate Change Energy Environment and Water, 2022).

Greenhouse gas (GHG) emissions and removals were calculated for the baseline (pre-restoration) and project (after restoration) scenarios (Table 1; detailed methods in Tables S6 and S7). Annual carbon abatement was calculated as the baseline GHG emissions minus the baseline CO_2 removals plus the project CO_2 removals minus the project GHG emissions, all in CO_2-e (carbon dioxide equivalents). The extent of baseline land-uses were estimated from available spatial datasets (Table S6). Carbon estimation areas (linked to ecosystem categories) for the project scenario were estimated based on the pre-clear vegetation mapping (Tables S3 and S4).

Soil carbon sequestration was estimated using national default values for mangroves, saltmarsh and supratidal forest. Above-ground biomass carbon accumulation over 25 years was modelled from values of mature carbon stocks of mangroves, saltmarsh and supratidal forest from different climate regions of Australia using a logistic growth curve. Below-ground biomass carbon was estimated using the proportion of above-ground biomass to below-ground biomass (root to shoot ratio) for mangroves, saltmarsh and supratidal forest. Carbon accumulation was assumed to initiate when natural vegetation becomes established, in year one after tidal flow is reinstated (year 0). We did not apply the suggested above-ground biomass and soil carbon multiplier of 0.7 for scrub mangroves in tropical climates to Fitzroy Basin, because of uncertainty in elevation data and the transition to scrub mangroves in the mid-high intertidal zones. Methane (CH_4) and nitrous oxide (N_2O) emissions from mangroves, saltmarsh and supratidal forest were estimated using nationally-derived emission factors for different climate regions of Australia. We did not include CO_2 emissions from vegetation die-off due to ecosystem transitions, because of uncertainty with the biomass carbon of existing terrestrial vegetation, and of transitions of one coastal wetland type to another under sea-level rise.

To assess uncertainty with model inputs on carbon abatement forecasts, carbon abatement was calculated using median values for soil carbon accumulation rates, above-ground biomass carbon stocks, and root to shoot ratios, and then using the lower and upper 95% confidence interval values for each carbon pool.

Potential restoration sites identified in the Ord region were mapped by the state as grazed native vegetation. Hydrological modifications impeding tidal flows were not mapped or there aren't any. Therefore, potential carbon abatement was estimated by multiplying the area of remnant vegetation within each potential restoration site by our conservative estimate of carbon abatement (0.3 CO_2-e ha yr^{-1}) for avoided CO_2 and N_2O emissions from preventing soil disturbance from reduced grazing (supplementary material S1).

2.5. Cost-benefit analysis

We used net present value (NPV) to analyse the economic feasibility of converting agricultural land-use to coastal wetlands under the carbon market (Hagger, Waltham and Lovelock, 2022; Roebeling et al., 2007). In agricultural regions, the costs and benefits of a blue carbon project are likely to be accrued by the landholder, therefore we included the financial benefit from carbon abatement (the sale of carbon credits), opportunity cost from income foregone from agricultural production, and restoration and maintenance costs (equation (1)). Restoration cost data was limited to capital costs, however tidal restoration can incur high costs associated with obtaining legal permits (Bell-James, Foster and Shumway, 2023), which is a limitation of the NPV analysis. NPV was not calculated for the Ord, due to uncertainty with the condition of grazed native vegetation and thus carbon abatement.

The *NPV* for each restoration site (i) was calculated as:

$$NPV_i = \sum_{t=1}^{T} \left(\frac{(B_t - C_t) - FGM_t}{(1+r)^t} \right) - C_0 \tag{1}$$

Where T is the time period, B_t is the financial benefit from carbon abatement in each year, C_t is the maintenance cost in the first five years, FGM_t is the annual farm gross margin, r is the annual discount rate, and C_0 is the one-off restoration cost at the start.

The NPV was calculated over the 25-year crediting period, however projects were assumed to have a 100-year permanence to minimise the risk of reversal (Department of Climate Change Energy Environment and Water, 2022). The Australian Carbon Credit Unit (ACCU) spot price of AUD $57 per Mg CO_2-e on January 24, 2022 (Clean Energy Regulator, 2022a) was used to calculate carbon credits, since higher carbon prices can be obtained for projects with high co-benefits (Lou et al., 2022) and may be even higher on the voluntary market (Kuwae et al., 2022). Farm gross margins (FGM, AUD yr^{-1} per site) were estimated for beef cattle grazing (supplementary material S2). We used median restoration costs reported for mangroves and saltmarsh for developed nations (Australia, United States of America and United Kingdom) using natural recovery hydrological restoration (without planting) (Bayraktarov et al., 2016). Costs in USD ha^{-1} at 2010 were converted to AUD using the 2010 exchange rate (1.09) (Feenstra, Inklaar and Timmer, 2015), and then escalated from December 2010 to December 2022 using the relevant CPI (all groups, Brisbane; 36% increase) (Australian Bureau of Statistics, 2021). The majority of costs reported were for capital costs, however, some projects also included maintenance costs. We used the lower saltmarsh cost of AUD $9257 ha^{-1}, assuming hydrological restoration involves mainly earthworks for modification of drains/bunds. Given natural recovery requires minimal maintenance, AUD $750 ha^{-1} yr^{-1} for the first five years of the project was applied (Waltham et al., 2021). We applied cost reduction rates on restoration and maintenance costs based on economies of scale for larger terrestrial restoration projects (Strassburg et al., 2019) (Table S10). We applied a discount rate (r) of 1% per annum, considered realistic for climate change mitigation projects, like coastal wetland restoration, that improve with age and their accumulation of carbon stocks and co-benefits (Costanza et al., 2021).

