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Coastal Wetland Restoration for Blue Carbon in Australia

Values-based approach for selecting restoration sites

Valerie Hagger, Phoebe Stewart-Sinclair, Renee Rossini, Nathan J. Waltham, Mike Ronan, Maria Fernanda Adame, Paul Lavery, Will Glamore, and Catherine E. Lovelock

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Disclaimer

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Cover image: Bund wall limiting tidal exchange in north Queensland (Image: Nathan Waltham)

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Acknowledgement to Country

The Marine and Coastal Hub acknowledges Aboriginal and Torres Strait Islander people as the first peoples and Traditional Owners and custodians of the land and waterways on which we live and work. We honour and pay our respects to Elders past, present and emerging.

Coastal Wetland Restoration for Blue Carbon in Australia

Valerie Hagger[1], Phoebe Stewart-Sinclair[1], Renee Rossini[2], Nathan J. Waltham[3], Mike Ronan[4], Maria Fernanda Adame[5], Paul Lavery[6], Will Glamore[7], and Catherine E. Lovelock[1]

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Acronyms

ACCU Australian Carbon Credit Units
AGB Above-ground biomass
BGB Below-ground biomass
CE cost-effectiveness
CER Clean Energy Regulator
CH_4 Methane
CO_2-e Carbon dioxide equivalents
DEM Digital Elevation Model
DIN Dissolved Inorganic Nitrogen
ERF Emissions Reduction Fund
GHG Greenhouse gas
GLM Grazing land management
FGM Farm gross margin
HAT Highest Astronomical Tide
NESP National Environmental Science Program
N Nitrogen
N_2O Nitrous oxide
NPV Net present value
NRM Natural resource management

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Executive summary

Diverse forms of coastal wetlands contribute significantly to global carbon stores and climate change mitigation. Opportunities to incentivise coastal wetland restoration by leveraging carbon markets is growing in Australia as methods become available and markets emerge. However, little is known of the feasibility of blue carbon restoration at scale, or the quantities of ecological and social co-benefits that could accompany restoration.

This project quantified the blue carbon restoration potential in three geographically unique case study regions across Australia. It used a multi-stage approach that identified the biophysical suitability of coastal wetland restoration sites, their carbon abatement and co-benefits, and their economic feasibility using a cost-benefit analysis under different carbon pricing and other financial factors. Sites were identified by intersecting restorable land (productive agriculture on historic coastal wetlands) within the highest astronomical tide levels and determining the presence of drains and tidal barriers, and thus with potential for tidal restoration. A set of metrics were identified and quantified for ancillary ecosystem services, or co-benefits, including biodiversity, fisheries, water quality, and coastal protection. Cultural benefits were identified as the potential for leadership and collaboration by Indigenous land managers at sites. Cost-effectiveness analyses were then performed, to identify sites that were profitability under different carbon prices and maximised the provision of co-benefits under different possible stakeholder weightings.

We reveal that identifying blue carbon restoration opportunity is more complex than identifying biophysically suitable sites. Opportunities for blue carbon projects with tidal restoration varies among regions, with variation in tidal range, land-uses, and hydrology impacting abatement forecasts. The presence of threatened species can modify what land is deemed as suitable for restoration, for example from avoiding disturbance to the critically endangered Capricorn Yellow Chat on banked habitats in the Fitzroy Basin region in central Queensland. Or where a restoration site has low profitability or carbon abatement it may be attractive because it supports threatened species and community recovery (e.g. temperate salt marsh in the Peel-Harvey and Purple-crowned fairy wren habitat in the Ord River).

We identified 13,874 ha of land as potentially restorable via the Australian Government blue carbon tidal restoration method across the Fitzroy Basin region with much less in the Peel-Harvey region in south-west Western Australia (348 ha). In the Ord River region in east Kimberley Western Australia, 24,123 ha of land was identified as potentially restorable via restoration activities other than tidal restoration, such as removal of cattle to prevent coastal wetland degradation. Restoration of all potential sites in the Fitzroy Basin and Peel-Harvey would equate to a net carbon abatement of 162,000 Mg CO_2-e yr^{-1} and 4,312 CO_2-e yr^{-1}, respectively. Though not eligible under the blue carbon tidal restoration method, conservative estimates of avoided emissions for the Ord River suggest another 7,237 Mg CO_2-e yr^{-1} could be abated if methods that allow rehabilitation of degraded natural wetlands were available.

Carbon price and permanence period impacted profitability in the Fitzroy Basin, but not Peel-Harvey. Forty-eight Fitzroy Basin sites (51% of area) became profitable under a high carbon price and 21 sites (7% of area) over 100 years, whilst no Peel-Harvey sites were profitable under any scenario because sites were small (additionally input data was limited in this case study region).

Traditional Custodians were interested in restoring coastal wetlands and many wetlands hold significant cultural value. We did not find exclusive Native Title across the potential restoration sites, however there are areas of non-exclusive Native Title in the Fitzroy Basin sites and Native Title is currently being resolved in the Peel-Harvey region, with opportunities for land purchase. There is potential for First Nations people to benefit from long-term blue carbon projects that are led by Traditional Custodians by enhancing connection to Country and culture, protecting cultural sites, and maintaining Indigenous food systems. The Ord River region has high restoration potential but is not eligible under the blue carbon tidal restoration method, and other methods that focus on reducing disturbance and enhancing condition of coastal wetlands might be more useful for Traditional Custodians to engage and implement blue carbon projects.

Our results indicate that high carbon prices are needed to make projects feasible on land used for beef production. A combination of lower restoration costs and lower farm gross margins from reduced agricultural productivity in areas affected by seawater intrusion may create greater incentives for implementing blue carbon projects in the future. There are a range of data deficiencies in accurately predicting the feasibility of blue carbon projects, including higher resolution tidal planes, accurate income data from different land-uses, accurate costing of restoration, and mapping of hydrological modifications.

Importantly, we found that carbon abatement and many co-benefit metrics covaried and that trade-offs were limited. Therefore, there is potential to find hotspots where multiple ecosystem services can be bundled to attain higher carbon prices for restoration projects with co-benefits, or to undertake projects under other emerging markets, such as biodiversity stewardship. Increasing the range of case study regions that encompass different characteristics would help in understanding potential trade-offs.

There is significant opportunity to incentivise coastal wetland restoration under blue carbon markets to enhance Australia's carbon stocks and associated co-benefits. Our analysis highlights that identifying potential sites must go beyond biophysical attributes and consider the range of factors that will impact the long-term profitability and sustainability. It also highlights the rich ecological and cultural benefits that could be produced. Each region was unique in the characteristics of sites, the data available to inform decision making, and profitability outcomes. Our research indicates that a framework to select sites for coastal wetland restoration for blue carbon in Australia should be regionally specific, creating an approach that inherently engages Traditional Custodians and incorporates local knowledge.

1. Introduction

Coastal wetlands, including mangroves, saltmarshes, seagrasses and tidally influenced floodplain forests dominated by *Melaleuca* and *Casuarina spp.* sequester high amounts of atmospheric carbon dioxide in their soils and biomass, contributing to blue carbon stocks and climate change mitigation (Adame et al., 2020; Serrano et al., 2019). They also provide vital ecosystem services that benefit coastal populations, including fisheries production, pollutant removal, 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 habitats as nursery grounds and foraging habitat (Nagelkerken et al., 2015; Rog et al., 2020). Yet, large losses of coastal wetlands have occurred across Australia, particularly from drainage, infilling and flood mitigation works as part of agricultural, industry and urban expansion (Rogers et al., 2016). Sustained funding models that demonstrate a return on investment are needed to attract government and private capital to accelerate coastal wetland restoration efforts and support the goals of the United Nations (UN) Decade on Ecosystem Restoration and the UN Decade of Ocean Science for Sustainable Development (Waltham et al., 2020).

The Australian Government Clean Energy Regulator (CER) recently developed a blue carbon method to characterise the climate change mitigation benefits (or carbon abatement) of restoring coastal wetlands, where restoration of tidal flows and subsequent coastal wetland ecosystem recovery can be awarded Australian Carbon Credit Units (ACCUs) that can be sold for carbon offsets (Clean Energy Regulator, 2022). Identifying opportunities for the use of the blue carbon tidal restoration method could stimulate its uptake, which can provide a wide range of ecosystem services as well as opportunities for landholders to diversify the income from their land. This project aimed to develop a process to identify sites suitable for coastal wetland restoration across Australia based on a value-based framework that considers biophysical suitability for restoration, wetland co-benefits (biodiversity, fisheries, water quality, coastal protection, and Indigenous heritage), potential benefits to Traditional Custodians, regulation and policy constraints, and economic feasibility.

Australia has large blue carbon stocks many of which have been degraded or converted to alternative land-uses since European colonisation (Serrano et al., 2019). For example, mangroves, saltmarshes and *Melaleuca* forests have been converted to grazing and sugarcane land (Hagger et al., 2022). These altered floodplain landscapes may provide opportunities for coastal wetland restoration for ACCUs (for projects that follow the tidal restoration method) that provide incentives for landholders and Indigenous land managers to undertake coastal wetland restoration. However, the factors influencing the opportunity for coastal wetland restoration vary across Australia's coastline because of variation in land-uses, the levels of carbon abatement that could be achieved (Kelleway et al., 2017), and the variation in laws and policies that regulate land-use change and land ownership (Bell-James et al., 2022; Shumway et al., 2021). Additionally, the provision of other ecosystem services of coastal wetland restoration varies regionally and locally (Adame et al., 2015; Ouyang et al., 2018). Trade-offs among different ecosystems services (e.g. carbon abatement vs. biodiversity) have been documented in forests and mangroves (Hua et al., 2022; Uddin et al., 2022), and should be considered for coastal wetland restoration projects. Thus, an analysis of the variation in economic feasibility of coastal wetland restoration for blue carbon over regions with different land uses, varying levels of potential for carbon abatement, other ecosystem services, and regulatory contexts can provide insights to target the development of coastal wetland restoration projects.

An assessment of opportunities for coastal wetland restoration for blue carbon in the Wet Tropics region of north Queensland found large areas (4,534 ha) of low-lying sugarcane and grazing land that could be economically feasible for restoration using a carbon price of AU$25 per tonne (Mg) of CO_2-e. The study considered carbon sequestration in vegetation and soils, and avoided greenhouse gas emissions from ceasing the baseline land-use (Hagger et al., 2022). This project builds on this earlier assessment to include additional factors that influence the suitability of sites for coastal wetland restoration. Here, we: 1) identify the opportunity for restoration for blue carbon in other climatic and land-use contexts in northern and western Australia, 2) refine land suitable for restoration through a value-based framework that considers co-benefits, including opportunities for Traditional Custodian-led blue carbon projects to generate cultural benefits, and 3) recognises the variation in policy and regulations among regions for identifying sites with high potential for restoration.

We applied our analyses in three case study regions that vary widely in their climatic characteristics and land-uses, coastal wetland types, potential blue carbon abatement, and which were in different states, giving rise to variation in regulatory contexts. These included the Fitzroy Basin in central-eastern Queensland (QLD), Peel-Harvey and South West catchments in south-west Western Australia (WA), and the Ord River floodplain in east Kimberley, WA. Firstly, for each case study region we used available biophysical data to identify opportunities for tidal restoration of coastal wetlands through modification of drains and/or tidal exclusion structures to allow tidal reintroduction. We estimated the carbon abatement from carbon sequestration in biomass and soils and avoided emissions following the blue carbon tidal restoration method, which uses Australian specific emission factors and regional carbon data (Kelleway et al., 2017; Lovelock et al., In Review).

After this initial biophysical assessment of the opportunity for coastal wetland restoration, we met with local stakeholders to discuss, verify, and refine the outcomes based on local knowledge and priorities. Stakeholder meetings included representatives from natural resource management (NRM) groups, government agencies, conservation organisations, and Indigenous groups. Stakeholder meetings allowed us to gather information on local priorities for coastal wetland restoration, data availability, Indigenous heritage values, land ownership and regulatory constraints, as well as facilitating knowledge sharing among stakeholders. Of special interest was the alignment of coastal wetland restoration within existing government priorities, such as catchment water quality improvement plans, and the inclusion of Traditional Custodians in decision-making to support partnerships. After incorporating the information gained from the stakeholder meetings, we identified sites that were economically feasible using a cost-benefit analysis which considered the financial benefit from carbon abatement (the sale of ACCUs), restoration and maintenance costs, and forgone income from ceasing the existing agricultural land-use. We assessed the sensitivity of economic feasibility to varying conditions, including variation in the carbon price, discount rate, restoration cost, farm gross margin, and permanence period (Hagger et al., 2022; Stewart-Sinclair et al., 2021). We then quantified the potential benefit of restoring sites to enhance biodiversity and fisheries, improve water quality, and provide coastal protection using indicators based on the availability of spatial data and stakeholder values.

The variation in case study regions, the process of consultation with stakeholders, the varying wetland values, and the economic feasibility assessments underpinned the development of an approach for selecting sites for coastal wetland restoration that comprises a holistic approach. This approach informs where sites are economically feasible and valued for biodiversity, fisheries, water quality, coastal protection, and Indigenous culture. Our use of case studies facilitated a comparative approach, which identified deficits in data availability among regions. Additionally, this approach considers variation in restoration opportunities under different climatic and land-use contexts, and identification of trade-offs between carbon abatement and other ecosystem services.

2. Methods

2.1 Study regions

We chose three case study regions in Australia with different climates, farming systems, pressures, ecosystem services, and potential carbon abatement. The regions were the Fitzroy Basin in central-east QLD, the Peel-Harvey and South West catchments in south-west WA and the Ord River floodplain in east Kimberly, north-east WA (Figure 1).

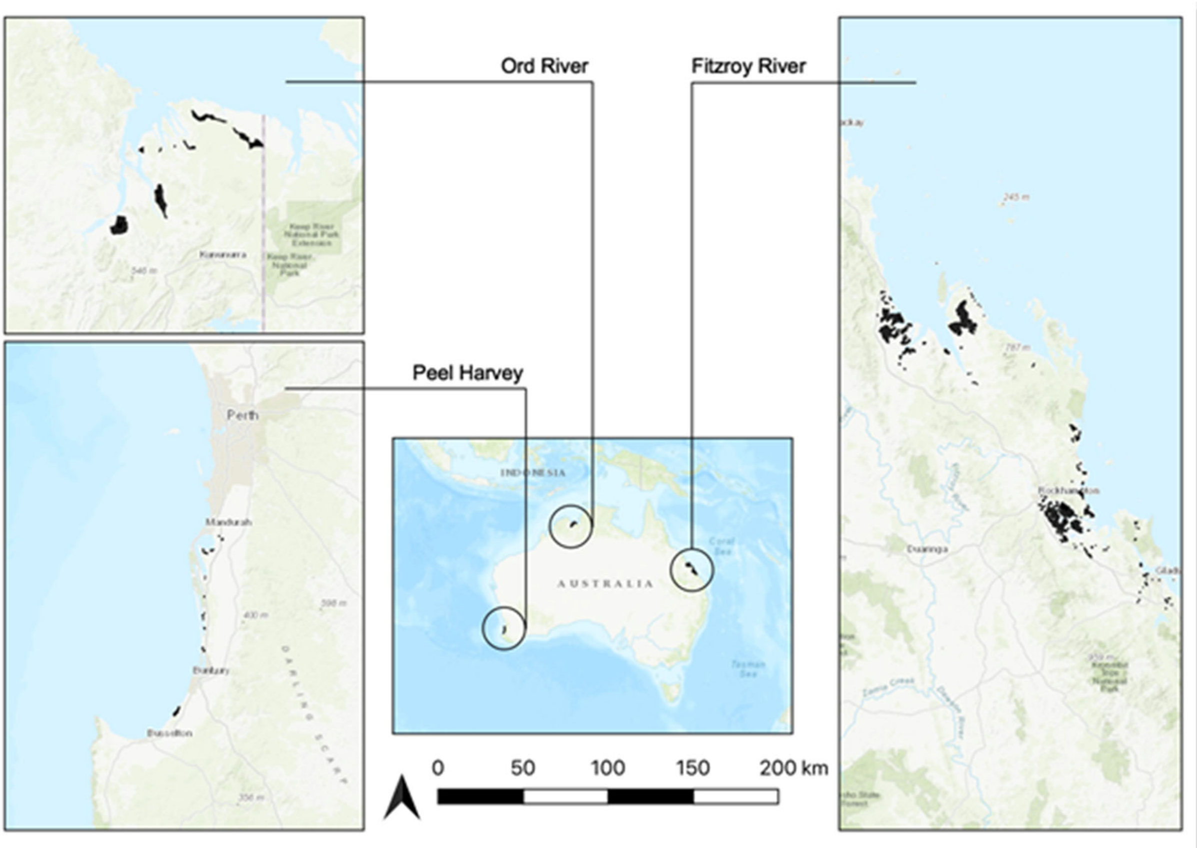

Figure 1 The Australian continent (centre) with the three case study regions circled. Each case study region expanded with potential coastal wetland restoration areas in black. The scale applies to the expanded regional maps.

2.1.1 Fitzroy Basin, Queensland

The Fitzroy Basin NRM region (Figure 2) comprises multiple catchments of the World Heritage listed southern Great Barrier Reef catchment, including the Styx, Shoalwater, Waterpark, Fitzroy, Calliope, Curtis Island, and Boyne, covering 156,000 km^2 of land. Open estuaries in this region are characterised by high water exchange with the ocean and nearby Broad Sound, Shoalwater, Keppel and Corrio Bays due to the large tidal range (up to 10m). The Fitzroy Basin encompasses Darumbal and Koinjmal land and sea Country. The Styx River spans both nations, whilst the Fitzroy River connects to other nations in its upper catchments including Baradah, Gabalbara and Gangalu (AIATSIS).

The region is located in the central Mackay coast bioregion and includes a coastal portion of the Brigalow belt north bioregion (DAWE 2012). It has summer dominant rainfall, ranging between 650-1200 mm median annual and a moderately dry winter (Bureau of Meterology, 2022). The hydrology and drainage of the Fitzroy Basin has been highly modified with 59 dams, weirs and barrages acting as barriers to fish migration (Marsden, 2015). Tides in the region are semidiurnal with ranges between 2.2 - 4.1 m for the Fitzroy River and 3 - 6.6 m for the Styx (Maritime Safety Queensland, 2021) (Table A3). The coastal wetlands in the region are dominated by mangroves low in the intertidal transitioning to scrub mangroves, saltmarsh, and salt flats at the higher intertidal zone (Bunt & Bunt, 1999). *Melaleuca* spp. and *Eucalyptus* spp. swamps occur at higher elevations, adjacent to the mangroves and saltmarsh (Neldner, 2017).

The Fitzroy Basin is the largest region draining into the Great Barrier Reef and has targets for sediment load reduction (25% by 2025) to improve water quality, however progress has been poor (cumulative reduction of 10.3% to June 2020) (Australian Government, 2020). The major town of Rockhampton (population 80,665 in 2020) is located in the south. Land use is predominantly grazing, followed by nature conservation and Indigenous Protected Areas (Merrin et al., 2018). In 2017, 83% of all wetlands in the region were considered modified (Department of Environment and Science, 2022a). Modifications included construction of bund walls to exclude tidal flows and hold freshwater (including introduced pasture species such as para grass (*Urochloa mutica*) and hymenachne (*Hymenachne amplexicaulis*), creating “ponded pastures” which are important for growing fodder for cattle during dry periods (Bell-James & Lovelock, 2019; Fitzroy Basin Association, 2015).