Sensitivity analyses (Firn et al., 2015) were conducted to assess how a higher mangrove restoration cost of AUD $76,893 ha^{-1}, higher international carbon unit spot price of AUD $132 on December 31, 2022 (Clean Energy Regulator, 2022b), higher 4% discount rate (House of Representatives Standing Committee on Infrastructure Transport and Cities, 2018), and higher FGM for beef cattle breeding changes the NPV (Table 2). In addition to variations in cost parameters, we also included scenarios that change the inputs to the carbon model parameters to assess how data processing decisions affect the NPV (Steegen et al.,

Table 2
Net Present Value (NPV) scenarios, including base and six sensitivity analyses with variations of carbon abatement, restoration cost, discount rate, carbon price, and farm gross margin (FGM)*. *For Peel-Harvey only. FGMs for Fitzroy Basin varied across sites given land type.

Scenario	NPV equation	Restoration cost (AUD ha^{-1})	Discount rate (%)	Carbon price (AUD Mg CO_2-e)	Carbon abatement	FGM (AUD ha^{-1} yr^{-1})*
1	Base	$9,257	1	$57	Default carbon values	$83.99
2	Lower carbon abatement	$9,257	1	$57	Lower 95% CI values	$83.99
3	Higher carbon abatement	$9,257	1	$57	Upper 95% CI values	$83.99
4	Higher restoration cost	$76,893	1	$57	Mean	$83.99
5	Higher carbon price	$9,257	1	$132	Mean	$83.99
6	Higher discount rate	$9,257	4	$57	Mean	$83.99
7*	Higher farm gross margin	$9,257	1	$57	Mean	$593.12

2016).

2.6. Estimation of co-benefits

We measured potential co-benefits of restoring the sites for biodiversity, fisheries, water quality, and coastal protection in non-financial value, using indicators likely to influence the provision of each benefit, based on scientific knowledge, local conditions, and data availability for each region (Table S11). We selected these four co-benefits because of the importance of coastal ecosystem restoration to address biodiversity loss (Lotze et al., 2006) and provide ecosystem services including flood protection for infrastructure (Menéndez et al., 2020), support for commercial fisheries (Jänes et al., 2020), and water quality improvement (Iram et al., 2022). Additionally, these co-benefits were considered important to stakeholders. Co-benefits are mostly likely to benefit industries and the community, with some accrual to the proponent, such as reducing the potential for salinity and acid sulphate soils (Rogers et al., 2023) and protecting properties from damages from storms (Bell-James et al., 2022). However, co-benefits have indirectly been considered in the NPV analysis, through the use of higher "spot" carbon prices, which are possible for carbon projects with co-benefits.

The biodiversity value of potential restoration sites was estimated as (1) connectivity to existing estuarine and freshwater wetlands to facilitate movement of animals and dispersal of plants (Buelow and Sheaves, 2015), (2) connectivity to Ramsar wetlands that likely have enhanced ecological character (Davidson et al., 2020), (3) threatened and migratory species diversity (Rog et al., 2020), and (4) potential habitat for a focal threatened species or ecological community in the study region. Although larger patch sizes maintain species diversity (Bryan-Brown et al., 2020), it was not included because it was highly correlated with other area-based indicators ($r > 0.5$). The fisheries value of potential restoration sites was represented by (1) the provision of potential nursery habitat (Sheaves et al., 2012), (2) connectivity with freshwater and marine environments providing fish habitat (Sheaves et al., 2006), and (3) connectivity with declared Fish Habitat Areas (if present, which are designated areas close to recreational and commercial fishing given their high value for fisheries production). Connectivity to wetlands for biodiversity was measured by the Euclidean distance to nearest wetland. However, connectivity for fish habitats was measured as the hydrological connectivity along a flow path (e.g. permanent watercourse).

Water quality improvements was represented by the capacity for potential restoration sites to remove Dissolved Inorganic Nitrogen (nitrate + ammonium, DIN) from the water, estimated by (1) DIN or Total Nitrogen catchment concentration, (2) total suspended solids concentration (if available), (3) hydraulic efficiency, and (4) estuarine water residence time (Adame et al., 2019; Kavehei et al., 2021). DIN removal was not calculated for the Ord region due to the lack of data on DIN or Total Nitrogen concentrations. Coastal protection was estimated as wetland vegetation providing flood mitigation by (1) indirect protection from inland flooding by reducing erosion (Thampanya et al., 2006), determined as the area of the potential restoration site within the 1% Annual Exceedance Probability river flood projection model and, (2) direct protection from coastal flooding by wave attenuation during storm surges (Temmerman, De Vries and Bouma, 2012), determined as the area of historic mangroves within the potential restoration site. In Peel-Harvey, only inland flood mitigation was assessed because the coastal dunes offer direct protection from the sea, while in the Ord region only coastal flood mitigation was assessed, because there is no flood model.

2.7. Consideration of cultural benefits

Cultural heritage was not included in the co-benefits analysis of potential restoration sites because this requires engagement with Traditional Custodians on specific sites of interest. Cultural benefits have been incorporated into our approach through exploring the interest for Traditional Custodians to undertake blue carbon projects through stakeholder engagement and potential for Traditional Custodian-led blue carbon projects or partnerships with landholders across potential restoration sites using government databases. This was identified as sites with (1) Native Title or Indigenous Land Use Agreement, (2) potential for Native Title claims (lease-hold, state-owned and commonwealth-owned land), (3) registered Aboriginal parties, and (4) Aboriginal sites or places of cultural importance listed on state databases (Table S11). Should projects be led by Traditional Custodians, then the costs of restoration and financial benefit of carbon abatement would be accrued by Indigenous groups, which would lead to multiple benefits to Indigenous people (Section 3.6).

2.8. Estimation of restoration feasibility

Restoration feasibility was the probability that tidal restoration can be successfully implemented and be effective. Reinstatement of tidal flows are likely to be more feasible on sites that have manmade drains or barriers and can be hydrologically restored by modifying that structure. Frequent tidal inundation can enhance natural recruitment and positive species interactions (Lewis et al., 2019; Renzi, He and Silliman, 2019) and decrease aquatic weed invasion (Abbott et al., 2020). Restoration feasibility in the Fitzroy Basin and Peel-Harvey regions was based on the presence of an existing drain or barrier and the likely frequency of tidal flows. Restoration feasibility was not calculated for the Ord region as no hydrological modifications were apparent within the potential restoration sites. Major up-stream hydrological change has occurred in this region as part of the Ord River Irrigation Scheme, however none of the irrigated agricultural areas are influenced by tidal flows (Table S12).

2.9. Economic prioritisation analysis

We adapted cost-effectiveness approaches (Klein et al., 2017; Laycock et al., 2009) to prioritise profitable sites that maximise the provision of co-benefits, using sites that returned a positive NPV, the provision of co-benefits for biodiversity, fisheries, water quality and coastal protection, and restoration feasibility. We used the NPV scenario with the higher carbon price given higher prices can be obtained on the voluntary carbon market for projects with high social or environmental value.

The ecosystem service multifunctionality approach was used to estimate co-benefits likely to be provided by restoration in non-financial values (Manning et al., 2018). Co-benefit indicator values were rescaled between 0 and 100 using the maximum value for each indicator in the set of restoration sites, multiplied by the weighting of the indicator, and summed. Co-benefits were initially given equal weights (0.25) which were divided equally among the indicators for each co-benefit. To analyse the sensitivity of the rankings to different possible stakeholder objectives, the economic prioritisation was repeated to give a higher weighting to each co-benefit in turn (0.7 and 0.1 to others).

The total co-benefit (B) of each restoration site (i) was estimated as:

$$B_i = \sum_{n=1}^{N} [I_n \times W_n] \quad (2)$$

Where N is the number of indicators, I_n is the percentage by which indicator n has been met, and W_n is the proportion weight of I_n.

The economic prioritisation (EP) of the site was then defined as the NPV multiplied by the expected co-benefits - potential co-benefits (B) multiplied by the restoration feasibility (F):

$$EP_i = NPV_i \times (B_i \times F_i) \quad (3)$$

2.10. Data analysis

Under each NPV and economic prioritisation scenario, sites were ranked from highest to lowest (with 1 being the highest ranking). To analyse the sensitivity of the NPV and economic prioritisation rankings to different scenarios, we undertook hierarchical cluster analysis and non-metric multi-dimensional scaling (nMDS) based on resemblance matrices. Multivariate statistics are traditionally used in community ecology, but are useful in conservation spatial planning to assess trade-offs in reserve-design scenarios (Harris et al., 2014). Using these methods, each scenario represents a 'sampling site', and restoration sites represent 'species' which each have a ranking from 1 to the maximum number of sites in that scenario. First, each table of scenarios (rows) with site rankings (columns) was transformed into a proportion table with each site given a proportion for that scenario. A Bray-Curtis resemblance matrix was constructed on the proportion tables using the vegdist function in the vegan package of the R statistics package (Oksanen et al., 2019). The Bray-Curtis method was chosen because it excludes joint absences and is better for rank data, while the Jaccard method is for binary data (Linke et al., 2011).To compare differences between scenarios, a complete hierarchical cluster analysis was performed using the hclust function in R and a dendrogram generated (R Core Team, 2020). To compare results of different clustering approaches, a non-metric multi-dimensional scaling (nMDS) ordination of the scenarios was also constructed using the metaMDS function in the vegan package, also based on the Bray-Curtis resemblance matrix. Spatial analysis was undertaken in ArcMap 10.8 (ESRI, 2019) and QGIS (Open-source software, 2002) with the data analysed in R 4.0.2 (R Core Team, 2020) (code in Data Availability).

3. Results

3.1. Areas for potential restoration

Based on the biophysical suitability of land and restoration of grazing areas, 31,686 ha were identified for tidal restoration across Fitzroy Basin. Much lower area was available in Peel-Harvey (348 ha). In the Ord region, 24,123 ha of mapped grazing native vegetation may be restorable via restoration activities other than tidal reintroduction (e.g. control of feral ungulates) (Fig. 3a). Potential restoration areas across the three regions increased significantly with sea-level rise projected by 2100 (Table 3). In Peel-Harvey potential restoration sites were small (up to 48 ha; Fig. S2b) as opposed to several large sites over 100 ha (and up to 5000 ha; Fig. S2a) in Fitzroy Basin. In the Ord region, we aggregated sites within 5 km of each other to form eight large sites up to 8,536 ha, because landholding size is large in northern Australia.