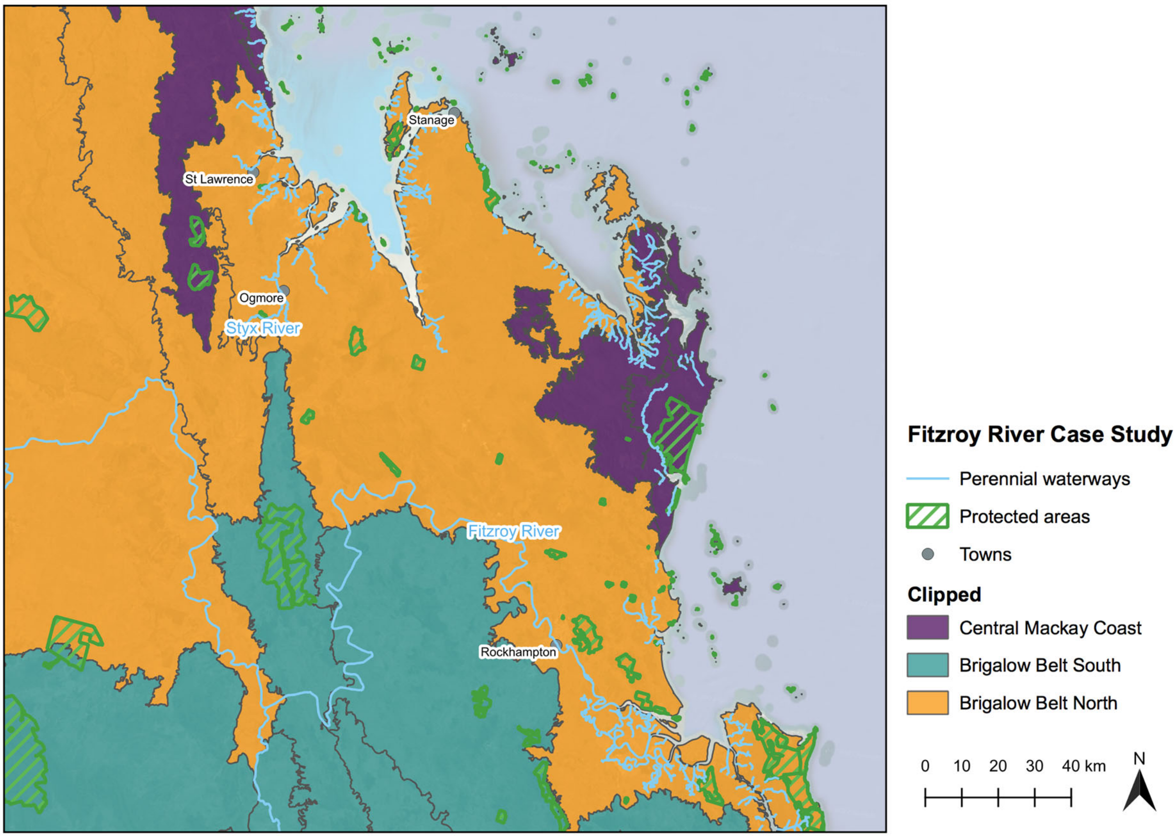

Figure 2 The Fitzroy Basin study region

2.1.2 Peel-Harvey

The Peel-Harvey and South West region (Figure 3) encompass the land and sea Country of Wardani, Kaniyang, Pinjarup and Wajuk (AIATSIS). It has broader connections to Wiilman Country via the upper catchments of rivers that feed into the Peel estuary. Within the region, the South West Native Title settlement includes the Gnaala Karla Booja and South West Boojarah #2 agreements (Department of the Premier and Cabinet, 2022).

The study region sits within the Swan Coast Plain bioregion (Department of Agriculture Water and the Environment, 2012). It has winter-dominant rainfall of >800 mm median annually and encompasses temperate climate zones with distinctly dry and warm summers (Bureau of Meterology, 2022). Tides are diurnal and microtidal, with a tidal range of 0.6 m (Valesini, 2010). Changes in atmospheric pressure and rainfall can induce changes in water levels in the region's estuaries that are comparable with tidal variations. The region has closed and semi-enclosed estuaries, some with large bays such as the Peel Inlet and Leschenault Estuary (Figure 3). Major rivers include the Capel, Preston, Collie, Brunswick, Harvey, Murray, and Serpentine. There are also several closed coastal lakes, the largest being Lake Clifton and Lake Preston. Ocean influences are restricted in this region due to a small tidal range and closed estuaries.

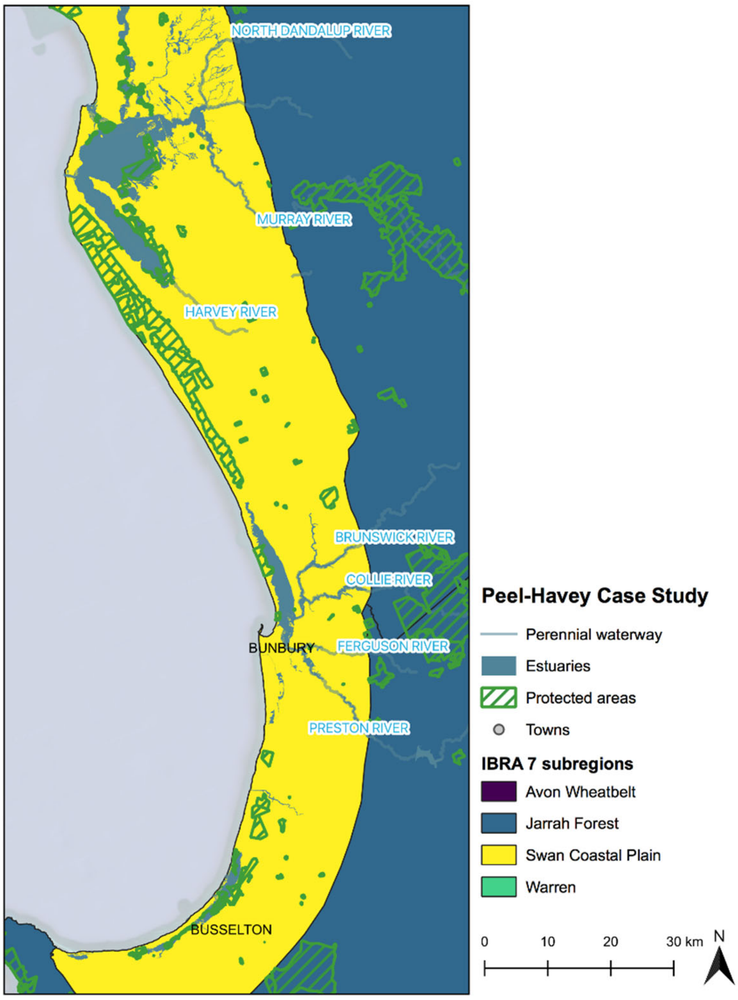

Figure 3 The Peel-Harvey and South West study region

The lowest intertidal areas are primarily halophile and sarcophyll communities of *Tecticornia spp.* and tidal mud flats, progressing into near coastal woodlands and shrublands including *Eucalyptus spp., Corymbia spp., Acacia spp.,* and *Callitris spp.* (Beard et al., 2013). A small stand of mangroves (*Avicennia marina,* approximately 10 ha) occurs in the Leschenault Inlet in the town of Bunbury as well as scattered individuals around the estuary (Semeniuk, 2000).

The study region falls across Peel-Harvey Catchment Council and the northern part of the South West Catchment Council and includes the coastal towns of Bunbury (population 85,620 in 2020) and Busselton (population 40,333 in 2020). In the Peel-Harvey, the main land-uses are native vegetation (47%), cropping (32%) and beef cattle (9.3%) (Kelsey et al., 2011). In the South West, 64% of people considered primary production as the main land-use, primarily beef cattle and dryland pasture (Department of Agriculture and Food Western Australia et al., 2006).

2.1.3 Ord River

The Ord River region (Figure 4) spans the land and sea Country of the Doolboong, Kadjerong and Yiiji nations, with upper catchment connections to Miriwoong Country, and falls within the Miriuwung Gajerrong native title determination.

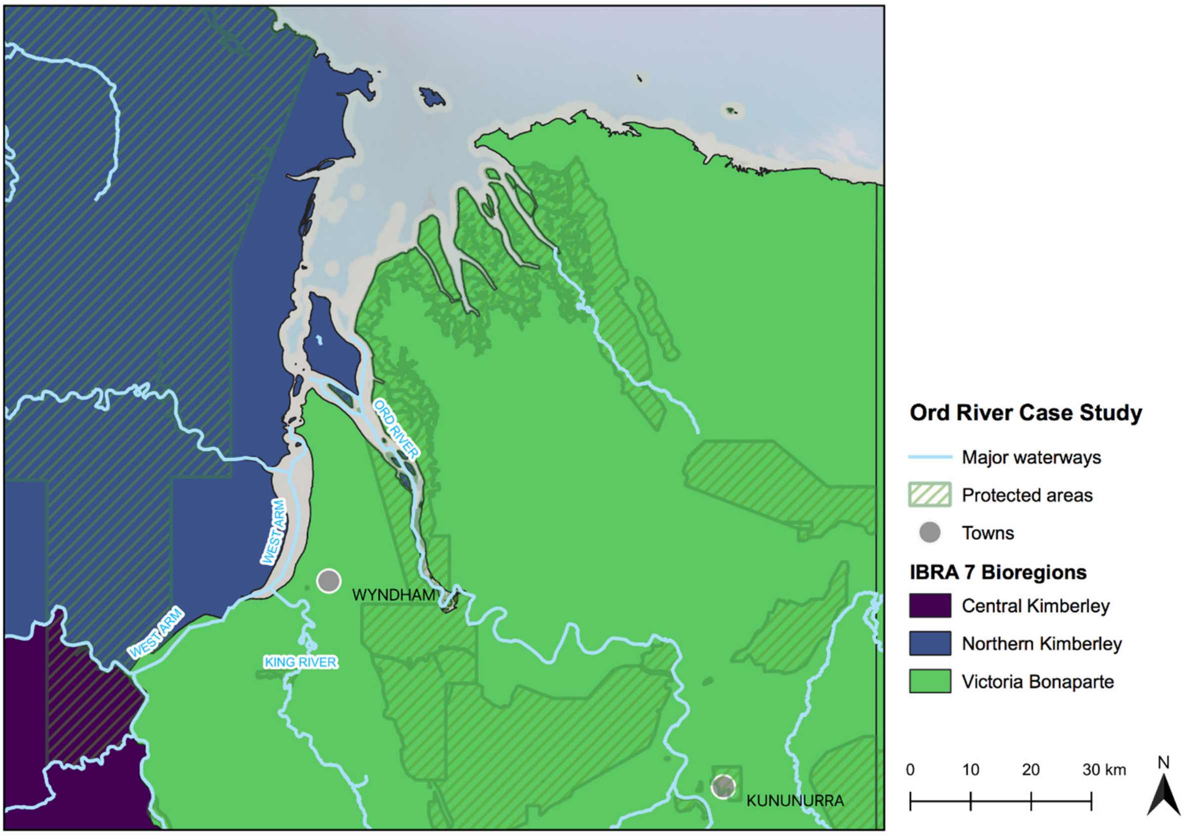

Figure 4 The Ord River study region

The study region is primarily within the Victoria Bonaparte bioregion, though the western side of the estuary is the Northern Kimberley bioregion. It has a summer-dominant rainfall of 650-1200 mm median annually (Bureau of Meterology, 2022) with a climate characteristic of savannah tropics. It is centred on the Cambridge Gulf and surrounding coast to the east and includes the Ord and King Rivers, as well as smaller tidal creeks and estuaries in the north-east. The study region is macrotidal with a tidal range of 7-9 m, meaning the influence and exchange with the ocean is high (Wright, 1973).

Coastal wetlands in this region are primarily mangrove, halophile and sarcophyll communities of *Tecticornia spp.* and tidal mud flats, and the north-eastern area includes coastal bunch grasslands progressing into riparian vegetation dominated by *Eucalyptus spp.* and *Corymbia spp.* (Beard et al., 2013).

The Ord River is the most remote of our three case study regions, with Kununurra the largest town in the south-east of the region (population 5,308 in 2020). It falls within the largest NRM region in Australia – the Rangelands NRM Coordinating Group – but is also under management and coordination of the Kimberley Land Council. The study region encompasses the Ord River Irrigation Area, comprising 22,000 ha of irrigated agriculture with water fed from the Ord River diversion dam and Argyle dam, with agricultural development continuing under the developing Northern Australia agenda (CRCNA, 2020).

2.2 Stakeholder engagement

We undertook workshops with stakeholders to discuss preliminary findings of the case studies and explore the opportunities for blue carbon projects to contribute to the development of a framework for identifying land suitable for restoration. For the Fitzroy Basin, we met with representatives of Fitzroy Basin Association, QLD Government Land Restoration Fund, and Traditional Custodian groups. For Peel Harvey, we met with representatives of Peel-Harvey Catchments Council, South West Catchments Council, WA Government agencies and non-government organisations. The aims of the workshops were to: (1) validate the restoration areas identified, (2) discuss agricultural land uses and current management practices, (3) identify restoration priorities and existing coastal and marine projects, (4) determine which ecosystem services were valued by stakeholders, (5) assess the representation and adequacy of the indicators selected for the ecosystem services, and the potential for inclusion of cultural heritage values, (6) discuss the opportunity for Traditional Custodian-led carbon projects, and (7) ascertain data availability to inform the identification of land for a restoration framework. We attempted to hold a workshop for Ord River, however it was not possible in the study timeframe. Initial discussions were held with Kimberley Land Council and Northern Land Council in the Indigenous Carbon Industry Network session described below.

We held a separate workshop with Traditional Custodian representatives along the coast of Fitzroy Basin including the Darumbal Enterprises, Darumbal People Aboriginal Corporation, Port Curtis Coral Coast Trust and Koinmerburra Aboriginal Corporation, to further explore opportunities and challenges for Traditional Custodian-led blue carbon projects. An information session was also held with 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. The objective of the session was to provide an overview of the project and the new blue carbon method to explore opportunities and challenges for blue carbon projects in northern Australia.

Other engagement included discussions with North Australia Indigenous Land and Sea Management Alliance on new carbon abatement methods. These include a proposed method for reducing disturbance to soil and vegetation by controlling feral buffalo, cattle, and other feral animals, and thereby avoiding emissions associated with coastal wetland degradation. The opportunity for this proposed carbon abatement method (and cost benefit analysis) was not fully assessed in all case study regions, but was explored in a preliminary analysis in the Ord River case study. However, the data collected and assembled in this project, as well as other data, including models of feral animal distribution, knowledge of control activities, and alignment with other national strategies, such as the National Feral Pig Action Plan 2021-2031 (Australian Pork Ltd, 2021) could be used to further elaborate opportunities for restoring the condition of coastal wetlands for carbon abatement in the future.

2.3 Identify restoration sites

We identified land uses that could be suitable for restoration from the regional land-use mapping programs (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 study regions. For Fitzroy Basin we also included Defence land in non-remnant areas extracted from the QLD regional ecosystem mapping (Department of Environment and Science, 2019a) because a large area of the region is occupied by Shoalwater Bay Training Area, as well as areas of wetlands used for agricultural production land on freehold or leasehold land extracted from the QLD cadastral data (Department of Resources, 2021a), and the location of hydrologically modified wetlands that are used as ponded pastures (H2M2, H2M3, and H2M5) from the QLD wetlands data (Department of Environment and Science, 2020c). There is no data for hydrological modified wetlands in WA. The Australian Land Use and Management Classification codes selected for each study region are given in Table A1. Initially, we explored the inclusion of cropping (3.3 and 4.3) and abandoned intensive animal production (5.2.8), however these land uses were small within the Highest Astronomical Tide (HAT) elevation contour and thus excluded from the analysis (Table A2). We have therefore focussed on assessing the economic feasibility of restoring land currently used for grazing.

We selected land use parcels that were ≥ 1 ha within the HAT elevation contour (based on Australian Height Datum, AHD), and could theoretically receive tidal waters, and historically had coastal wetland vegetation, which was determined from pre-clear or pre-European vegetation mapping (Department of Environment and Science, 2019a; Department of Primary Industries and Regional Development, 2017). Vegetation types selected for each study region encompass mangrove, saltmarsh, and supratidal forest comprising *Melaleuca*, *Eucalyptus*, *Casuarina* or *Acacia* spp (Table A3). We included land mapped as riverine forest, swamps, vine forest, sedgelands, and open water prior to clearing/settlement and occurred within the HAT. Our rationale was that agriculture can result in subsidence and compaction of organic soils (White & Kaplan, 2017), thus these areas may transition to supratidal wetlands. We assumed natural recovery of vegetation would follow reintroduction of tidal flows given the large areas of natural coastal wetland vegetation in the case study regions (Lovelock et al., In Review). Land that was mapped as floodplain forests and sedgeland prior to clearing and occurred within the HAT level were assumed to transition to the supratidal forest category used in the blue carbon tidal restoration method (e.g. *Melaleuca, Casuarina*) and saltmarsh depending on region (Lovelock et al., In Review).

The HAT layer was developed for each region using the mean sea level (MSL) and HAT tide predictions for standard ports and locations (Keysers et al., 2012; Maritime Safety Queensland, 2021) and a Digital Elevation Model (DEM; 5 m resolution) (Geoscience Australia, 2015). For the Fitzroy Basin, the intertidal zones were defined according to tide levels: low (MSL – mean high water neaps, MHWN), mid (MHWN – mean high water springs, MHWS) and high (MHWS – HAT). Average tide levels were obtained for predictions located in each catchment (Table A4; note an allowance of 2.5 mm per year for sea level change has been made in the MSL estimate for QLD tidal planes). The DEM was clipped for each catchment, reclassified to reflect average heights of the intertidal zones, converted to a polygon, and merged to create a single shapefile. For the Peel-Harvey and Ord River, the data available did not provide tide levels for MHWN and MHWS, therefore only a HAT layer was developed.

We also explored the potential for coastal wetland restoration opportunity with sea-level rise by applying an additional 1 m to the HAT in the DEM, which approximates higher sea level rise by 2100 (Lovelock et al., In Review) and 0.7 m which approximates sea level rise under RCP 8.5 by 2100 (Oppenheimer et al., 2019). Carbon sequestered is regarded as permanent if it is maintained for 100 years, which encompasses the projected time frame of rises in sea-level. Projects under the Emissions Reduction Fund (ERF) can choose a 25- or 100-year permanence period, although it is anticipated that blue carbon projects would continue for 100 years, even if landholders initially enter into a 25-year agreement.

All spatial analysis was undertaken in ArcMap 10.8 (ESRI, 2019) and QGIS (Open-source software, 2002) and data analysis in R 4.0.2 (R Core Team, 2020).

The restoration sites identified in the Ord River are all situated on sites with grazing of native vegetation, which is mapped as remnant vegetation, and not cropping, horticulture or pasture associated with the Ord River Irrigation Scheme. The potential for tidal restoration of these areas is probably limited and the economic analysis under the blue carbon tidal restoration method has therefore not been conducted due to uncertainty with estimating carbon abatement and the condition of the wetlands. Rather the opportunity for restoration under potential blue carbon methods, such as management of feral ungulates, and the co-benefits of the sites from avoiding disturbance of soils and vegetation are evaluated.