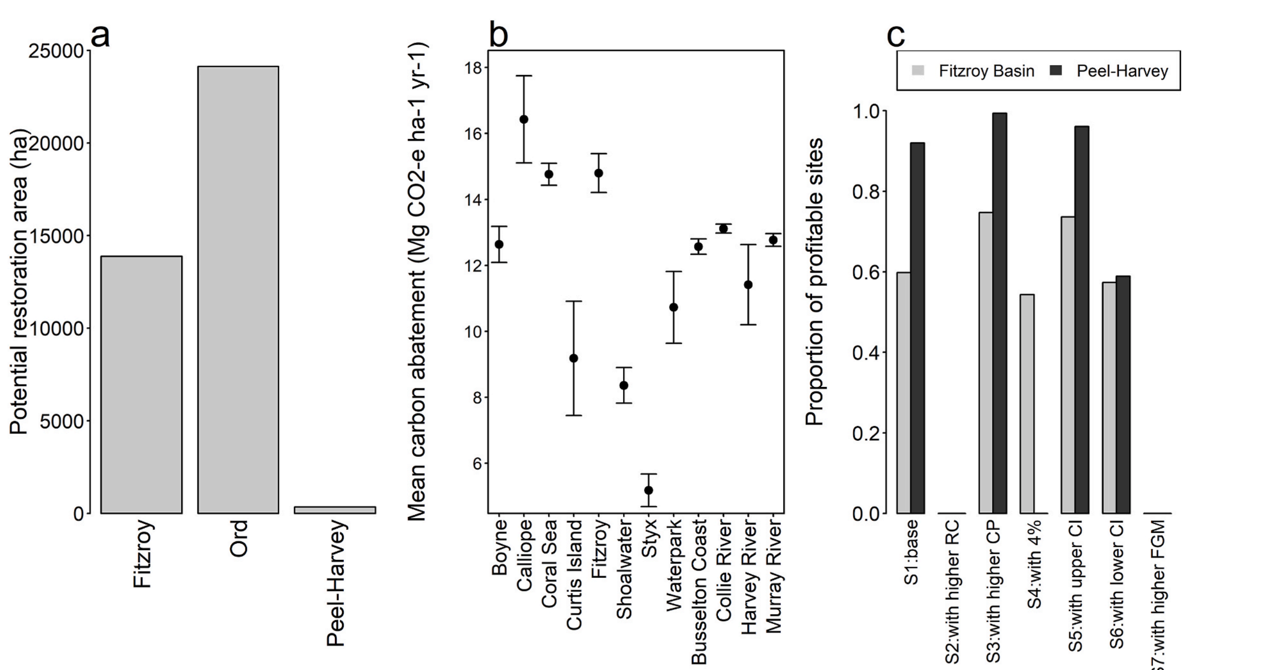

Fig. 3. Coastal wetland restoration opportunity. a, potential restoration area per catchment within the Fitzroy Basin, Peel-Harvey and Ord regions. b, mean carbon abatement with standard error bars per catchment for Fitzroy Basin (Boyne to Waterpark) and Peel-Harvey (Busselton Coast to Murray River). c, proportion of profitable sites for Fitzroy Basin and Peel-Harvey per NPV scenario. S1 (base scenario): 25 years permanence at 1% discount rate, lower $57 carbon price (CP), lower restoration cost (RC) and Farm Gross Margin (FGM) for beef trading; S2: base with higher restoration cost; S3: base with higher $132 carbon price; S4: base with 4% discount rate; S5: base with upper confidence interval (CI) for estimating carbon sequestration; S6: base with lower CI for estimating carbon sequestration; and S7: base with higher Farm Gross Margin (FGM) for beef cattle breeding (S7 only applies to Peel-Harvey).

Table 3
Areas for potential restoration in case study regions, excluding areas where the degraded coastal wetland is habitat for a threatened species and potential annual carbon abatement (averaged from 25 years).

Study region	Size (km^2)	Restorable area (ha)	+0.7 m sea-level rise (ha)	+1 m sea-level rise (ha)	With excluded areas (ha)	Site sizes	Carbon abatement (t CO_2-e yr^{-1})
Fitzroy Basin	156,000	31,686	60,142	67,097	13,874	Many small, 17 sites >100 ha	162,178
Peel-Harvey	11,000	348	1,762	2,765	348	Most small, largest 47.9 ha	4,312
Ord	50,000	24,123	30,394	51,566	24,123	Aggregated into 8 sites	7,237

3.2. Avoiding trade-offs with threatened species

In Fitzroy Basin, potential restoration sites where artificial bunded freshwater wetlands that support critically endangered Capricorn Yellow Chat populations were removed to avoid any adverse impacts on this species, reducing the potential restoration area (without SLR) to 13,874 ha (56% decrease) (Table 3).

3.3. Carbon abatement of different restoration approaches

Estimated carbon abatement per unit area from tidal restoration in Fitzroy Basin and Peel-Harvey (mean ± standard error (SE) of 9.94 ± 0.34 Mg CO_2-e ha^{-1} yr^{-1} and 12.44 ± 0.30 Mg CO_2-e ha^{-1} yr^{-1}, respectively) was much higher than possible carbon abatement from reduced grazing pressure in the Ord (0.3 Mg CO_2-e ha^{-1} yr^{-1}). Levels of carbon abatement with tidal restoration varied across catchments in Fitzroy Basin and Peel-Harvey (Fig. 3b), given variations in the extent of historic vegetation types (Table S4) and associated removals and emissions (Table S13). High variations in carbon abatement across catchments in Fitzroy Basin were likely because of the extent of mapped flooded agricultural land that emit methane and nitrous oxide and would be avoided with restoration (Table S13). Using the upper and lower 95% confidence intervals for soil carbon sequestration, AGB stocks, and BGB root to shoot ratios, the mean carbon abatement at Fitzroy Basin and Peel-Harvey ranged from 13.01 ± 0.35 to 8.45 ± 0.31 Mg CO_2-e ha^{-1} yr^{-1} and 14.69 ± 0.34 to 10.19 ± 0.22 Mg CO_2-e ha^{-1} yr^{-1}, respectively.

3.4. Profitability of blue carbon projects

For Fitzroy Basin and Peel-Harvey, 60% and 92% of the potential restoration area would be profitable under a carbon price of AU$ 57 over 25 years, which increases to 75% and 99%, respectively under a higher carbon price of AU$ 132 over 25 years. Under a higher 4% discount rate, profitability reduced to 54% and 0% for Fitzroy Basin and Peel-Harvey, respectively. But, under a higher restoration cost for Fitzroy Basin and Peel-Harvey or higher FGM for beef breeding which was only assessed at Peel-Harvey, no sites were profitable (Fig. 3c). The carbon model inputs also affected the economic feasibility assessment. When the upper 95% confidence intervals values were used to estimate carbon abatement, then 74% and 96% of the potential restoration area was profitable at Fitzroy Basin and Peel-Harvey, respectively, however using lower 95% confidence intervals reduced profitability to 57% and 59%, respectively. Both the hierarchical cluster analysis and nMDS ordination confirmed that NPV ranking of sites was sensitive to cost and carbon model parameters. At Fitzroy Basin, NPV calculated with a higher restoration cost, higher 4% discount rate, and upper 95% CI for carbon accumulation parameters were most dissimilar to the other scenarios (Fig. 5a, Fig. S2a). At Peel-Harvey, NPV calculated with higher restoration cost, upper and lower 95% CI for carbon accumulation parameters, and higher FGMs were most dissimilar to the other scenarios (Fig. 5b, Fig. S2b).

3.5. Alignment with co-benefits

Biodiversity benefits at potential restoration sites were rich across the three regions (Fig. 4a–e,i). In the Fitzroy Basin region, there were 170 records of threatened species and migratory birds within 1 km of the sites, including the Capricorn Yellow Chat and several shorebirds - Australasian Bittern (*Botaurus poiciloptilus*), Curlew Sandpiper (*Calidris ferruginea*), Beach Stone-curlew (*Esacus magnirostris*), Greater Sand Plover (*Charadrius leschenaultia*), Great Knot (*Calidris tenuirostris*), Red Knot (*Calidris canutus*), Lesser Sand Plover (*Charadrius mongolus*), and Western Alaskan bar-tailed godwit (*Limosa lapponica baueri*), and several sites were nearby the Shoalwater and Corio Bay Ramsar wetlands. In the Peel-Harvey region, most sites had records of migratory shorebirds within 1 km including the Curlew Sandpiper, Great Knot, Eastern Curlew (*Numenius madagascariensis*), and Greater Sand Plover and occurred within 10 km of a Ramsar or important wetland, containing nationally endangered temperate saltmarsh communities. In the Ord region, three of eight sites had records of the Purple-crowned Fairy-wren (*Malurus coronatus*), Australasian Bittern, Gouldian Finch (*Erythura gouldiae*), and the Knob Peak Camaenid Snail (*Ninbingia bulla*) and several migratory shorebirds within 1 km and six sites adjoined a Ramsar or important wetland. Species records were sparse in the Ord region in comparison to Fitzroy Basin and Peel-Harvey, possibly because of the remoteness of the region. Habitat requirements of the focal threatened species or community affected how the regions were measured for biodiversity. In Fitzroy Basin, we identified 3021 ha of restoration habitat for the Capricorn Yellow Chat surrounding existing populations. If restored, these sites could facilitate landward migration of chats with sea-level rise. In the Ord region, we identified 682 ha of restoration habitat for the Purple-crowned Fairy-wren and in Peel-Harvey, there could be restoration of 20 ha of the endangered temperate saltmarsh community. Enhancing habitat for threatened species and vegetation communities and migratory birds is an important component to consider when identifying restoration opportunities that can be underpinned by local data of species distributions and habitat requirements.

Fisheries, DIN removal, and flood mitigation benefits varied by study region (Fig. 4b–d,f-h,j-k). For example, in Peel-Harvey, sites far from perennial waterways (e.g. south of Bunbury) or with no connection to the ocean (e.g. Lake Clifton) had low fisheries (Fig. 4f). In Fitzroy Basin, high flood mitigation was provided at sites around Rockhampton within the Fitzroy River flood model extent and with historic mangrove cover (Fig. 4d). Surface elevation gains from restoring coastal wetlands may also increase resilience to sea-level rise, and therefore protect properties in the future (Reed et al., 2018), aligning with climate adaptation plans (Bell-James et al., 2022). In Peel-Harvey, higher DIN removal was generally found in sites with higher water residence time, higher Total Nitrogen river concentration, and higher connection to sea. However, one of the most important predictors of DIN removal is hydrology (Adame et al., 2019; Kavehei et al., 2021). This information is difficult to obtain at the study scale and could be a limitation of the measurement of DIN removal.