2.4 Estimate carbon abatement

2.4.1 Avoided greenhouse gas emissions from baseline land use

The avoided greenhouse gas (GHG) emissions from ceasing grazing land use were estimated following the Australian Government blue carbon tidal restoration method (Lovelock et al., In Review) (Table 1; calculation methods are in Table S.1.). Methane (CH_4)

and nitrous oxide (N_2O) emissions from flooded agricultural land and ponds and other constructed water bodies were estimated by applying nationally-derived emission factors. Soil carbon accumulation for degraded wetlands (tidally restricted wetlands and supratidal forests) on grazing land use were accounted for using national default values derived from hydrologically disturbed mangroves, saltmarsh and herbaceous settings. The removals from the degraded wetlands were converted to CO_2 and deducted from the baseline emissions per year in CO_2-e (equivalents).

Some activities from grazing that are outlined in the International Panel of Climate Change (IPCC) guidelines (Klein et al., 2006; Lovelock et al., 2019; Verchot et al., 2006) were excluded, including N_2O emissions from fertiliser application and urine and dung deposition, and CH_4 emissions from canals and drains. In grazing, these activities contribute approximately 10% of avoided emissions (Hagger et al., 2022), however, ACCUs can only be paid for activities outlined in the blue carbon tidal restoration method.

For the Fitzroy Basin, hydrologically modified wetlands classed as lacustrine, palustrine, and riverine (Department of Environment and Science, 2020c) were used to identify flooded agricultural land within each site. Ponds and other constructed water bodies within the HAT (without the SLR predictions) were identified from a combination of water storage points and reservoir data (Department of Resources, 2021c, 2021e). Water storage points classed as dams were selected and a radius of 14.1 m was applied around the points to reflect the average water storage size in QLD of 625 m^2 at full supply level (Malerba et al., 2021) and merged with the reservoir shapefile. Degraded wetlands were identified from a combination of mature regrowth and remnant vegetation (Department of Environment and Science, 2019a, 2020a) classed as wetland BVGs: 35a and 35b (tidally-restricted wetland) and 22a, 22b and 22c (supratidal forest).

In the Peel Harvey region of WA, pond and farm dam areas within grazing areas within the HAT were mapped using farm dam data and degraded wetlands were identified from remnant vegetation (Department of Primary Industries and Regional Development, 2020a, 2020b).

2.4.2 Greenhouse gas removals and emissions from coastal wetlands

The GHG removals and emissions from restoration were estimated following the blue carbon tidal restoration method (Lovelock et al., In Review) (Table 1; calculation methods in Table A5). Carbon sequestration in soils of potential restoration sites were estimated using national default values for mangroves, saltmarsh and supratidal forest. Above-ground biomass (AGB) carbon accumulation up to 100 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 (BGB) was estimated using the proportion of AGB to BGB (root shoot ratio, R:S) for mangroves, saltmarsh and supratidal forest. The CH_4 and N_2O emissions from flooded coastal wetlands were estimated using nationally-derived emission factors for different climate regions in Australian coastal wetlands. Carbon accumulation was assumed to initiate when natural vegetation becomes established, in year one after tidal flow is reinstated (year 0). In the method, an AGB and soil carbon multiplier of 0.7 is suggested for scrub mangroves in tropical climates in a standard tidal position index (STPI) of 0.32 - 1.0. We did not apply the multiplier to Fitzroy Basin, because of uncertainty in elevation data and the transition to scrub mangroves in the mid-high intertidal zones. Carbon accumulation was estimated over 25 and 100 years and converted to CO_2, and emissions per year were deducted from the wetland removals in CO_2-e.

The carbon abatement per site was calculated as the sum of the wetland removals and the baseline emissions per period (as CO_2-e).

Table 1 Greenhouse gas (GHG) emissions and removals estimated for baseline land uses and coastal wetlands

Land use / wetland	Emission / stock	Activity	Emission Factor (kg ha^{-1} yr^{-1})	Removal Factor (Mg C ha^{-1} yr^{-1})	Method
Baseline land uses - emissions					
Grazing	CH_4	Flooded agricultural land, managed wet meadow or pasture	325.0	-	IPCC Tier 2 - Median values of methane and nitrous oxide emissions from Australian coastal land published and unpublished data (Lovelock et al. 2021).
	N_2O	Flooded agricultural land, managed wet meadow or pasture	14.0	-	
	CH_4	Ponds and other constructed water bodies	226.3	-	
	C	Soil carbon loss	Table S.1	-	IPCC Tier 1/2 – default stock change factors (IPCC 2006) applied to site-specific soil organic carbon stocks from Australian baseline map of soil organic carbon (Viscarra-Rossell et al. 2014)
Baseline land uses - removals					
Tidally-restricted wetland (freshwater or brackish)	C	Soil carbon accumulation in hydrologically disturbed mangrove, saltmarsh, and herbaceous settings		0.47	IPCC Tier 2 – national default value from Kelleway et al. unpublished data and Jones, Lavery et al. unpublished data (Lovelock et al. 2021)
Supratidal forest	C	Soil carbon accumulation in disturbed supratidal forest		0.61	IPCC Tier 1/2 – national default value (Lovelock et al. 2021)
Coastal wetlands – removals					
Mangrove	C	Soil carbon accumulation	-	0.95	Tier 2 – national default values from Serrano et al. 2019, updated to include recently published and
Saltmarsh	C	Soil carbon accumulation	-	0.48	

Land use / wetland	Emission / stock	Activity	Emission Factor (kg ha^{-1} yr^{-1})	Removal Factor (Mg C ha^{-1} yr^{-1})	Method
Supratidal forest	C	Soil carbon accumulation	-	0.61	unpublished datasets (Lovelock et al. 2021)
Mangrove	C	Below-ground biomass carbon accumulation		R:S of 0.32	ratio to aboveground biomass (Lovelock et al. 2021)
Saltmarsh	C	Below-ground biomass carbon accumulation	-	R:S of 0	
Melaleuca	C	Below-ground biomass carbon accumulation		R:S of 0.27	
Mangrove	C	Above-ground biomass carbon accumulation	-	167 max (tropical); 70.4 max (temperate)	Tier 3 – log model of AGB carbon accumulation (Table S.2) from mature stock values for mangroves in tropical (humid and monsoon) and temperate Australia (Lovelock et al. 2021)
Saltmarsh	C	Above-ground biomass carbon accumulation ≤ 1 year	-	1.36 max (tropical); 7.89 max (temperate)	Tier 3 – mature stock values for saltmarsh in tropical (assumes subtropical value as no data for tropical) and temperate Australia grown in the first year (Lovelock et al. 2021).
Supratidal forest	C	Above-ground biomass carbon accumulation	-	192 max (tropical); 178 (temperate)	Tier 3 – log model of AGB carbon accumulation (Table S.2) from mature stocks for supratidal forest in tropical (humid and monsoon) and temperate Australia (Lovelock et al. 2021)
Coastal wetlands - emissions					
Mangrove	CH_4	Flooding of mangroves	2.19	-	Tier 2 – Median values of methane and nitrous oxide emissions from tropical (humid) climate region in Australian coastal wetlands from published and unpublished data (Lovelock et al. 2021).
	N_2O	Flooding of wetlands	0.24	-	
Saltmarsh	CH_4	Flooding of wetlands	0.11	-	
	N_2O	Flooding of wetlands	0.13	-	
Supratidal forest	CH_4	Flooding of wetlands	-2.19	-	
	N_2O	Flooding of wetlands	0.25	-	

2.4.3 Approach for estimating carbon abatement with reduced grazing

Although there is no ERF blue carbon method for estimating carbon abatement with management of grazing in wetlands, preliminary estimates of abatement for rehabilitation of coastal wetlands of the Ord River were made using information from Gehrke (2009) and Robson et al. (2013) and parameters within BlueCAM (Lovelock et al., In Review). We focussed on the potential for carbon abatement by removing grazing pressure on the lands of the Lower Ord River. These have been characterised into different zones: (1) freshwater riverine, (2) tidally influenced freshwater zones of the landscape (meandering sections of the lower Ord), characterised by grasses such as *Paspalum, Cynodon, Phragmites* and *Typha*, and trees such as *Melaleuca*, *Pandanus* and *Barringtonia*, that can be characterised as supratidal coastal wetlands, (3) transition zone, with vegetation that is tolerance of brackish conditions (saltmarsh, mangroves, supratidal forests), (4) estuary mouth zone, and (5) tidal creeks and flats zone that are dominated by fringing mangrove communities (Beard, 1967; Gehrke, 2009; Robson et al., 2013).

Land uses such as mining, grazing, and cropping greatly increase sediment yields above the low background values in the Ord River region (Gehrke, 2009). For example, 23.5 x 10^6 t year^{-1} of sediment entering Lake Argyle was attributed to gully erosion exacerbated by overgrazing (Wark, 1987; Wasson et al., 2002). Gehrke (2009) reports that rates of soil erosion tend to decline during the wet season because of the increased ground cover provided by grass (Williams, 1969). However, if grass cover is low due to low rainfall or grazing pressure then periods of heavy rainfall led to increased erosion and increased sediment delivery to the river (Paul Novelly pers. comm. in Gehrke, 2009). These observations suggest that reducing grazing pressure can reduce erosion, which could provide avoided CO_2 emissions, as well as avoided N_2O emissions associated with grazing (Lovelock et al., In Review).

To estimate conservative levels of carbon abatement we used an avoided emission of 0.3 CO_2-e ha^{-1} year^{-1} (0.1 CO_2-e ha^{-1} yr^{-1} from avoiding N_2O and 0.2 CO_2-e ha^{-1} yr^{-1} from avoided soil organic carbon losses). Levels of abatement could be increased with evidence that grazing influences woody biomass accumulation in supratidal forests (e.g. *Melaleuca*) and mangroves (e.g. *Barringtonia*) (Fensham & Fairfax, 2003), or the occurrence of methane and CO_2 emissions associated with influence of ungulates on soils and water bodies (from nutrient enrichment and soil degradation with pugging). Additionally, increased precipitation projected for the region with climate change is likely to increase erosion of soils damaged by grazing and could therefore increase avoided abatement.

We multiplied our conservative estimate of abatement (0.3 CO_2-e ha yr^{-1}) by the area of remnant vegetation within each site.

2.5 Estimate co-benefits

2.5.1 Ecosystem service multifunctionality approach

Taxonomically diverse ecosystems have higher levels of ecological functions than less diverse systems (Cardinale et al., 2012), sustaining multiple ecosystem services at high levels, known as multifunctionality (Isbell et al., 2011). However, focussing on measures of biodiversity as a proxy for ecosystem function may be problematic, as there can be trade-offs among different services (Maskell et al., 2013). In contrast, measures of multifunctionality consider the dependence of relationships between biodiversity and ecosystem services on environmental conditions and can be tailored to the needs of stakeholders (Allan et al., 2015). We used the ecosystem service multifunctionality approach to estimate co-benefits likely to be provided by tidal restoration in non-financial values. This approach represents the supply of ecosystem services relative to human demand, rather than overall ecosystem function. However, all multifunctionality measures comprise a subset of all possible functions or services, thus it only represents a fraction of 'true' multifunctionality. This approach takes the sum of the standardised values of service indicators, which are weighted according to stakeholder objectives. It assumes the service indicators have a linear supply-benefit relationship, rather than a threshold over which the benefit is provided (Allan et al., 2015; Manning et al., 2018).

The ancillary ecosystem services (or co-benefits) of coastal wetlands were selected in accordance with their importance in the case study regions, in consultation with stakeholders. These are additional to carbon abatement, and include biodiversity, fisheries, water quality, and coastal protection. The level of each service was measured using indicators known to influence the supply of each service based on scientific literature, expert knowledge, and data availability for the regions. Indicator measures were first rescaled between 0 and 100 using the maximum value for each indicator in the set of restoration sites, and then given varied weightings under different scenarios before applying the multifunctionality equation (see Section 2.8).

2.5.2 Biodiversity

Coastal wetlands are significant primary and seasonal resources for a diversity of terrestrial vertebrates (Rog et al., 2017), marine megafauna (Sievers et al., 2019), and coastal fisheries (Abrantes et al., 2019). Mangroves provide breeding and refuge habitat for bird species during migration, winter seasons and drought (Kutt 2007). Coastal wetlands also support a number threatened and migratory species and threatened ecological communities (Houston, Black, et al., 2020).

Connectivity among diverse habitats in coastal ecosystem mosaics (interlinked estuarine, freshwater and terrestrial habitats) is a key process that facilitates the movement and migration of animals to complete their life histories and ontogeny, and dispersal of coastal plant propagules (Buelow & Sheaves, 2015). It also influences many ecosystem processes such as nutrient transport and cycling, food-web dynamics, predator–prey interactions, and transfer of genetic diversity, which are essential for ecosystem functioning, and the structuring of community assemblages (Sheaves, 2009). In mangroves in north Australia, the surrounding habitat matrix can significantly affect the composition of bird communities (Mohd-Azlan & Lawes, 2011). Patch size is also important in maintaining species diversity within ecological communities (Bryan-Brown et al., 2020). In particular, the ecological character of Ramsar wetlands is in better condition than other wetlands (Davidson et al., 2020).

We estimated the value of restored coastal wetlands for biodiversity as (1) connectivity to existing wetlands, (2) connectivity to Ramsar wetlands, (3) threatened species diversity, and (4) habitat for a threatened species or community in the study region. Patch size was correlated with several of the other indicators and was therefore excluded (Figure B4 and Figure B5). Measures of indicators varied by study region according to data availability and are described below. However, the biodiversity co-benefit in the three study regions was an evenly weighted set of four indicators. Methods and spatial datasets used in the co-benefits analysis are summarised in Table A5.

Fitzroy Basin

In this region the focal threatened species is the critically endangered Capricorn Yellow Chat (*Epthianura crocea macgregori*). Isolated populations of the chat occur on grassy marine plains with fresh to hypersaline wetlands bordering salt flats and mangroves at Broad Sound to the north, Fitzroy River delta in the south, and Curtis Island in the south-east (Houston et al., 2013). Habitat for the chat includes treeless vegetated marine plains between 1.5 and 2 m above MSL, associated with grass-sedge swamps and surrounding grasslands in particular dominated by the salt-tolerant sedge *Schoenoplectus subulatus* which provides tall cover and nesting habitat (Houston, Black, et al., 2020). Almost all chat sites have tidal-exclusion banks which pool the freshwater, extending the hydroperiod and the area of wetland habitat available (Houston, Black, et al., 2020). These banks are also important in protecting chat habitat from sea level rise (Houston, Elder, et al., 2020). Restoration sites intersecting known chat sites were excluded from the set of restoration sites to avoid any adverse impacts on this species from removing the banks. Central coordinates of known chat sites were buffered by 1 km and all restoration sites intersecting this buffered area were excluded. The restoration of surrounding wetlands can potentially facilitate landward migration of chat habitat with sea level rise to enhance conservation of this species in the future. Likely habitat was mapped from relevant pre-clear regional ecosystems within a 1 km radius of all known chat sites.

Verified records of threatened and migratory flora and fauna in the study region were sourced from the Wildnet database (Department of Environment and Science, 2022b), and the number of records within a 1 km buffer of the site counted. Connectivity with wetlands was calculated as the Euclidean distance of the site boundary to the nearest Ramsar site (Department of Agriculture Water and the Environment, 2020b) and to an existing terrestrial, estuarine or marine wetland (Department of Environment and Science, 2020c).

Peel-Harvey

Habitat for a threatened species or community was focussed on the threatened ecological community of temperate coastal saltmarsh. Vegetation types listed in the EPBC Act advice (Department of Sustainability Environment Water Population and Communities, 2013) were used to identify all areas of applicable vegetation types within the study region (Department of Primary Industries and Regional Development, 2020b) (Table A10), then clipped to include areas in remnant status. This data was complimented with the National intertidal and sub-benthic habitat mapping (University of Tasmania, 2018), as it contained other areas of the saltmarsh community not present in the WA vegetation data.

Records for threatened and specially protected (e.g. migratory) flora and fauna in the study region were sourced from the Threatened and Priority Flora and Fauna databases (Department of Biodiversity Conservation and Attractions, 2020, 2021), and the number of records within a 1 km buffer of the site counted. Connectivity with wetlands was calculated as the Euclidean distance of the site boundary to the nearest Ramsar site (Department of

Agriculture Water and the Environment, 2020b) and to a wetland identified in the National register of important wetlands (Environment Australia, 2010). Whilst these metrics co-vary, there are several large areas of wetland without Ramsar status in the region so both metrics were retained.

Ord River

Habitat for a threatened species or community was focussed on the endangered Purple-crowned Fairy Wren (*Malurus coronatus*). The species extirpated from the Ord River in 2003 (Skroblin, 2010), with hydrological change and the impact of cattle and invasive plants on the remaining core riparian habitat linked to decline (Department of Agriculture Water and the Environment, 2022). Suitable habitat for the species was mapped as riparian habitat of any major river in the study region, buffered by 1 km as previous records for the species ranged between 0-2 km from major watercourses. This layer was intersected with vegetation types mentioned in EPBC Act advice (Threatened Species Scientific Committee, 2015) or with previous records of the species in the region (Table A10).

Threatened taxa and connectivity to wetlands were calculated using the same method for Peel-Harvey.

2.5.3 Coastal fisheries

Estuaries, freshwater, tidal wetlands and shallow coastal waters provide important habitat to many recreationally and commercially important fish and invertebrates, including diadromous fish, that use these areas as nursery, feeding and/or reproduction grounds (Abrantes, Barnett, et al., 2015; Baker et al., 2019). The value of coastal wetlands (mangroves, saltmarsh, floodplain wetlands) to fishery species (both marine and diadromous fish) in north Queensland is influenced by its nursery habitat to support juvenile fisheries such as shallow water, muddy intertidal banks and mangroves (Abrantes et al., 2019; Sheaves et al., 2012). Additionally, it is influenced by the connectivity with seagrass meadows (Gilby et al., 2018), connectivity between marine, estuarine and freshwater habitats (Baker et al., 2019; Gilby et al., 2018; Nagelkerken et al., 2015), and through the contribution to basal food sources (Abrantes, Johnston, et al., 2015; Abrantes et al., 2019; Sheaves et al., 2012). In the Fitzroy Basin, floodplain wetland pools and their fish diversity was also positively influenced by connectivity with freshwater and marine environments (Sheaves et al., 2006).

There is no clear distinction between the trophic importance of different coastal ecosystems to fisheries nutrition (all are presumed to be equally important in some complex way to fish production). Thus, the importance of restored coastal wetlands for fisheries was represented by (1) the provision of nursery habitat, (2) connectivity with existing Fish Habitat Areas (if present), and (3) connectivity with permanent watercourses providing likely fish habitat. Indicator measures varied by study region according to data availability (Table A10).

Fitzroy Basin

The fisheries co-benefit was an evenly weighted set of all three indicators. The lower intertidal zone likely has two high tides daily in the Fitzroy Basin providing nursery habitat, there are several Fish Habitat Areas, and third order streams and above are likely to contain fish habitat and higher fish populations (Department of Agriculture Fisheries and Forestry, 2013). Thus, we calculated (1) the area of low intertidal zone within the site (MSL-MHWN, estimated from the HAT layer), (2) the closest Euclidean distance of the site boundary to a declared Fish Habitat Area (Department of Environment and Science, 2020c), and (3) the closest Euclidean distance of the site boundary to the nearest third order stream and above (Department of Resources, 2021f).

Peel-Harvey

The fisheries co-benefit for the Peel-Harvey was an evenly weighted set of two indicators, as we were unable to develop an intertidal zone layer for WA and there are no designated Fish Habitat Areas in the region. Habitat for fisheries in this region was assumed to be all areas of major perennial waterway and floodplains. The area of riverine habitat was estimated by averaging the width of major rivers in the area, then applying a 100 m buffer to the hydroline dataset (Crossman & Li, unknown). Floodplain areas were extracted from the flood projection model (Department of Water and Regulation, 2022). Connectivity to fisheries habitat is unique in this region given the closed nature of many of the estuaries, except Peel Inlet and Leschenault, which have openings to the sea. The logic applied here was that marine reserves provide refuges for fisheries, and fishes can access potential restoration sites via connections to the ocean then flow paths to the site (whether they be via an estuary or a riverine area). The total flow path distance between the site and a connection to the ocean was calculated using ArcGIS network analyses, then added to the Euclidean distance to the nearest marine reserve (Department of Agriculture Water and the Environment, 2020a).

Ord River

The fisheries co-benefit for the Ord River was also an evenly weighted set of two indicators. Habitat for fisheries in this region was assumed to be all areas of major perennial waterway (buffered by 100 m) and areas of intertidal and sub-tidal habitat (University of Tasmania, 2018). Connectivity to fisheries habitat were considered the flow paths to the potential restoration site from marine reserves (whether they be via an estuary or a riverine area). The total flow path distance between the site and a connection to the ocean was calculated using ArcGIS network analyses, then added to the Euclidean distance to the nearest marine reserve (Department of Agriculture Water and the Environment, 2020a).

2.5.4 Water quality

Assisting the recovery of coastal wetlands can enhance dissolved inorganic nitrogen (DIN) removal from tidal waters through denitrification (dinitrogen (N_2) losses to the atmosphere), N plant uptake, and N retention in sediments and biomass (Duarte & Krause-Jensen, 2018; Jickells et al., 2016; Reis et al., 2017).

The removal of DIN and Total Nitrogen (TN) in treatment wetlands (constructed wetlands and vegetated drains) in QLD is positively associated with high inflow concentrations (> 0.2 mg L^{-1} of DIN, > 0.7 mg L^{-1} of TN), low total suspended solids (TSS), high vegetation cover and high hydraulic efficiency (length:width ratio) to improve water retention time (Kavehei, Hasan, et al., 2021). Other studies have also found higher denitrification rates with higher estuarine water residence times (Jickells et al., 2016; Jickells et al., 2014; Mitsch, 2016), anthropogenic N enrichment (Reis et al., 2019) and accumulation of organic matter

(Fernandes et al., 2016; Reis et al., 2017). Treatment wetlands with very high TSS concentrations had low DIN removal, likely due to sediments smothering and limiting the establishment of macrophytes and nitrifier-denitrifier microbial communities (Kavehei, Hasan, et al., 2021). Treatment wetlands also have high particulate nitrogen removal capabilities which is important for meeting water quality objectives for N removal (Wallace et al., In Review).

The potential for restored coastal wetlands to remove DIN was based on (1) DIN concentration, (2) TSS concentration, (3) hydraulic efficiency, and (4) estuarine water residence time. Indicator measures varied by study region according to data availability (Table A10).

Fitzroy Basin

The water quality co-benefit was calculated from evenly weighted set of four indicators. DIN and TSS loads were obtained from catchment pollutant reports (Australian and Queensland Government, 2019; Waters et al., 2014) and converted to concentrations based on mean annual flow of catchments (tonnes L^{-1} yr^{-1}). Hydraulic efficiency was estimated as the area of the site intersecting a permanent watercourse (3 order streams and above) (Department of Resources, 2021f), calculated by applying a 100 m buffer to the length of the stream. Water residence time was estimated as the area of low intertidal zone within the site.

Peel Harvey

The water quality co-benefit for Peel-Harvey was calculated as an evenly weighted set of two indicators – the total nitrogen (TN) concentration per catchment in 2018 (kg/km^2) from the catchment nutrient reports (Department of Water and Regulation, 2018) as an indicator of DIN availability (this was not available for all catchments) and predicted water residence time. The TN concentration was transcribed onto a sub-set of shapefiles from the hydrolines dataset. Each site was allocated the TN concentration of any river with data that it connected too – for those on estuaries with multiple riverine sources this was the sum of all rivers. Residence time considered all areas that have fisheries habitat as potential holders of water, and then ranked sites based on this distance to a point of ocean exchange – closed systems such as Lake Clifton were considered to have indefinite residence time, whilst sites near an oceanic exchange had shorter residence times.

Ord River

No water quality co-benefit was calculated for the Ord River as no waterway health monitoring data was available at the time of analysis.

2.5.5 Coastal protection

Mangroves and saltmarsh can provide protection from coastal flooding during storm events and in regular conditions. Globally mangroves provide flood protection benefits exceeding $US 65 billion per year to more than 15 million people (Menéndez et al., 2020). Coastal vegetation increases resistance to the energy and flow of flood waves, such as storm surges, reducing inland flood water levels (Temmerman et al., 2012). In temperate saltmarshes, coastal flood mitigation arises from localised wave attenuation, but more dominantly from estuary-scale surge attenuation (Fairchild et al., 2021), while in tropical and subtropical regions, mangroves provide higher wave attenuation from higher drag forces associated with branch surface area and leaf size compared to saltmarshes (van Hespen et al., 2021).

Coastal wetlands can also provide indirect protection via sediment accretion and soil stabilisation (Lovelock et al., 2015; Thampanya et al., 2006), reducing erosion from both inland and coastal flooding.

We estimated flood mitigation benefits as (1) indirect protection during average conditions – reduction of erosion from inland flooding, determined from river flood mapping, and (2) direct protection during storm events – wave and erosion attenuation from coastal flooding, determined as coastal vegetation with potential for wave and flow resistance. Indicator measures varied by study region according to geomorphological and hydrological conditions (Table A10).

Flood mitigation inherently refers to the protection of communities and assets (Calil et al., 2015). While we have not explicitly considered the distance of restoration sites to population centres and infrastructure, this is indirectly incorporated into flood mapping extents.

Fitzroy Basin

Flood mitigation co-benefit was calculated as (1) the area of the site within the 1% AEP (100 year flood) from the Fitzroy River flood mapping (Department of Natural Resources, 2015) and (2) area of the site likely to restore to mangroves (BVG 35a) identified from the pre-clear regional ecosystem mapping (Department of Environment and Science, 2019a).

Peel-Harvey

As a system dominated by semi-enclosed estuaries and saltmarsh or supra-tidal vegetation with no mangroves, attenuation of floods from inland waters comes from the ability of coastal wetland vegetation to capture sediment and reduce erosion. In similar temperate saltmarshes of South Australia, sediment accumulation in vegetated samphire flats was between 2.6 and 9.4 mm/year (Dittmann, 2016). In the Leschenault estuary, the edges of the estuary are dominated by closed herblands, grasslands and sedgelands (Pen et al., 2000), which have potential to capture more sediment than saltmarsh forbelands. In the Peel-Harvey, similar and equally complex communities occur (McComb et al., 1995). The capture of such sediments elevates a site and protects it against flooding from the flooding of estuaries, created by the lag between the filling of the estuary and the time that water takes to drain into the ocean.

Potential for restoration sites to provide mitigation of flood waters was calculated using the area of saltmarsh to be restored (identified from the pre-European vegetation mapping) within the site that falls within the 100-year flood projection model envelope (Department of Water and Regulation, 2022). The restorable saltmarsh areas were buffered by 2 km as the currently mapped saltmarshes are all under water within the flood projection model. No direct flood mitigation co-benefit was calculated for coastal flooding due to storm surge, as we assume protection from oceanic forces (waves and storm surges) is provided by the dune system rather than coastal wetland vegetation.

Ord River

No flood projection model was available for the Ord River catchments; therefore we were unable to estimate the inland flooding benefit. However, we have estimated the potential for restoration sites to protect against coastal flooding by calculating the area of mangrove to be restored within each site.

2.5.6 Indigenous heritage

Indigenous heritage values were not included in the co-benefits, as there was insufficient data to identify places and areas of local cultural significance across potential restoration sites. Gathering local data would require engagement with Traditional Custodians on specific sites of interest.

Indigenous heritage has been incorporated into our approach through identification of the potential to support Traditional Custodian-led blue carbon projects or partnerships with landholders, recognising that Aboriginal and Torres Strait Islander people are the First Nations people of Country that is proposed for coastal wetland restoration. We engaged with Indigenous groups and Traditional Custodian representatives to explore the interest for First Nations people to undertake blue carbon projects and the potential challenges to identify strategies by which this could be achieved.

We explored the opportunity for Traditional Custodian-led blue carbon projects across potential restoration sites in each study region. 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) contains any Aboriginal sites or places of cultural importance listed on state databases (Table A10). Spatial data on the restoration opportunities will be provided to the NRM groups and Indigenous groups relevant to the study regions, to support development of blue carbon work packages.

Fitzroy Basin

In Fitzroy Basin, there are no registered Aboriginal places in the study region (Department of Seniors Disability Services and Aboriginal and Torres Strait Islander Partnerships, 2021), therefore the potential for Traditional Custodian-led or co-management of blue carbon projects was identified as restoration sites with (1) 10 ha or more of native title (National Native Title Tribunal, 2020), or (2) 10 ha or more of potential native title identified from cadastre data (Department of Resources, 2021a), and (3) has a registered Aboriginal party interest (Department of Resources, 2022).

Peel-Harvey

The area encompassing Aboriginal heritage places (Department of Planning Lands and Heritage, 2022) was calculated for each restoration site within the Peel-Harvey. The South West Native Title settlement is in progress but data regarding the potential for land to return to Indigenous ownership and management are yet to be released (Department of the Premier and Cabinet, 2022). Therefore, restoration sites were overlaid upon a raster of the assessment areas for all relevant parties in the settlement and sites ascribed a Traditional Custodian body who would be the most appropriate collaborator for that site.

Ord River

The Kimberley Land Council have resolved many Native Title settlements for the Ord River region and have identified parties for the area covering the potential restoration sites. However, no formal resolutions have been made for the areas within our study sites, thus identification of native title for sites was not possible at this time.

2.6 Estimate restoration feasibility

Our restoration feasibility indicator was based on the presence of an existing drain or barrier and the frequency of tidal flows to account for the likelihood of implementation and effectiveness. Reinstatement of tidal flows are likely to be more feasible on sites that have manmade drains or barrage/weirs and can be hydrologically restored by removal of that structure. Frequent tidal flushing an enhance natural recruitment and biotic interactions such as predation and competition (Lewis et al., 2019; Zhang et al., 2018) and can decrease invasion by aquatic weeds (Abbott et al., 2020).

2.6.1 Fitzroy Basin

Restoration sites in Fitzroy Basin were intersected with the canal lines mapping (Department of Resources, 2021b) and barriers to fish passage data (Marsden, 2015) to identify existing drains/barriers. No canals intersected the sites, therefore, we assigned an implementation probability of 1 for sites with 10 or more barriers, 0.75 to sites with more than 1, 0.5 to sites with 1, and 0.25 with none. To estimate frequency of tidal flows, we used the intertidal zone layer developed for the low, mid and high intertidal zones. In the central coast of Queensland, regions are likely to experience two high tides daily within the low intertidal zone. An effectiveness probability of 1 was therefore assigned for sites that had 40% or more of its area within the low intertidal zone, 0.75 for sites that had 20% or more, 0.5 for sites than had 10% or more, and 0.25 for sites than had less than 10%. Our indicator of restoration feasibility was calculated as the average of the barrier occurrence and tidal inundation probabilities at each site, ranging low (0.25) to high (1).

2.6.2 Peel-Harvey

In Peel-Harvey, the tidal range is very low and tidal flow in semi-closed estuaries is influenced via connectivity to the ocean. Therefore, restoration feasibility was calculated by proximity to an oceanic input and the length of modified waterways. Proximity to oceanic input used the same metric as the riverine component of fisheries connectivity. As sites with hydrological impoundments are likely to be more feasible for returning of natural hydrology, a dataset was constructed from existing data on large drains (Crossman & Li) and hand-digitised barriers (e.g. roads) and small drains extracted from aerial imagery.

We assigned an implementation probability based on length of drains and barriers per site: 0 m = 0.25, 1-50 m = 0.25, 51-100 m = 0.5, 101-150 m = 0.75, and >150 m = 1. An effectiveness probability per site were assigned based on connectivity to the ocean. Missing data for connectivity to ocean were assigned 1000 km and 0.25 probability. Restoration feasibility was calculated as the average of the drain length and connectivity to ocean probabilities, ranging from low (0.25) to high (1).

2.6.3 Ord River

Restoration feasibility for tidal restoration was not calculated for the Ord River 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. Vegetation across the potential restoration sites is mapped as remnant, despite evidence of degradation causing extirpation of threatened species (see Section 2.5.2.2). As there was no available data to estimate degraded wetland area and no current ERF method for estimating carbon abatement from preventing or reducing disturbance to coastal wetlands, we did not undertake the economic analyses for the Ord River.

2.7 Cost benefit analysis

The net present value (NPV) generated from conversion of agricultural land use to coastal wetlands was calculated for each site using a discounted cash flow analysis (Equation 1) (Hagger et al., 2022; Roebeling et al., 2007). NPV considers financial benefit from carbon abatement, annual opportunity cost from agricultural production, and restoration and maintenance costs. The NPV was evaluated over 25 years to reflect project permanence, using a discount rate of 1% per annum. A 1% discount rate is considered realistic for climate change mitigation projects, like coastal wetland restoration, that improve with age and their accumulation of carbon stocks and provision of co-benefits, not depreciate (Costanza et al., 2021; Drupp et al., 2018). However, the recommended rate for public investments in infrastructure projects in Australia is 4% (House of Representatives Standing Committee on Infrastructure Transport and Cities, 2018). Therefore, we conducted a sensitivity analysis to assess how a higher 4% discount rate changes the NPV. A weighted average carbon price of AU$ 16.22 per Mg CO_2-e calculated from the Australian Government last three auction results (October 2021, April 2021 and September 2020) (Clean Energy Regulator, 2021b) was applied to annual carbon abatement (in CO_2-e) to calculate the financial benefit of restoration. Higher prices can be obtained on the voluntary market for projects with high social or environmental value, with ACCU spot prices reaching AU$ 40 per Mg CO_2-e (Clean Energy Regulator, 2021a) and may be even higher on the private voluntary market (Kuwae et al., 2022). We also conducted sensitivity analyses to assess how a higher carbon price and a longer 100-year permanence period changes the NPV (Firn et al., 2015).

The NPV of each restoration site *i* was defined as the discounted sum of the differences between the restoration gross margin (financial benefit, *B* minus maintenance cost for first five years, *C*) and the farm gross margin (*FGM*). The later occurs in each year *t* over the defined time horizon T, together with a one-off restoration cost, which occurs at the outset (C_0), and *r* is the discount rate:

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

FGMs (AUD ha-1 yr-1) for grazing vary according to enterprise, climate and pasture growth. For the Fitzroy Basin, the gross margins per adult equivalent (AE) have been estimated for five productivity groupings based on the enterprise, herd structure, animal production, and animal input costs determined by the beef CRC templates and updated to 2015 prices (Star et al., 2017) (Table A7). We escalated gross margins from June 2015 to December 2021 prices using the relevant Consumer Price Index (all groups, Brisbane; 14% increase) (Australian Bureau of Statistics, 2021). In the coastal region of the Fitzroy Basin, specifically the Kunwarara area, beef enterprises and stocking densities for sustainable beef production have been recommended based on land types and their productivity. These recommendations were done acknowledging that freshwater plains, including ponded pastures, are the most productive system to be used for fattening cattle during the dry season (up to 8 months from May-December) (Black Speargrass, Unknown). Recommended stocking densities range from 0.1-0.49, and are within the average reported for Queensland Southern Coast (0.26 AE/ha) in Northern Beef Report (McLean et al., 2014). Land types and associated stocking densities have been assigned to productivity groupings to match the available data on returns from beef production (Table A8).

Firstly, productivity groupings were assigned to Kunwarara land types based on their soil and vegetation descriptions and given their location with the high rainfall coastal region (Table A7). Land types for each restoration site were identified from the GLM land type mapping (Department of Agriculture and Fisheries, 2021) and supplemented with the Wetlands data (Department of Environment and Science, 2020c) to identify freshwater plains, as these areas are mapped as marine plains in the GLM land type mapping. The GLM land types were assigned to Kunwarara land types to identify stocking densities and productivity groupings. Annual FGM for each restoration site in the Fitzroy Basin region was calculated as the GLM land type area multiplied by the stocking density, multiplied by the gross margin per AE specific to the productivity grouping for that land type.

For the Peel Harvey and South West region, we used available FGMs from South Australia (South Australian Grains Industry Trust, 2022) because WA government data was not accessible. We applied FGMs for beef cattle from high rainfall regions to reflect the climate of the study region. The FGMs are the difference between the annual gross income and the variable costs directly associated with the enterprise (bull purchases, animal health, supplementary feed, sale costs, transport), based on a stocking density of 10 dry sheep equivalent (DSE) per hectare. For beef cattle in high rainfall regions, FGMs are given for beef cattle (breeding young cattle for local trade, grass fattened, AU$ 593.12 per ha) and beef trading (finishing young cattle for local trade, grass fattened, AU$ 83.99 per ha). Annual FGM for each restoration site in the Peel region was calculated as the restoration area multiplied by the FGM for beef trading. We conducted a sensitivity analysis to assess how a higher farm gross margin achieved by beef cattle breeding changed the NPV.

The NPV analysis used median restoration costs reported for mangroves and saltmarsh for developed nations (Australia, USA and UK) using natural recovery hydrological restoration (without planting) (Bayraktarov et al., 2016). Costs in USD ha^{-1} at 2010 base estimate, were converted to AUD using the 2010 exchange rate (1.09) (Feenstra et al., 2015), and then escalated from December 2010 to December 2021 using the relevant CPI (all groups, Brisbane; 26% increase) (Australian Bureau of Statistics, 2021). The majority of costs reported were for capital costs; however, some projects also included maintenance costs. Assuming hydrological restoration involves mainly earthworks for modification of drains/bunds, the lower (saltmarsh) cost (AU$ 8,591 ha^{-1}) was used with a sensitivity analysis conducted to assess how an upper (mangrove) cost (AU$ 71,363 ha^{-1}) affects the NPV. Given natural recovery requires minimal maintenance, AU$ 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 A9).

NPV was calculated for the base scenario and sensitivity analyses (Table 2).