We incorporate within our analysis a spatially explicit landscape approach to select profitable sites for blue carbon that maximise potential benefits for biodiversity, fisheries, water quality, and flood

Fig. 4. Spatial distribution of biodiversity, fisheries, Dissolved Inorganic Nitrogen (DIN) removal and flood mitigation benefits of potential restoration sites across the Fitzroy Basin (a–d), Peel-Harvey (e–h), and Ord regions (i), with summed indicators for each co-benefit displayed as scaled icons on their centroid. All indicators are shown in their scaled form (a score of 0–100).

mitigation. We found that traditional cost-effectiveness analysis (e.g. Klein et al., 2017) is not appropriate when NPV is negative (the restoration action would result in an overall financial loss over 25 years), because it provides a relative indicator of which site is the lowest cost per percent of co-benefits, and low-cost sites were typically small sites with minimal restoration and maintenance costs that also resulted in low carbon abatement. The sensitivity analyses showed that varying the weightings of different co-benefits altered the prioritisation rankings, so that a high weighting for nitrogen removal and biodiversity resulted in different outcomes in the ranking of sites for Fitzroy Basin (Fig. 5c, Fig. S2c) and a high weighting for flood protection and biodiversity changed the ranking of sites for Peel-Harvey (Fig. 5d, Fig. S2d).

3.6. *Traditional Custodian willingness to engage in blue carbon projects*

Traditional Custodians of the Fitzroy Basin coast were interested in leading sustainable, long-term blue carbon projects as part of their cultural obligations which includes caring for their areas of Country,