Table 2 Net Present Value (NPV) scenarios, including base and variations of restoration cost discount rates, carbon price, project permanence, and enterprise and associated farm gross margin (FGM) for sensitivity analyses. *For Peel only

Scenario	**NPV equation**	**Restoration cost (AUD ha^{-1})**	**Discount rate (%)**	**Carbon price (AUD Mg CO_2-e)**	**Permanence (years)**	**FGM (AUD ha^{-1})**
1	Base	$7,174	1	$16.22	25	$83.99
2	Higher restoration cost	$59,586	1	$16.22	25	$83.99
3	Higher carbon price	$7,174	1	$40	25	$83.99
4	Higher discount rate	$7,174	4	$16.22	25	$83.99
5	Longer permanence	$7,174	1	$16.22	100	$83.99
6*	Higher farm gross margin	$7,174	1	$16.22	25	$593.12

2.8 Cost-effectiveness analysis

We used a cost-effectiveness (CE) analysis to prioritise sites, considering NPV, restoration feasibility, and the provision of ancillary ecosystem services (biodiversity, fisheries, water quality and coastal protection) using the multifunctionality approach (Equations 2 and 3) (Hagger et al., 2022; Manning et al., 2018). CE analyses were conducted for the NPV scenario at the current carbon price (S1) and higher carbon price (S4). Measures of ecosystem service indicators were scaled between 0-1, multiplied by the weighting of the ecosystem service indicators, and summed. The weights sum to one and provide an indication of the relative importance of each ecosystem service. Ecosystem services were initially each given equal weights (0.25) which were divided among the indicators for each service. For example, biodiversity had four indicators, so each biodiversity indicator was given a weighting of 0.0625. However, ecosystem services are valued differently by stakeholders in every region. To analyse the sensitivity of rankings to varying importance of ecosystem services, the CE analysis was repeated to give a higher weighting to each ecosystem service in turn (0.7 and 0.1 to others). Applying the reduced weighting to each indicator within each service avoids overrepresentation of similar functions and substitutability issues (Manning et al. 2018).

The ecosystem service multifunctionality of each site *i* (ES_i), given the weighted sum of scaled indicator measures, was estimated as:

$$ES_i = \sum_{n=1}^{N} [I_n \times W_n] \qquad \text{Equation 2}$$

where each site has a total of *N* indicators; I_n is the percentage by which indicator *n* has been met (relative to the maximum value that can be attained by that indicator in the set of restoration sites); and W_n is the proportion weight of I_n to the site multifunctionality (ES_i).

As NPV can be either negative (a cost) or positive (a profit), two CE equations were applied to allow ranking of the sites given effectiveness. If NPV was negative, the CE of site *i* (CE_i) was calculated by the expected NPV (potential NPV_i divided by the feasibility (F_i), divided by the ES_i, so that a lower score equated to a higher cost per percent co-benefits:

$$CEcost_i = (NPV_i \div F_i) \div ES_i \qquad \text{Equation 3}$$

If NPV was positive, CE_i was calculated so that a higher score equated to a greater profit and percent co-benefits:

$$CEprofit_i = (NPV_i \times F_i) \div (100 - ES_i) \qquad \text{Equation 4}$$

Under each NPV and CE scenario, sites were ranked from highest to lowest (with 1 being the highest ranking).

2.9 Statistical analysis

To explore the profitability of restoration, the number of sites with positive values were calculated across the NPV scenarios. Relationships between NPV and CE rankings were assessed using hierarchical cluster analysis and non-metric multi-dimensional scaling (nMDS) based on resemblance matrices to analyse the sensitivity to different assumptions and assess trade-offs in scenarios (Harris et al., 2014). Each table of NPV or CE 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 tables using the vegdist function in the vegan package (Oksanen et al., 2019). A complete hierarchical cluster analysis was performed using the hclust function in the R stats package (R Core Team, 2020) and dendogram plotted, to compare the resemblance matrix among scenarios. To visually compare scenarios, a nMDS ordination of the scenarios was constructed using the metaMDS function in the vegan package, also based on the resemblance matrix. To explore the benefit of the NPV and CE prioritisation approaches and trade-offs in achieving multiple ecosystem services, we compared the sum of the restoration area, financial benefit (NPV), carbon abatement, and equal weighted co-benefits that would in the top 10 sites ranked in the NPV and CE prioritisations for S1 and S3 versus the top 10 sites ranked purely by carbon abatement or co-benefits.

To assess trade-offs and synergies between carbon abatement and co-benefits, we looked at the relationship of each scaled co-benefit indicator and the mean annual carbon abatement per site fitting a linear regression and slope to each relationship (Allan et al., 2015). Relationships between co-benefits were assessed by Pearson correlation coefficients and pairs plots. We also mapped co-benefits (sum of the scaled indicator measures) to assess variation in the spatial distribution of ecosystem services and to find hotspots of multiple ecosystem services that can be prioritised for restoration (Allan et al. 2015).

3. Results

3.1 Restoration opportunity

3.1.1 Fitzroy Basin

We identified 455 potential restoration sites on coastal grazing land in Fitzroy Basin, totalling 31,686 ha that could be suitable for hydrological restoration to coastal wetlands. With the prediction of a +0.7 m or +1 m sea-level rise, this increased to 60,142 ha or 67,097 ha, respectively. With removal of sites containing Capricorn Yellow Chat populations, the potential was reduced to 425 sites over 13,874 ha (Figure 2). The majority of potential restoration area were in the Fitzroy catchment (42%), followed by Styx (36%) and Shoalwater (14%; Figure 5a). Many of the sites were small, with 17 sites over 100 ha (Figure B.1 for the distribution of restoration site sizes). Grazing was the dominant land use in the HAT (97%), however the potential restoration area only comprised 38% of the total agricultural land within the HAT boundary, because of the removal of the Capricorn Yellow Chat sites. The potential restoration area was historically mostly saltmarsh (7,436 ha) and floodplain woodland (5,523 ha), followed by sedgeland (264 ha), Melaleuca wetland (248 ha), mangroves (146 ha), estuary (251 ha) and waterholes (1 ha).

3.1.2 Peel-Harvey

We identified 43 potential restoration sites in Peel Harvey, totalling 348 ha (Figure 3), which increased to 1,762 ha or 2,765 ha with the prediction of a 0.7 m or 1 m sea-level rise, respectively. The potential restoration area was in the Busselton Coast (58%), followed by Murray River (19%), Collie River (15%) and Harvey River (8%; Figure 7a). All of the sites were small, with the largest site being 47.9 ha. Beef grazing was the dominant land use and the majority of this was non-irrigated (97.3%). The potential restoration area was historically thicket (dominated by *Acacia spp*., *Casuarina spp*. and *Melaleuca spp*.; 210 ha), low woodland (58 ha), woodland (48 ha), low forest (19 ha), salt pan (10 ha), and a forest mosaic (4 ha). These vegetation types were assumed to transition to 338 ha of supratidal forest and 10 ha of saltmarsh.

3.1.3 Ord River

We identified 24,123 ha of land, aggregated into eight potential restoration sites within 5 km of each other within the Ord region that may be suitable for restoration (Figure 4). This increased to 30,394 ha or 51,566 ha with prediction of +0.7 m or +1 m sea-level rise, respectively. Unlike the other study regions, all the Ord River restoration area is mapped as remnant vegetation, and local impoundments or drains impeding tidal flows are not present. The potential for tidal restoration of these areas is therefore limited and the economic analysis under the blue carbon tidal restoration method was not conducted due to uncertainty with estimating carbon abatement and the condition of the wetlands. Instead, a preliminary method assuming the condition of grazed remnant vegetation would improve with the exclusion of ungulates is applied to estimate potential abatement.

In the pre-European vegetation mapping, the potential restoration area was primarily saltmarsh (11,096 ha; 46% of area) followed by mud and salt flats (6,995 ha; 29%), mangroves (4,101 ha; 17%) and short salt grasslands (1,689 ha; 7%). The mud and salt flats were generally located on the landward side of the mangroves and would likely transition to saltmarsh or mangroves. The short salt grasslands were described as high grass savanna woodland on sandstone: Bloodwood (*Corymbia dichromophloia*), stringybark (*Eucalyptus tetrodonta*) over curly spinifex (*Triodia bitextura*) and sorghum (*Sorghum* spp.) (Beard et al., 2013) and would likely transition to supratidal vegetation.

3.2 Carbon abatement

3.2.1 Fitzroy

Restoration of all 13,874 ha at Fitzroy Basin could abate 162,178 Mg CO_2-e per year (Table 3), of which 61% would be from carbon sequestration in biomass and soils of restored wetlands (net removals, with wetland emissions deducted) and 39% from conversion of the grazing land use (avoided emissions, with baseline wetland removals deducted). The highest removals arise from restored supratidal wetlands, because of the high extent of cleared floodplain woodland and *Melaleuca* forest assumed to transition to supratidal forest. On a per unit area basis the highest net removals come from restored mangroves, because of higher biomass and soil carbon accumulation. The highest avoided emissions are from CH_4 and N_2O emissions associated with flooded agricultural land, because of the large areas of ponded pastures (mapped as hydrologically modified wetlands) in the Fitzroy Basin. The mean annual carbon abatement per ha varied widely across catchments, being highest in Calliope, and lowest at Styx (Figure 5b).

3.2.2 Peel-Harvey

Restoration of all 348 ha at Peel Harvey would abate much less per year (4,312 Mg CO_2-e yr^{-1}, Table 3) compared to the Fitzroy Basin case study, of which 98% would be from wetland net removals, and only 2% from baseline net emissions. While in total there was a large carbon abatement difference between the Peel-Harvey and Fitzroy Basin regions, on average the carbon abatement per hectare in the temperate region of Peel-Harvey was slightly higher than the tropical region of Fitzroy Basin (12.38 Mg CO_2-e ha^{-1} yr^{-1} and 11.68 Mg CO_2-e ha^{-1} yr^{-1}, respectively). This result is because of the higher proportion of area in Peel-Harvey that would likely restore to supratidal forest, which has higher levels of carbon sequestration in biomass.

3.2.3 Ord River

Using our conservative approach for carbon abatement for rehabilitation of coastal wetlands of the Ord River by removing grazing pressure, rehabilitation of all 24,123 ha of land currently mapped as remnant mangroves, saltmarsh and supratidal vegetation would abate 7,237 Mg CO_2-e per year in the Ord River region. Levels of abatement could be increased with evidence that grazing reduced woody biomass accumulation.

Table 3 Annual avoided emissions and removals from ceasing agriculture, and coastal wetlands removals and emissions from restoration of all potential sites (above-ground biomass (AGB) has been averaged from 25 years). The net carbon abatement is the sum of the sub-totals for baseline and wetlands. A positive value for supratidal emissions indicate a net sink (uptake) relative to the atmosphere.

Land use / wetland type	Emission / removal	Fitzroy Basin Mg CO_2-e yr^{-1}	Peel-Harvey Mg CO_2-e yr^{-1}
Baseline			
Grazing (emissions)	CH_4 Flooded agricultural land, managed wet pasture	48,697	No data
	N_2O Flooded agricultural land, managed wet meadow or pasture	19,853	No data
	CH_4 Ponds and other constructed water bodies	1,951	1.5
	CO_2 Soil carbon loss	4,712	209
Wetlands (removals)	CO_2 Soil carbon accumulation in hydrologically disturbed mangrove, saltmarsh, and herbaceous settings	-11,286	NA
	CO_2 Soil carbon accumulation in disturbed supratidal forest	-347	-116
Sub-total baseline		*63,579*	*95*
Coastal wetlands			
Mangrove (removals)	Above ground biomass	2,984	NA
	Below-ground biomass	806	NA
	Soil carbon	1,386	NA
Saltmarsh (removals)	Above-ground biomass carbon	1,537	12
	Soil carbon saltmarsh	13,566	18
Supratidal forest (removals)	Above-ground biomass carbon	49,817	2703
	Below-ground biomass carbon	15,942	730
	Soil carbon	12,929	757
Mangrove (emissions)	CH_4 Flooding of wetlands	-24	NA
	N_2O Flooding of wetlands	-25	NA
Saltmarsh (emissions)	CH_4 Flooding of wetlands	-24	-0.03
	N_2O Flooding of wetlands	-265	-0.35
Supratidal forest (emissions)	CH_4 Flooding of wetlands	+354	+21
	N_2O Flooding of wetlands	-383	-22
Sub-total wetlands		*98,599*	*4217*
Net carbon abatement		**162,178**	**4,312**

3.3 Net present value

3.3.1 Fitzroy Basin

NPV over 25 years per restoration site using the higher carbon price varied between AU$ -10,897,526 (cost) to AU$ 23,568,214 (profit) (Figure 5c). We found that 48 (of 455) sites totalling 7,117 ha (51% of the potential area) were profitable in the Fitzroy Basin in the Calliope, Fitzroy, Shoalwater, and Waterpark catchments over 25 years using the higher carbon price (AU $40 per Mg CO_2-e), lower restoration cost and discount rate of 1% (S3). Over 100 years, profitability reduced to 26 sites totalling 943 ha (7%). However, for the other scenarios using the current carbon price, higher restoration cost, or discount rate of 4%, none of the sites returned a positive NPV (Figure 5d).

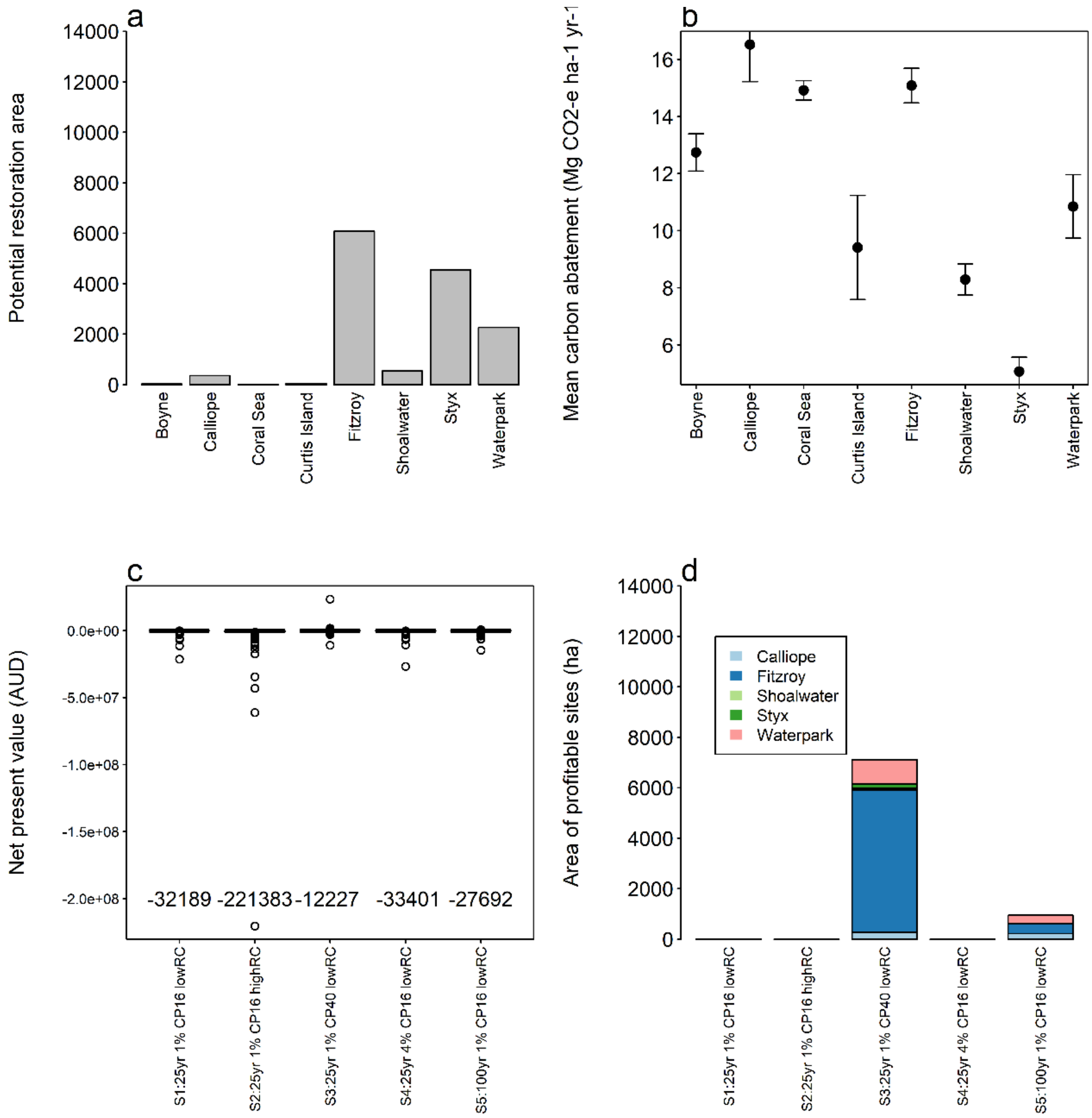

Figure 5 Restoration opportunity (a) and mean annual carbon abatement per ha with standard error bars across the Fitzroy Basin catchments (b), range of net present value (NPV) at restoration sites represented as box and whisker plots with minimum, quartiles, median, and maximum for different NPV scenarios (median values reported) (c), and area of profitable sites per catchment for different NPV scenarios (25 or 100 years, 1 or 4% discount rate, $16 or $40 carbon price, lower or upper restoration cost).

Comparison among NPV rankings showed that NPV calculated using the higher carbon price (S3) was most dissimilar to the other scenarios (Figure 6a). There was little dissimilarity in NPV rankings between the base scenario (S1) and the scenarios with the higher restoration cost (S2), 7% discount rate (S4), and longer project permanence (S5), indicating that NPV was most sensitive to variation in the carbon price.

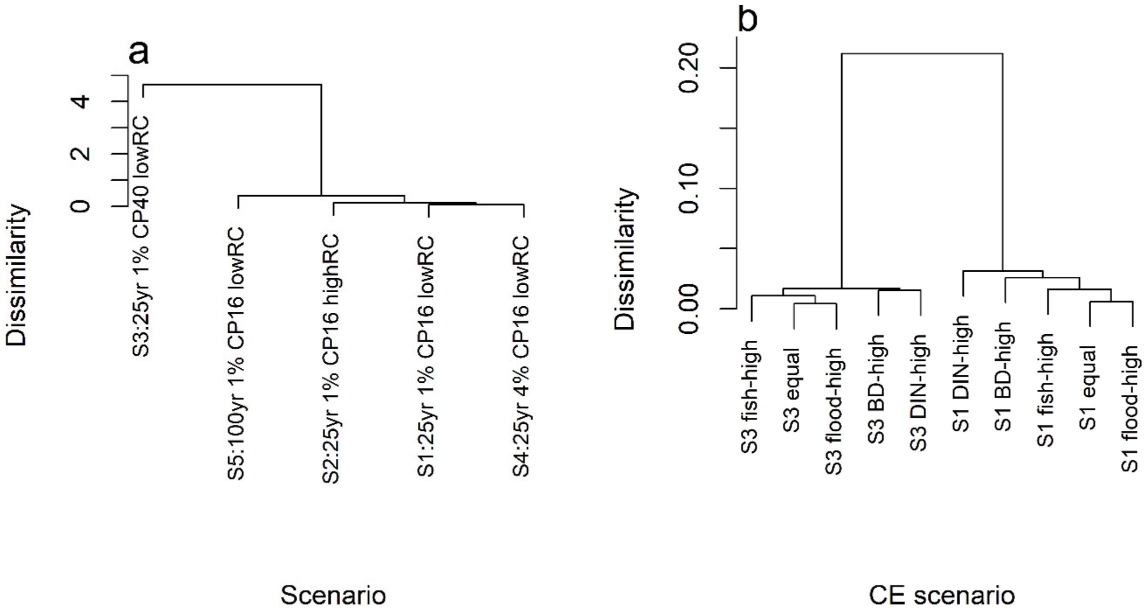

Figure 6 Dendograms from hierarchical cluster analysis of Fitzroy Basin case study showing relationships among (a) net present value (NPV) scenarios (25 or 100 years, 1 or 4% discount rate, $16 or $40 carbon price, lower or upper restoration cost; Table 1), and (b) cost-effectiveness (CE) scenarios based on NPV at 25 years, 1% discount rate, current carbon price and lower restoration cost (S1) or with higher carbon price (S3), with different weighting combinations (equal weighting of indicators, and higher weighting of biodiversity (BD), fisheries, Dissolved Inorganic Nitrogen (DIN) removal or flood mitigation indicators).