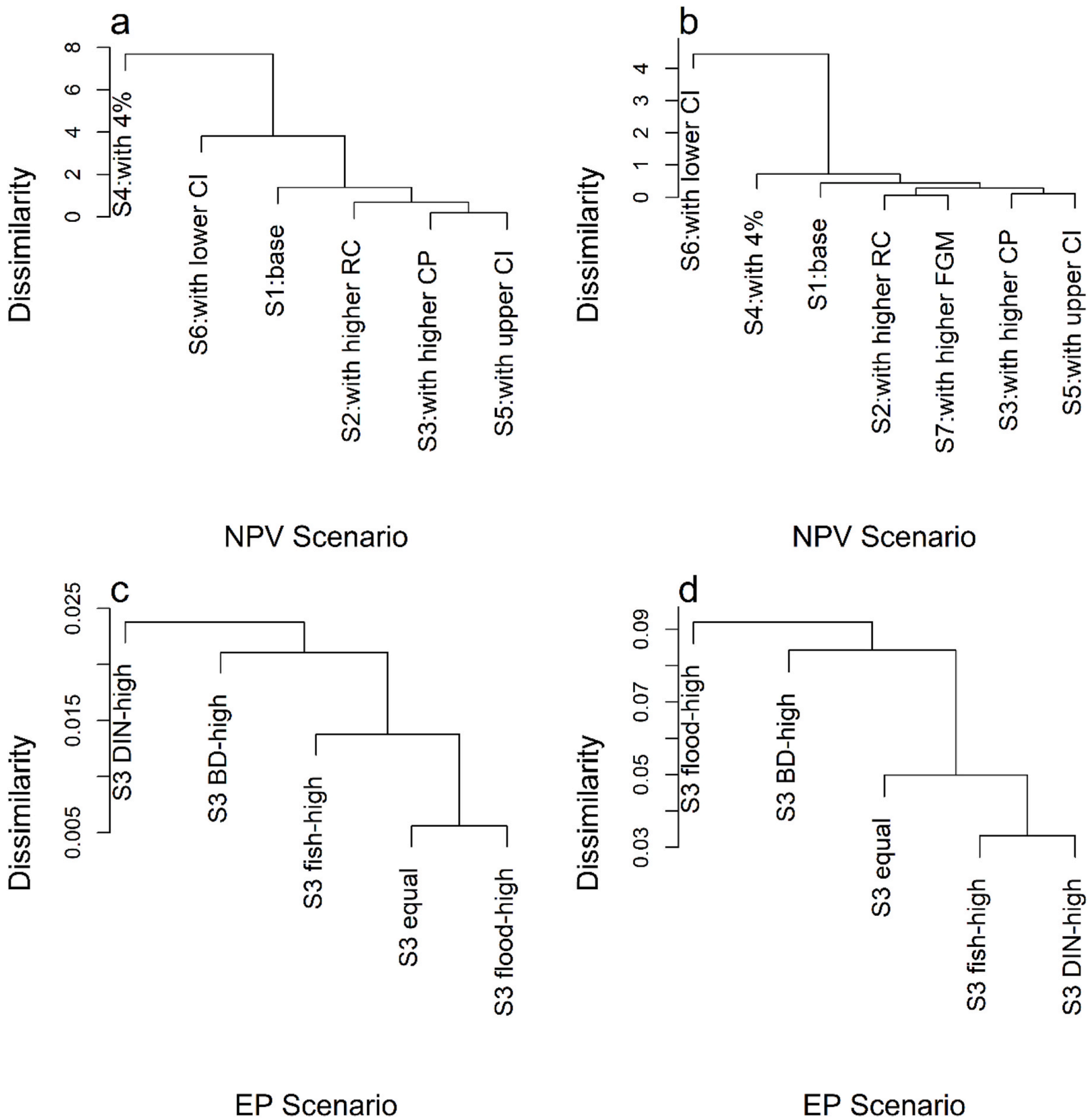

Fig. 5. Dendograms from hierarchical cluster analysis showing similarities among net present value (NPV) and economic prioritisation (EP) scenarios for Fitzroy Basin (a and c, respectively) and Peel-Harvey (b and d, respectively). NPV scenarios: S1 (base scenario): 25 years crediting at 1% discount rate, lower $57 carbon price (CP), lower restoration cost (RC) and farm gross margin (FGM) for beef trading; S2: base with higher restoration cost; S3: base with higher $132 carbon price; S4: base with 4% discount rate; S5: base with upper confidence interval (CI) for estimating carbon sequestration; S6: base with lower CI for estimating carbon sequestration; and S7: base with higher FGM for beef cattle breeding (S7 only applies to Peel-Harvey). Economic prioritisation based on NPV S1 with different weighting combinations of co-benefits (equal weighting of indicators, and higher weighting of biodiversity (BD), fisheries, flood mitigation, or Dissolved Inorganic Nitrogen (DIN) removal indicators). Ordinations shown in Fig. S2.

addressing poverty and unemployment, enhancing connection to Country, culture, and the protection and maintenance of Indigenous food systems. There is an opportunity to address a suite of degraded land and marine environments through activities such as the reduction of feral animal impacts and weed management in blue carbon projects. Projects would need to offer sustainable, long-term funding through ongoing management and monitoring. Traditional Custodians indicated that to create an environment for successful restoration requires the projects to be led by Traditional Custodians with authority and with demonstrated successful governance capabilities. Traditional Custodian-led blue carbon projects will involve a blend of Traditional Knowledge & Science and integrated Western Science applications. Traditional Custodians are interested in projects that allows opportunity to work with their neighbouring groups to develop and support work packages/strategies, bundling multiple restoration projects, and working with the Fitzroy Basin Association and other industry partners. There was an overall aim to 'heal and restore sick Country', deliver sustainable employment and education outcomes, embed funded mentors and Elders into these packages and strategies that enable and enrich knowledge-sharing with an aim to increase understandings of the laws and responsibilities of Country.

In Fitzroy Basin only 696 ha (5%) of the potential restoration sites contain non-exclusive Native Title, where Indigenous people land rights coexist alongside other property rights. In Peel-Harvey, Noongar native title claims are being resolved under The South West Native Title Settlement Agreement including a trust for the purchase of land (Department of the Premier and Cabinet, 2022) and therefore native title was unable to be identified. However, 50 ha of the Peel-Harvey potential restoration sites contained a cultural heritage site, including significant areas of ceremonial sites, burial sites, and artefact scatters. The Noongar people are claiming native title for the region (Department of the Premier and Cabinet, 2022) and resolution of land title disputes may increase restoration opportunities in Peel-Harvey. In the Ord, non-exclusive native title exists across 23,025 ha (95.4%) of the potential restoration sites which may be available for other restoration activities, such as control of feral ungulates.

4. Discussion

The level of restorable area was linked to catchment size, tidal range and extent of grazing land among the regions examined. Fitzroy Basin has the largest catchment area, followed by the Ord and Peel-Harvey regions (Table 3). In Peel-Harvey, tidal ranges are low compared to Fitzroy Basin and the Ord, and the proportion of grazing land-use versus other land-uses within the HAT was much less (2.13%) than Fitzroy Basin (23.5%) and the Ord (100%; Table S2). Levels of carbon abatement were also linked to the extent and type of predicted wetland vegetation, which varies with biophysical factors such as climate, geomorphic settings, sediment, and hydrological regimes as well as ecological processes (Osland et al., 2017; Rovai et al., 2021; Twilley, Rovai and Ruil, 2018). Restorable area may be increased with inclusion of other non-urban land-uses (Worthington and Spalding, 2018). Our consideration of projected sea-level rise to 2100 shows that many sites will expand in area, and new sites will emerge as sea level encroaches. Allowing accommodation space will minimise the effects of coastal squeeze on landward migration of coastal wetlands with sea-level rise (Schuerch et al., 2018). With sea-level rise intensifying land degradation from saltwater intrusion (Haj-Amor et al., 2022), coastal wetland restoration may provide supplementary income to transition businesses through climate-related loss of agricultural production, becoming an option for "climate-smart" farming in the region.

Many potential restoration sites identified were small. In the Ord region we demonstrated the potential for aggregation of small sites across a catchment. Future studies should identify parcels of land that could be aggregated. Traditional Custodians from multiple parties across a region indicated willingness to work together to develop blue carbon projects. This may also be a possibility for farmers who have small parcels of land that could be restored and are interested in carbon projects to diversify their income. The benefits of aggregated agreements are shared costs and expertise to restore landscapes (Canning et al., 2021).

We found the presence of threatened species can significantly modify what land is suitable for restoration. Potential conflicts have also been reported between mangrove restoration and shorebird conservation (Choi et al., 2022), especially for those species whose native habitats have been lost. In Australia, many freshwater wetlands have been drained, and artificial freshwater wetlands behind levees are the only remaining habitat (Waltham et al., 2019). Consideration of potential negative impacts on species is necessary to ensure restoration does not imperil threatened or migratory birds (Choi et al., 2022).