3.3.2 Peel-Harvey

In the Peel-Harvey region, none of the sites were found to be profitable under any scenario. Even using the higher carbon price, NPV over 25 years per restoration site had a cost of between AU$ -74,334 and -2,032 (Figure 7c). While there was some annual profit after year five, this was not enough to offset the initial restoration cost and the maintenance costs in the first five years. This is because the sites are small and therefore have low carbon abatement per site and poor economies of scale. Further, data deficiency in incomes from land uses, mean that standard farm gross margins reported from South Australia were applied to all sites irrespective of land type, which were AU$ 83.99 ha^{-1} yr^{-1} for cattle fattening (S1) and AU$ 593.12 ha^{-1} yr^{-1} for cattle breeding (S6). While Fitzroy Basin used data on farm gross margins per land type, which varied from AU$ 0 – 174.83 ha^{-1} yr^{-1}. Comparison among NPV rankings showed that NPV calculated using the higher carbon price (S3) was again most dissimilar to the other scenarios (Figure 8a), with little dissimilarity between NPV rankings in the other scenarios.

3.3.3 Ord River

We did not estimate the NPV and CE of restoration of the Ord River areas, because of uncertainty with carbon abatement from management of grazing in wetlands.

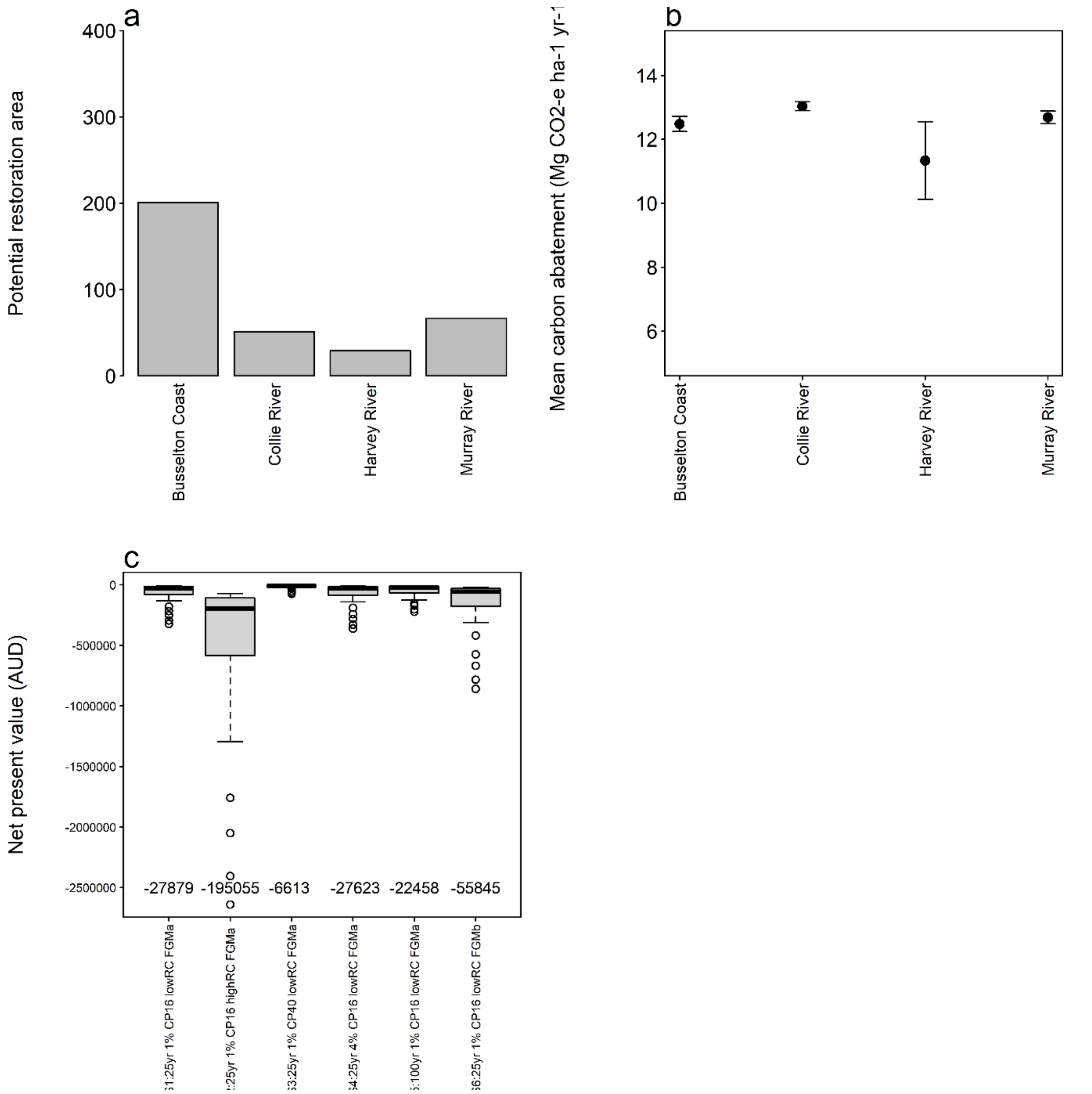

Figure 7 Restoration opportunity (a) and mean annual carbon abatement per ha with standard error bars across the Peel-Harvey and South West catchments (b), range of net present value (NPV) at restoration sites represented as box and whisker plots with minimum, quartiles, median, and maximum for different NPV scenarios (median values reported, 25 or 100 years, 1 or 4% discount rate, $16 or $40 carbon price, lower or upper restoration cost, farm gross margin (GM) a fattening or b breeding) (c).

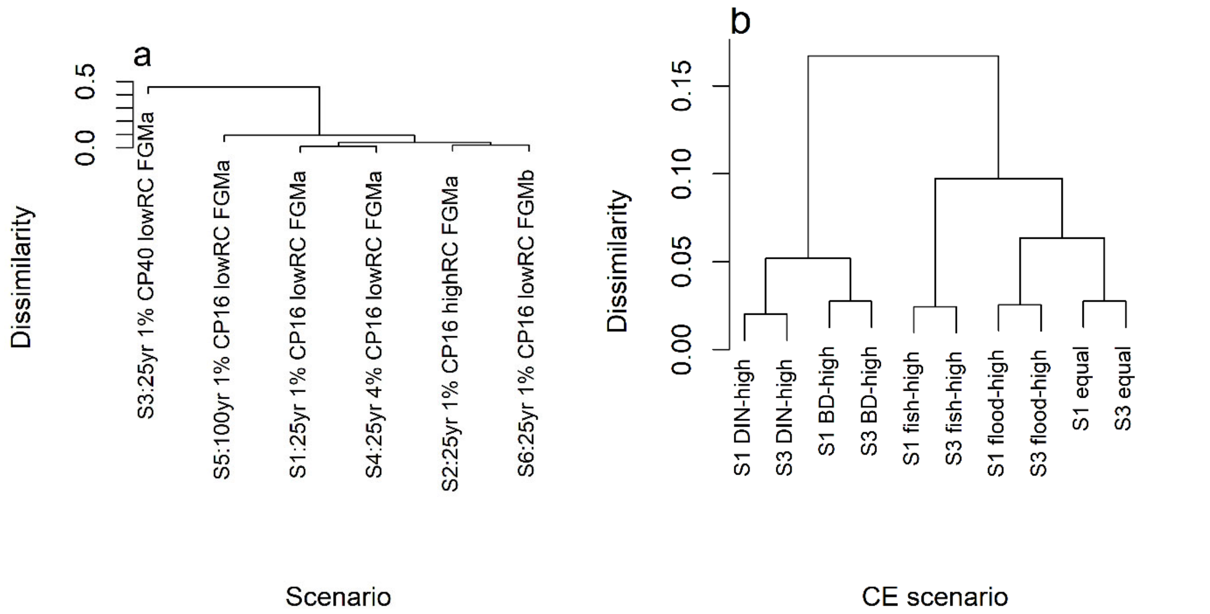

Figure 8 Dendograms from hierarchical cluster analysis of Peel-Harvey case study showing relationships among (a) net present value (NPV) scenarios (25 or 100 years, 1 or 4% discount rate, $16 or $40 carbon price, lower or upper restoration cost, farm gross margin (FGM) a fattening or b breeding; Table 1), and (b) cost-effectiveness (CE) scenarios based on NPV at 25 years, 1% discount rate, current carbon price and lower restoration cost (S1) or with higher carbon price (S3), with different weighting combinations (equal weighting of indicators, and higher weighting of biodiversity (BD), fisheries, or Dissolved Inorganic Nitrogen (DIN) removal indicators).

3.4 Cost-effectiveness prioritisation

3.4.1 Fitzroy Basin

In the Fitzroy Basin region, thirty sites had positive CE scores using the higher carbon price (S3), corresponding to those sites that returned a positive NPV. In these cases, sites with the highest profit and unit of co-benefit were given the highest rankings. Under the scenario with the current carbon price (S1), all CE scores were negative because of negative NPVs. In these cases, sites with the lowest cost per unit of co-benefit were given the highest rankings. The CE rankings therefore varied greatly depending on the NPV scenario. Varying the weightings of different ecosystem services also altered the CE rankings, so that a high weighting for biodiversity resulted in different outcomes in the rankings compared to other weighting combinations (Figure 6b).

3.4.2 Peel-Harvey

In the Peel-Harvey region, all CE scores were negative under the scenario with the current and higher carbon price (S1 and S3), corresponding to negative NPVs. The CE rankings therefore did not vary given the NPV scenario used. Varying the weightings of different ecosystem services altered the CE rankings, so that equal and high weighting for biodiversity provided different outcomes in the ranking to a high weighting for fisheries, DIN-removal and flood mitigation (Figure 8b).

3.5 Variation in prioritisation under different scenarios

Selecting restoration sites that maximise carbon abatement or co-benefits alone would restore the greatest area, however this would also cost the most because of loss of grazing land. Across both the Peel-Harvey and Fitzroy Basin, restoration of larger areas is required to provide higher carbon abatement. However, provision of co-benefits is not as dependent as carbon abatement on the area restored. There were larger differences between total carbon abatement and co-benefits in the top 10 sites in the Peel-Harvey region in comparison to the Fitzroy Basin region, indicating that trade-offs between carbon and co-benefits vary between regions (Tables 4 and 5).

In the Fitzroy Basin, under the current carbon price, selecting sites that provide the least cost per unit of co-benefits reduced the carbon abatement and co-benefits by 99% and 23% respectively. Under a higher carbon price, selecting sites that return the most profit and unit of outcome reduced the carbon abatement and co-benefits by 13% and 11%, respectively (Table 4). However, in Peel Harvey, NPV was negative across all scenarios, therefore even under a higher carbon price selecting sites that provide the least cost per unit of co-benefits also reduced the carbon abatement and co-benefits substantially (94% and 20%, respectively; Table 5).

Table 4 Fitzroy Basin case study. Total restoration area, net present value (NPV; given the respective scenario used), carbon abatement and co-benefits (equal weighted sum of scaled co-benefit indicators) from the sum of the top 10 sites ranked by carbon abatement, co-benefits, least cost/most profit (NPV) and cost-effectiveness (CE). The scenarios were based on NPV at 25 years, 4% discount rate, and lower restoration cost, with current (S1), or higher carbon price (S3).

Totals top 10 sites	Carbon abatement (S1)	Co-benefits equal (S1)	NPV (S1)	NPV (S3)	CE (S1 equal)	CE (S3 equal)
Area (ha)	8025	6007	12	5940	13	6052
NPV (AU$)	-40284053	-26674992	-85472	31473044	-103375	31418625
Carbon abatement (Mg CO_2-e 25 years)	3102935	2556064	6629	2700329	6015	2733284
Co-benefits equal (summed %)	380	412	320	366	305	370

Table 5 Peel Harvey case study. Total restoration area, net present value (NPV; given the respective scenario used), carbon abatement (equal weighted sum of scaled co-benefit indicators) from the sum of the top 10 sites ranked by carbon abatement, co-benefits, least cost/most profit (NPV) and cost-effectiveness (CE). The scenarios were based on NPV at 25 years, 4% discount rate, farm gross margin for fattening enterprise, and lower restoration cost, with current carbon price (S1), or higher carbon price (S3).

Totals top 10 sites	Carbon abatement	Co-benefits equal	NPV (S1)	NPV (S3)	CE (S1 equal)	CE (S3 equal)
Area (ha)	252	105	12	12	15	13
NPV (AU$)	-2001776	-850439	-121365	-70030	-154325	-79533
Carbon abatement (Mg CO_2-e 25 years)	79718	33636	3858	3858	4222	4362
Co-benefits equal (summed %)	214	443	255	255	369	345

3.6 Trade-offs between carbon and co-benefits

3.6.1 Fitzroy Basin

The distribution of co-benefits for biodiversity, fisheries, water quality, and flood mitigation varied across the restoration sites (Figure 9). While restoration sites within a 1km buffer of Capricorn Yellow Chat sightings were excluded from the potential restoration area, the remaining restoration sites surrounding the northern subpopulation, which is centred on Broad Sound, and the southern and south-eastern subpopulations encompassing the Fitzroy River delta and Curtis Island, may provide habitat that is used or occupied by the chat at some time. If restored, these sites could provide 3,021 ha of chat habitat (preclear regional ecosystems described as grassy marine plains and adjoining grasslands and samphire, Table A10). There were 170 records of threatened and migratory species within the restoration sites and their buffers, including 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*). The Fitzroy region contains one Ramsar listed wetland, Shoalwater and Corio Bay, and six Fish Habitat Areas in Cawarral Creek, Balban Dara Guya, Broad Sound, Corio Bay, Calliope River, and Fitzroy River.

We did not find any trade-offs between mean annual carbon abatement and co-benefit indicators at the site level (Figure 10). Indicators measured by area were generally synergistic with carbon abatement, including number of threatened species, potential chat habitat, low intertidal zone, major watercourse area, 100-year flood zone, and potential mangrove area, however these analyses were influenced by outliers (sites with large areas). Indicators measured by distance or catchment did not have any relationship with carbon abatement, including distance to an existing wetland, Ramsar wetland, major watercourse, and fish habitat area, and DIN and TSS catchment loads. We did find a possible trade-off between indicators within the DIN removal service. Catchments with high DIN concentrations, also have high TSS concentrations (*r* = 0.62, Figure B4). This can potentially limit denitrification capacity from benthic bacteria (Kavehei, Roberts, et al., 2021), but wouldn't limit N processing by plants (Wallace et al., In Review).

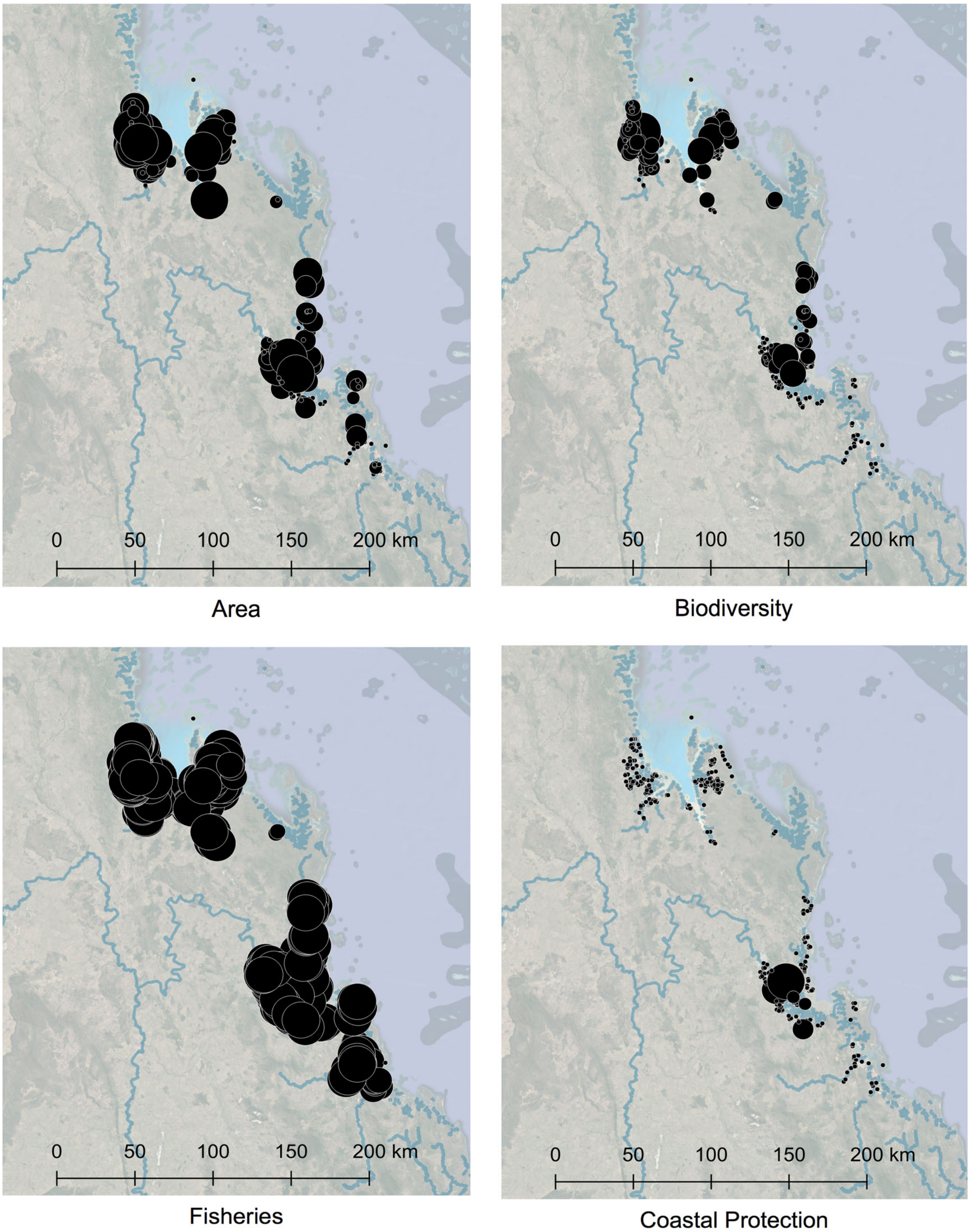

Figure 9 Spatial distribution of Fitzroy Basin restoration sites, with area, and the summed indicators for each co-benefit displayed as scaled icons on their centroid. All attributes are shown in their scaled form (a score of 0-100).