Coastal wetlands in poor condition may have the potential for other restoration activities which may enhance carbon abatement (Macreadie et al., 2017), highlighting the desirability for a suite of blue carbon methods to optimise opportunities in different regions. Introduced ungulate species disturb soils causing significant GHG emissions from pigs globally (O'Bryan et al., 2022) and from sheep in Australia (Limpert, Carnell and Macreadie, 2021). In northern Australia, soil disturbances from grazing could be managed to avoid soil carbon losses (Gehrke, 2009; Robson et al., 2013). In the Ord region, estimated carbon abatement per unit area from reduced grazing pressure was much lower than carbon abatement from tidal restoration for Fitzroy Basin and Peel-Harvey. Conservative abatement estimates in the Ord may be increased with evidence that ungulate animals also reduce woody biomass accumulation or enhance soil methane emissions (Sloane et al., 2021). Across northern Australia there may be large opportunities for blue carbon projects to be Traditional Custodian-led, for example, in the Kimberley region, 85% of the coastal land is native title (National Native Title Tribunal, 2023).

Increasing the carbon price from AU$57 to AU$132 increased the percentage of profitable area by 15% and 7% at Fitzroy Basin and Peel-Harvey, respectively. However, the sensitivity analysis showed that the ranking of sites by NPV was mostly affected by restoration cost, discount rate, farm gross margin, and uncertainty with carbon sequestration rates. Furthermore, high restoration cost and high farm gross margin associated with beef cattle production rendered all sites unprofitable. While tidal restoration of sugarcane land in northern Queensland would be profitable using carbon prices of AU$13 per tonne CO_2-e, this was only the case with high estimates of avoided emissions and low restoration cost (Hagger, Waltham and Lovelock, 2022). Economic feasibility will likely also be affected by costs of multiagency government approvals (Shumway et al., 2021), conversion of other compatible land-uses (Rogers et al., 2023), land acquisition costs (Strassburg et al., 2019), and risks of stochastic events (Wilson et al., 2011), which were not accounted for in this cost-benefit analysis. In these instances, it may be preferable to calculate the breakeven carbon price required to offset costs (Philipson et al., 2020).

Carbon prices are likely to increase over the life of carbon projects, and many carbon credits are expected to be sold at higher prices in the voluntary market (Kuwae et al., 2022), with 30% higher carbon prices obtained for projects with the highest co-benefits (Lou et al., 2022). Delivery of co-benefits is also important for restoration with public funds for additional positive environmental and socio-economic outcomes (Department of Environment and Science, 2023). Recognition of co-benefits in addition to the financial benefits can also increase farmers participation in carbon markets (Fleming et al., 2019). In some regions of Australia, like the Great Barrier Reef catchments and South-east Queensland, nitrogen offsets and markets could provide additional payments for a restoration project that provides additional water purification. Nature markets are also emerging globally, including in Australia, which allow trading of biodiversity credits or certificates (Taskforce on Nature Markets and Pollination, 2023). Across the three regions, there was high potential to restore habitat for nationally endangered species or the temperate saltmarsh community, which could generate separate biodiversity credits or be bundled with carbon credits.

Data gaps, limitations, and future directions

Mapping of hydrological modifications in Western Australia would improve determining suitable land for tidal restoration and accurate data on farm gross margins and tidal restoration costs would improve cost-benefit analysis. Local knowledge of catchment hydrology would also be required to verify seawater ingress using hydrodynamic modelling (Abbott et al., 2020; Karim et al., 2021).

The cost-benefit analysis is based on multiple inputs, including a modelled approach to estimate carbon abatement, which also has multiple parameters that have differing levels of uncertainty. Monte-Carlo simulations found that Australian blue carbon accounting model outputs were similar to the 40th percentile of simulated outputs and the median values adopted for the cost-benefit analysis are therefore conservative estimates of carbon abatement (Lovelock et al., 2022).

Our case studies spanned regions with (1) large and small tidal ranges, (2) different levels of direct hydrological modification, and (3) different co-benefit values. The evaluation of restoration sites can be supported by local data, which limits the potential value of national level assessments. We develop a regional approach to select sites for blue carbon restoration and delivery of co-benefits (Fig. 1). In the UN Decade of Ecosystem Restoration, approaches to facilitate landscape-scale coastal wetland restoration are needed to help achieve the Kunming-Montreal Global Biodiversity Framework goal of 30% of degraded areas under restoration by 2030.

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CRediT authorship contribution statement

Valerie Hagger: Writing – review & editing, Writing – original draft, Visualization, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Phoebe Stewart-Sinclair:** Writing – original draft, Visualization, Investigation, Formal analysis, Data curation. **Renee Anne Rossini:** Writing – original draft, Visualization, Investigation, Formal analysis, Data curation. **Maria Fernanda Adame:** Writing – review & editing, Supervision. **William Glamore:** Writing – review & editing, Supervision. **Paul Lavery:** Writing – review & editing, Supervision. **Nathan J. Waltham:** Writing – review & editing, Supervision. **Catherine E. Lovelock:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Valerie Hagger reports financial support was provided by National Environmental Science Program. Renee Rossini reports financial support was provided by National Environmental Science Program. Phoebe Stewart-Sinclair reports financial support was provided by National Environmental Science Program. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The spatial workflows, datasets and R script for reproducing the analyses for the case study regions have been deposited in eAtlas (https://doi.org/10.26274/76JJ-TK94) (Hagger, Stewart-Sinclair and Rossini, 2024).

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2024.122287.

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