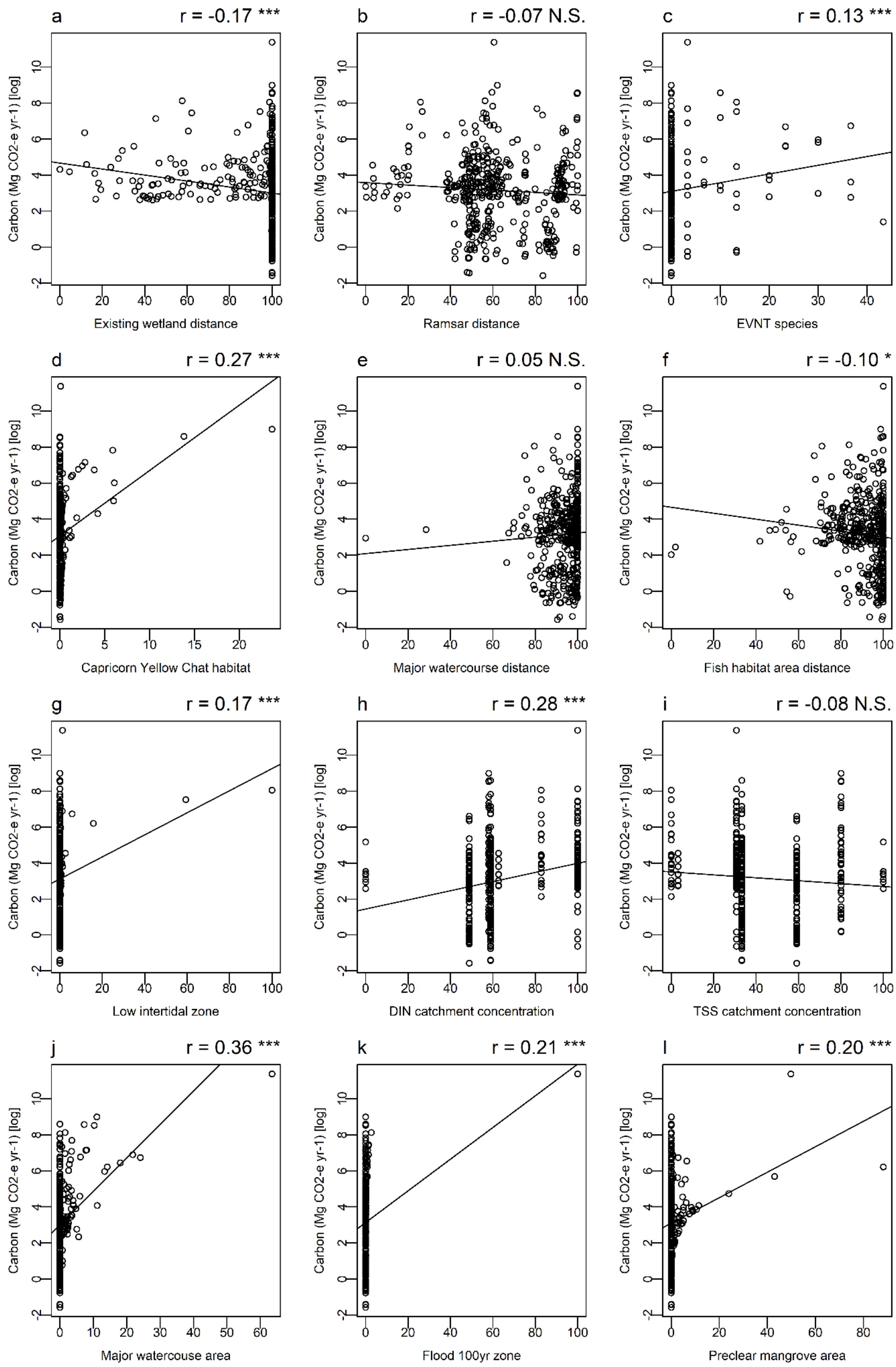

Figure 10 The relationships between mean annual carbon abatement and co-benefit indicators per Fitzroy Basin restoration site: Biodiversity - distance to existing wetland (a), distance to Ramsar wetland (b), number of threatened species (c), habitat for the Capricorn Yellow Chat (d); Fisheries - distance to major watercourse (e), distance to fish habitat area (f), area of low intertidal zone (g); Water quality - Dissolved Inorganic Nitrogen (DIN) catchment concentration (h), Total Suspended Solid (TSS) catchment concentration (i), area of major watercourse (j); Flood mitigation - area of 100 year flood zone (k), and area of preclear mangroves (l). Lines show fits from a linear regression. Pearson correlation coefficients (r) and significance levels ($p<0.05$*, $p<0.01$**, $p<0.001$***) are shown for each relationship. Indicator values have been scaled between 0-100.

3.6.2 Peel-Harvey

Patterns in co-benefits for biodiversity, fisheries, and water quality varied across sites (Figure 11). There were generally high biodiversity co-benefits across all sites, with the lowest scores restricted to sites on the Collie River and the northern set of sites south of Bunbury. This is a product of most sites having numerous records of threatened taxa within their buffers and occurring within 10 km of an important wetland or Ramsar wetland. Threatened species that are most likely to benefit from restoration were primarily migratory shorebirds including the curlew sandpiper (*Calidris ferruginea*), great knot (*Calidris tenuirostris*), eastern curlew (*Numenius madagascariensis*), greater sand plover (*Charadrius leshenaultii*). However other aquatic species such as Carter's freshwater mussel (*Westralunio carteri*) occurred within site buffers, as well as many species that are in proximity to sites and may benefit indirectly from wetland restoration (e.g. mammals such as the Western ringtail possum (*Pseudocheirus occidentalis*), reptiles such as green and loggerhead turtles (*Chelonia mydas* and *Caretta caretta*), and many species of black cockatoo). There was higher variance in the potential co-benefits for fisheries, with sites far from perennial waterways (south of Bunbury) or with no connection to the ocean (e.g. Lake Clifton). Likewise, high variance in potential benefits to water quality showed strong variance across sites, with sites on major tributaries (or combinations of same) having the highest co-benefit.

Similar to Fitzroy Basin, co-benefit indicators measured by area were generally synergistic with mean annual carbon abatement per site, and indicators measured by distance or catchment did not have any relationship with carbon abatement. However, we did find a possible trade-off between carbon abatement and DIN removal services, with lower carbon abatement in sites with higher water residence time in closed estuary systems (Figure 12). However, the capacity for the lakes to remove N may be limited because of low sediment-vegetation-water DIN contact. There was also a possible trade-off between fisheries and DIN removal services, with lower fish connectivity to marine reserves in sites with higher water residence time in closed estuary systems (r = -0.41, Figure B5).

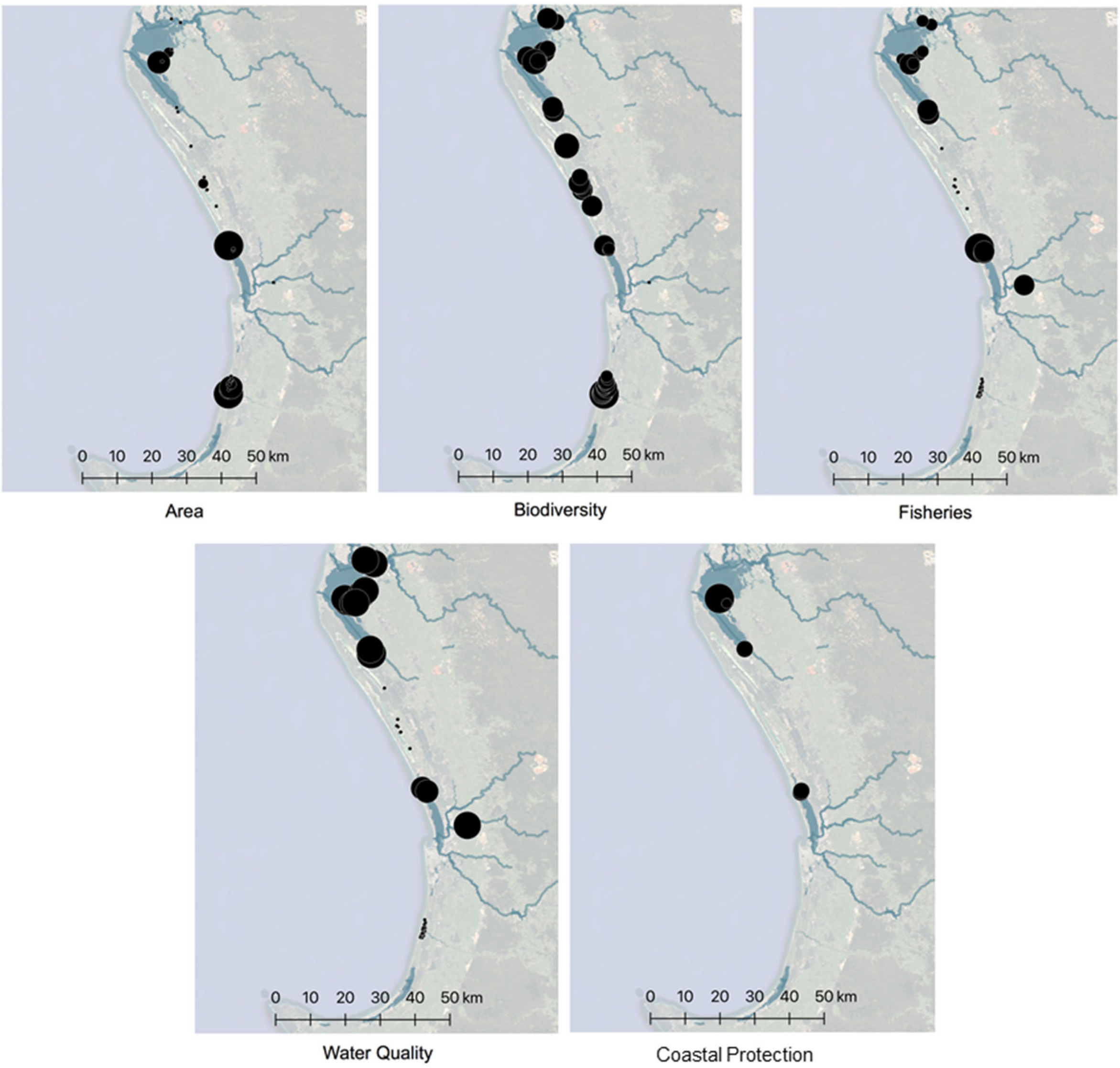

Figure 11 Spatial distribution Peel-Harvey restoration sites, with area and summed indicators for each co-benefit displayed as scaled icons on their centroid. All attributes are shown in their scaled form (a score of 0-100).

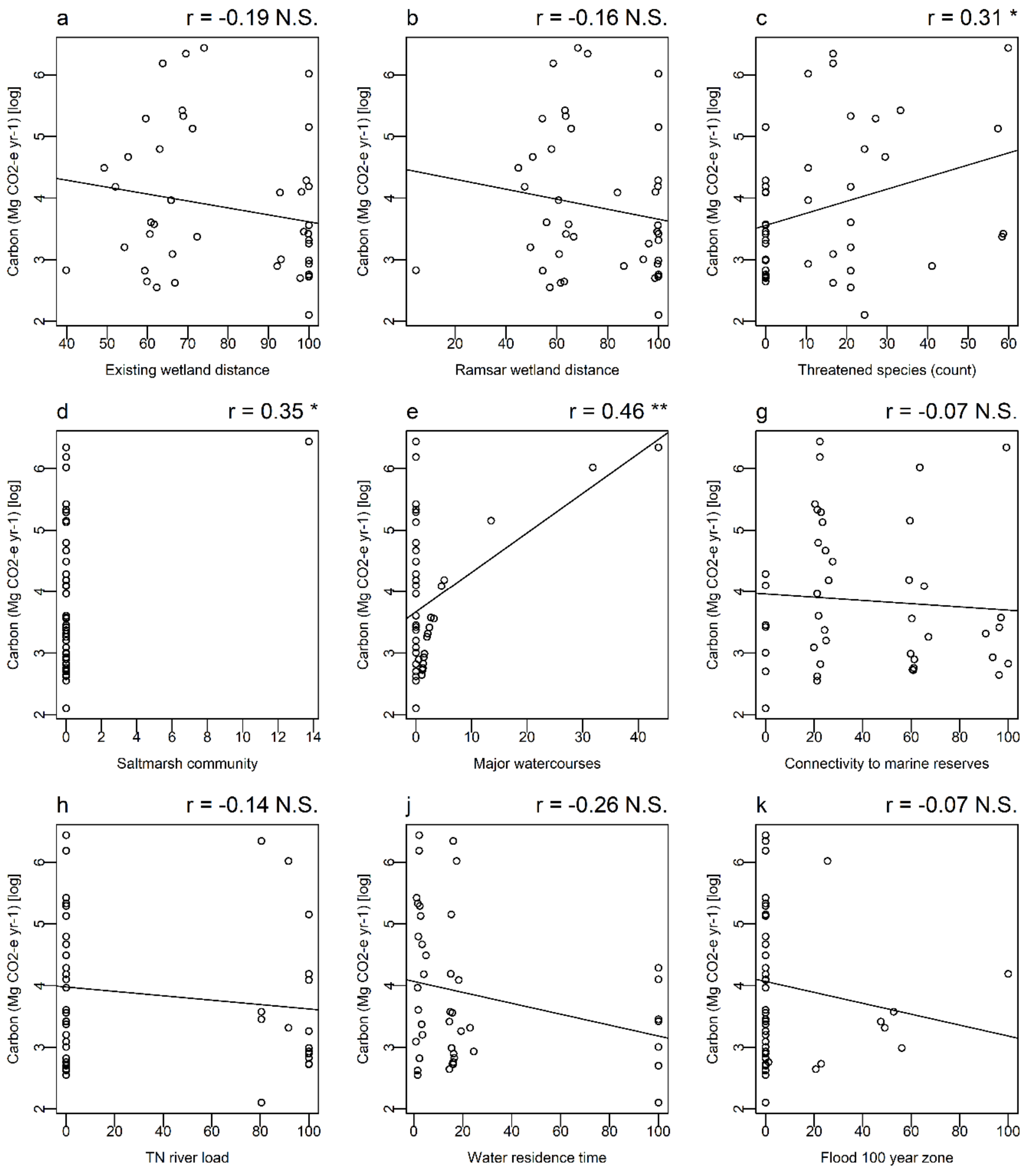

Figure 12 The relationships between mean annual carbon abatement and co-benefit indicators per Peel-Harvey restoration site: Biodiversity - distance to existing wetland (a), distance to Ramsar wetland (b), number of threatened species (c), habitat for the saltmarsh threatened ecological community (d); Fisheries – area of major watercourse (e), connectivity with marine reserves (f); Water quality - Total Nitrogen (TN) river concentration (h), and water residence time (j); Flood mitigation - area of 100 year flood zone (k). Lines show fits from a linear regression. Pearson correlation coefficients (r) and significance levels ($p<0.05$*, $p<0.01$**, $p<0.001$***) are shown for each relationship. Indicator values have been scaled between 0-100.

3.6.3 Ord

Biodiversity co-benefits were rich across the region but not distributed evenly across potential restoration sites (Figure 13). Records of threatened taxa are sparse in the region but three of eight sites have records within a 1km buffer. Species within restoration sites and their buffers include the Purple-crowned Fairy-wren (*Malurus coronatus*), Australasian Bittern (*Botaurus poiciloptilus*), Gouldian Finch (*Erythura gouldiae*), and the Knob Peak Camaenid Snail (*Ninbingia bulla*) as well as several migratory shorebirds. A potential 682 ha of Purple-crowned Fairy-wren habitat could exist within restoration sites, with one site having a record of the species from 2018 in mangrove habitat. Five of eight sites adjoin a Ramsar wetland, and six of eight join an important wetland area. Those that did not join a wetland are within 25 km of wetlands.

Fisheries co-benefits were present on all sites and more evenly distributed than biodiversity (Figure 13). A total of 11,295 ha of potential fish habitat were identified, including areas of major perennial waterways and intertidal and sub-tidal habitat. Three out of eight sites were on a perennial tributary or intertidal bay, and all other sites were within 1 km of one. These two elements combined mean the potential for creating positive fisheries co-benefits in the Ord is high.

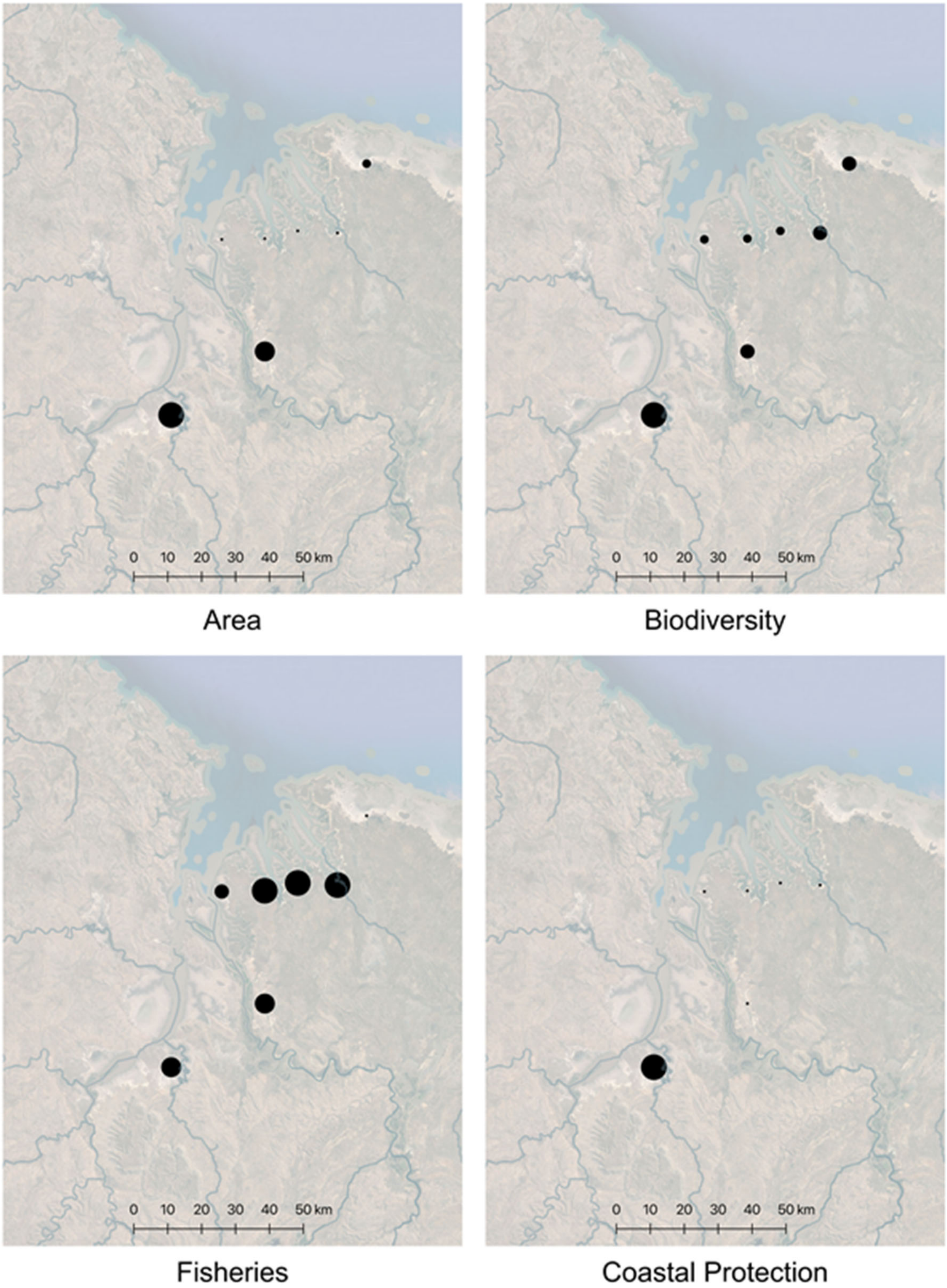

Figure 13 Spatial distribution of potential restoration sites in the Ord River study region, with the area and summed indicators for biodiversity, fisheries and coastal protection co-benefit displayed as scaled icons on their centroid. All attributes are shown in their scaled form (a score of 0-100).

3.7 Indigenous heritage

Indigenous heritage values were not included in the co-benefits measure and cost-effectiveness analyses because this requires engagement with Traditional Custodians on specific sites of interest. Rather we explored the interest for Traditional Custodians to lead or co-manage blue carbon projects through consultation with First Nations people in the Fitzroy Basin and assessed the potential for Traditional Custodian-led blue carbon projects on native title across the case study regions.

Traditional Custodians of the Fitzroy Basin coastal region are interested in undertaking long-term blue carbon projects as part of their cultural obligations which includes caring for their areas of Country, 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, we identified 12 restoration sites with 696 ha of non-exclusive native title and 275 ha of potential for native title agreements (lease hold, state or crown land) that may be suitable for restoration in partnership with registered Aboriginal parties, including the Bailai, Gurang, Gooreng Gooreng, Taribelang Bunda People and the Darumbal People. There is also potential for partnership with the Koinjmal people, who are the custodians of the sites north of the Styx including St Lawrence.

In the Peel-Harvey region, 50 ha of restoration sites contained a cultural heritage site. This included significant areas of ceremonial sites, burial sites, and artefact scatters. Restoration sites primarily fell within the Gnaala Karla Booja native title assessment area (72%), with the remaining sites within the south-west Boojarah 2 native title assessment area. A database containing lots and land parcels to resume Indigenous ownership and management has been constructed as part of the native title proceedings for the area, and the precise delineation of sites that would directly benefit Traditional Custodians will be identifiable once this data is available publicly.

In the Ord River region, all sites are situated east of the Ord River and the majority fall within the lands of the Miriuwung Gajerrong people, whose native title rights and interests are managed by Yawoorroong Miriwing Gajerrong Yirrgeb Noong Dawang Aboriginal

Corporation, with the remainder within Balanggarra Country (National Native Title Tribunal, 2022).

Spatial data on the restoration opportunities will be provided to the NRM groups and the First Nations representatives that attended the workshop, to support development of blue carbon work packages.

3.8 Regulation and policy constraints

Regional variation in policy pertaining to coastal restoration will be an important factor to consider when we are assessing the feasibility of sites for blue carbon projects. In Queensland, there are a number of policy enablers and barriers that could impact the legal risk and uncertainty surrounding coastal restoration (Bell-James & Lovelock, 2019). At a state level, future enabling policies could include incorporation of restoration in coastal management plans which could be implemented at finer scale levels of local government areas. Other enablers such as the Emissions Reduction Fund offer payments for reductions in carbon emissions to eligible projects, while the Land Restoration Fund in Queensland offers funding for Queensland-based projects that provide co-benefits. Priorities of the Land Restoration Fund include restoration of wetlands and coastal ecosystems that contributes to the health of the Great Barrier Reef under the Reef 2050 Water Quality Improvement Plan (Australian and Queensland Government, 2018). These priorities align with and could be co-opted to fund blue carbon restoration projects. In the past, a lack of accepted methodology for blue carbon accounting has been a barrier in Australia but with the adopted blue carbon method, tidal restoration is now an eligible activity (Clean Energy Regulator, 2022), but consideration of other activities, in addition to tidal restoration, would increase project opportunities.

In Queensland, current barriers include the multiagency approval process where approvals for coastal restoration projects are split between fisheries, environment, and planning departments, which can be time and cost prohibitive (Shumway et al., 2021). In the stakeholder workshop it was identified that fish barrier remediation in Fitzroy Basin (which is similar to tidal restoration), incurred AU $45,000 in approval fees associated with impacts to marine plants, tidal works and maritime safety, coastal management district tidal works, and waterway barrier works, and required expertise to complete the approval documentation. For environmental projects, half of the fees can be reimbursed from the state assessment and referral agency. Further, approval, when gained, usually falls into two categories: development and research permits. Development permits are intended for infrastructure projects, while research permits are usually for short, pilot projects, neither of which are suitable for long-term environmental restoration projects. Queensland has convened a working group to find solutions to support uptake of restoration, for example the Prioritisation of Rehabilitation and Research for Aquatic Ecosystems (Queensland Wetlands Program, 2020).

In Western Australia, there have been policy initiatives that support coastal restoration projects (Department of Planning, 2021). For example, the Department of Planning, Lands and Heritage have grants available to undertake rehabilitation and restoration under their Coastwest scheme. Similarly, the Department of Biodiversity, Conservation and Attractions have a Riverbank Grants Scheme that supports projects that undertake foreshore protection and rehabilitation in the Swan and Canning Riverpark. Similar schemes that integrate coastal restoration within wider management plans could be implemented in other regions in Western Australia, depending on the needs of the local areas and communities. In the Western Australia the approval process for coastal restoration was not discussed in the Peel-Harvey workshop, however a national survey identified that permitting criteria and approval processes are a major barrier across Australia (Saunders et al., 2022).

The permit and approval process from state governments could be streamlined for tidal restoration projects, and clearer articulation of steps to gain approvals would help reduce the barrier to project implementation (Saunders et al., 2022).

4. Discussion

We developed a process to select sites for tidal restoration of coastal wetland ecosystems for blue carbon across different regions of Australia. Based on biophysical suitability of land (receives highest astronomical tides and was historically wetland vegetation) and the potential for conversion of agricultural land-uses, a total of 13,874 ha was identified as potentially restorable via the blue carbon tidal restoration method across the Fitzroy Basin region in central QLD with much less in the Peel-Harvey region in south-west WA (348 ha) than the Fitzroy Basin. In the Ord River region in east Kimberley north-west WA, 24,123 ha of land may be potentially restorable via other restoration activities to prevent coastal wetland degradation. Variation in the tidal range among the three different regions assessed was linked to the level of the restoration opportunity identified. In temperate south-west WA, the tidal ranges are low (0.5 -1 m) compared to 2.2 - 6.6 m for the dry tropics of central QLD and 7 – 9 m for the monsoon tropics of north-west WA, which affected the restoration opportunity identified. Restorable areas were much greater when sea level rise was considered, indicating that opportunities may increase in the future.

We used the Australian Government blue carbon method to estimate net carbon abatement from baseline and coastal wetland emissions and removals and calculate the potential carbon credits. We were only able to apply this to the Fitzroy Basin and Peel-Harvey regions, because of the limited potential for tidal restoration found in the Ord River region. Restoration of all potential area in the Fitzroy Basin and Peel-Harvey would equate to a net carbon abatement of 162,000 Mg CO_2-e yr^{-1} and 4,312 CO_2-e yr^{-1}, respectively. We applied a preliminary approach to estimate potential carbon abatement with reduced grazing in the Ord River. Rehabilitation of all the potential area in the Ord River could abate 7,237 Mg CO_2-e yr^{-1} from avoided emissions and soil organic carbon losses. Levels of abatement could be increased with evidence that grazing reduced woody biomass accumulation. We did not undertake the economic analysis for the Ord River because of uncertainty with this estimate and lack of data on condition of wetlands and farm gross margins for the region.

We find that for Fitzroy Basin, 51% of 13,874 ha of the potential restoration area would be profitable under a higher carbon price that can be achieved for carbon projects with social benefits. But under the current average carbon price of AU$ 16.22 per tonne CO_2-e, tidal restoration of grazing land would not be economically feasible in the Fitzroy Basin or Peel-Harvey, where land-uses are dominated by beef production. In contrast, tidal restoration of sugarcane and grazing land in the Wet Tropics of northern Queensland was found to be profitable using a carbon price of AU$ 13.85 per tonne CO_2-e on 3,399 ha (67%) of restoration opportunity. The reduction in economic feasibility reflects differences in the net carbon abatement calculated from different accounting methodologies, as well as differences in the land-uses and coastal wetland vegetation.

The Wet Tropics study followed IPCC methodologies and accounted for N_2O emissions from fertiliser application, but did not include CO_2 removals from degraded wetlands in the baseline land use, which increased the net carbon abatement. The mean carbon abatement in the Wet Tropics varied across catchments, with a mean of 47.3 Mg CO_2-e ha^{-1} yr^{-1} (Hagger et al., 2022). The mean carbon abatement in Peel Harvey and Fitzroy Basin also varied across catchments but was overall much lower (12.4 Mg CO_2-e ha^{-1} yr^{-1} and 11.7 Mg CO_2-e ha^{-1} yr^{-1}, respectively). Thus, profitability using the conservative approach adopted in the blue carbon tidal restoration method may be enhanced by incorporating co-benefits which may allow blue carbon projects to attain higher carbon prices by bunding ecosystem services (Hagger et al., 2022). However, even under a higher carbon price, no profitable sites were found in Peel-Harvey, even though mean carbon abatement per hectare was slightly higher in Peel-Harvey than Fitzroy Basin. This is because most of the sites were small (up to 48 ha) as opposed to large sites of up to 5,000 ha in Fitzroy Basin, and efficiencies from economies of scale that may be achieved in the Fitzroy Basin are not available in Peel-Harvey and therefore cannot reduce the impact of capital costs on long-term profits. Also, there was no available data to assess variation in farm gross margins across land types in the Peel-Harvey. While higher carbon prices are likely needed to out complete beef production, it is possible that restoration in Peel-Harvey could also be financially feasible if restoration costs or farm gross margins are lower than assumed in our analyses. Sustainable production can also be incorporated into the farm enterprise, for example, seasonal grazing of restored coastal wetlands to manage introduced grasses may be an option to minimise the opportunity cost of beef production (Queensland Wetlands Program, 2008). We also found the opportunity for restoration to increase significantly with a +1 m sea-level rise across the three case study regions, which is projected by 2100. With sea-level rise intensifying agricultural land degradation (Rowland et al., In Review), blue carbon restoration may provide supplementary income to transition businesses through climate-related loss of beef production. Carbon prices are likely to increase over the life of a blue carbon project, and many ACCUs are expected to be sold to private buyers at higher prices available in the private voluntary market (Kuwae et al., 2022).

We incorporate within our analysis a spatially explicit multifunctional landscape approach to select economically feasible sites for blue carbon restoration that maximise co-benefits for biodiversity, fisheries, DIN removal, and flood mitigation using publicly available data. We revealed that the CE prioritisation does not work when NPV is negative (restoration is a cost). This is because the function finds the least cost per percent of summed co-benefits, which also provides the least carbon abatement because the site is small. This is problematic when there is no pool of profitable sites to choose from. In regions where most of the restoration opportunity comprises many small sites, it may be preferable to select sites based on optimising carbon abatement and co-benefits (for example, by including carbon abatement in the multifunctionality measure) for a budget (Adame et al., 2015; Possingham et al., 2015). It also may be possible to aggregate sites to achieve economies of scale (Canning et al., 2021). In the Ord River we aggregated sites within 5 km of one another to form 8 large sites, which is likely to be possible because the size of landholdings are much larger in northern Australia than in southern Australia. We suggest incorporating aggregation of sites into the framework, which will require cadastral and property ownership data to identify parcels of land that can realistically be aggregated. Traditional Custodians from multiple parties across a region are willing to work together with the NRM group and industry partners to develop work packages. This may also be a possibility for farmers who are interested in blue carbon projects to diversify their income. The benefits of an aggregated agreement are shared costs and expertise, however effective governance would be needed. This may be effective if the NRM group has funding to facilitate partnerships and provide technical support on legal agreements, approvals and implementation.

We developed co-benefit indicators based on scientific understanding of what influences biodiversity and fisheries enhancement, water quality improvements, and flood mitigation in coastal wetland ecosystems. Measures of those indicators were based on availability of spatial datasets, which varied between case study regions. However, measures of indicators can be adapted to areas of interest. Our indicators are based on likely outcomes after restoration, and not monitoring of ecosystem functions, which is usually expected for claiming of co-benefits in restoration projects (Butler, 2021). This is useful for spatial prioritisation of blue carbon restoration sites that provide co-benefits, however payments for those co-benefits will be subject to monitoring under verified methods, such as Co-benefit Standard used by the Land Restoration Fund (Queensland Government, 2021). Unlike the blue carbon tidal restoration method, where the proponent can use the national blue carbon calculator (BlueCAM) to calculate carbon credits based on delineating carbon estimation areas across the restoration site (Lovelock et al., In Review).

We did not find any trade-offs between estimated co-benefits and carbon abatement among different sites similar to other studies globally (Adame et al., 2015). Generally those sites that were larger in area had higher carbon abatement and also co-benefits for area-based indicators, such as potential habitat for threatened species and communities. But for distance-based indicators, such as connectivity with existing wetlands and Ramsar wetlands, there was no synergy or penalty with higher carbon abatement. Within the DIN removal service however there may be a trade-off if catchments have both high DIN and TSS concentrations, where sediments can limit denitrification (Kavehei, Roberts, et al., 2021; Reis et al., 2019). While we incorporated this within the weighted multifunctionality measure for Fitzroy Basin, it wouldn't affect DIN removal by plants (Wallace et al., In Review). Additionally, one of the most important predictors of DIN removal is hydrology, which includes residence time, connectivity and inundation frequency (Adame et al., 2019; Kavehei, Hasan, et al., 2021). This information is difficult to obtain at the scales of this study and should be considered a limitation of the co-benefit of water quality improvement.

For fisheries production, the number and measurement of indicators also varied by case study region given characteristics of the system and data availability. In Fitzroy Basin, sites in the lower intertidal zone and with connectivity to fish habitat areas and permanent waterways were assumed to provide habitat for fisheries production. While in the Peel-Harvey and Ord River, sites with permanent waterways and connectivity to marine reserves via flows paths were considered to provide fisheries habitat. Similarly with coastal protection, sites in the Fitzroy Basin were assessed for the potential to protect against both coastal and inland flooding. While in the Peel-Harvey and Ord River, sites were assessed for inland flood and coastal flood mitigation, respectively, because the coastal dunes offer direct protection in the Peel-Harvey and there is no flood projection model for the Ord River, presumably because it's a remote region. Omission of indicators doesn't affect the multifunctionality measure overall, because a higher weighting is given to that service indicator within the service.

In both the Fitzroy Basin and Peel-Harvey, land management priorities are to improve the extent and condition of coastal ecosystems to improve water quality and estuary health (Australian and Queensland Government, 2018; Department of Environment and Science, 2020b; Environmental Protection Authority, 2008), thus blue carbon projects should benefit from local incentives for these services, such as Reef Credits which provide offset credits for DIN reductions in the Great Barrier Reef catchments (Green Collar, 2022). In the Peel-

Harvey, the NRM groups also value protecting the ecological character of several internationally important Ramsar wetlands. Preserving habitat for threatened and migratory birds (e.g. Capricorn Yellow Chat and waterbird foraging and breeding habitats) was an important component to consider when identifying restoration opportunities in both the Fitzroy Basin and Ord River. Given the different habitat requirements of species, the identification of restoration sites need to be supported by local data of species distributions and habitat preferences, which may limit the potential for national level assessments. Increasing landscape resilience to sea level rise was not a high priority raised during the stakeholder meetings, yet maintaining wetland cover may increase tidal attenuation and therefore protect human land-uses in the future (Reed et al., 2018), aligning with climate adaptation plans and allowing habitat migration for diverse range of species (Bell-James et al., 2022).

Cultural heritage values were incorporated into our analyses by identifying opportunities for Traditional Custodian-led blue carbon projects on native title, leasehold and crown land, and Country with registered parties. Our analyses revealed that there are opportunities for Traditional Custodians to lead or co-manage blue carbon projects, although there is currently uncertainty in Aboriginal land title determinations. In the Fitzroy Basin 12 sites may be available for partnerships, while in the 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). Even though our assessment did not identify profitable restoration sites in the Peel-Harvey because sites are small (and data sources were limited), levels of co-benefits from restoration of coastal wetlands are high and are likely significant in restoring Country, protecting Ramsar wetlands, and improving estuary health. In the Ord River, we did not identify opportunities for Traditional Custodian-led tidal restoration projects, but other activities, e.g. management of disturbance from feral animals or reduction of the impacts of infrastructure including rail, roads and associated culverts and other structures, may be possible on native title land. In the Fitzroy Basin, Traditional Custodians are interested in undertaking long-term blue carbon projects as part of their cultural obligations. Stakeholder meetings indicated that to achieve success projects need to offer sustainable, long-term funding and be Traditional Custodian-led to 'heal and restore sick Country', deliver sustainable employment and education outcomes, and embed funded mentors and Elders to enrich knowledge-sharing.

In the absence of Australian coastal wetland restoration costs, we used global published costs on mangrove and saltmarsh restoration projects (Bayraktarov et al., 2016), using a lower estimate from hydrological restoration of saltmarsh in developed countries in the base scenario and a higher estimate from hydrological restoration of mangroves in developed countries to test for sensitivity. However, we found that increasing the restoration cost did not greatly affect the number of profitable sites returned (none of restoration sites were profitable in the base scenario with the lower restoration cost in either study region), nor did it significantly affect the ranking of sites by NPV. Better data on tidal restoration costs in different regions of Australia, for example for removal of a bund wall at Mungalla wetlands in north Queensland (Karim et al., 2021) is needed to identify where efficiencies in restoration planning and implementation may help to enhance profitability. Coastal land is also highly regulated and state approvals are needed under various legislation. Government approval processes were identified as a major barrier across Australia (Saunders et al., 2022) and approval costs can be substantial in some cases, likely reducing the feasibility of coastal wetland restoration at small scales. Restoration and maintenance costs and other costs not included in the analyses (e.g. approvals) should be included in the project costs in a case-by-case and standardised manner (Kavehei, Hasan, et al., 2021).

Another important policy consideration for implementing blue carbon tidal restoration projects is land tenure; who has the right to the land on which the project is located and for how long, and in turn, who has the right to carbon credits flowing from the project. On leasehold land, the lease term and purpose is required to meet the duration of the blue carbon project. On freehold land, landowners may need to be granted use of land below the tidal boundary, which is generally owned by the state (Bell-James & Lovelock, 2019). Altering leases (if legally possible) and negotiating carbon rights adds an additional administrative burden on projects which incurs costs. Clear policies to support tidal restoration of coastal wetlands, could increase the attractiveness of coastal wetland restoration, particularly if administrative processes were simpler and easier to negotiate (lower costs to projects).

We found that economic feasibility of restoring grazing land varies with the type of beef enterprise. In the Fitzroy Basin, native wetland vegetation has been cleared and replaced by ponded pastures dominated by introduced plants to allow grazing during drier seasons (Jamieson & Bourne, Unknown; Queensland Wetlands Program, 2008). Ponded pastures are highly productive and valued by farmers. We put a premium on ponded pastures by assigning higher farm gross margins on freshwater plain land types, however in some of these properties restoration was still profitable, likely because of the large avoided N_2O and CH_4 emissions from these wetlands (Iram et al., 2021). While on a paddock level they might be profitable, at a property level they might not be, and future economic analysis may benefit from bio-economic modelling for whole of farm decision making (Kragt et al., 2016).

Our stakeholder workshop with Fitzroy Basin Association indicated that the most realistic opportunities for blue carbon tidal restoration projects on grazing land in the Fitzroy Basin are improving degraded natural systems by removing cattle, not through tidal restoration of ponded pastures. We also revealed limited opportunity for tidal restoration in the Ord River floodplain; intensive agricultural areas that have been cleared for irrigated cropping and grazing as part of the Ord River irrigation scheme are not influenced by tidal flows. Furthermore, all restoration opportunity in the Ord River region is mapped as native remnant vegetation that is used for grazing.

Our mapping of vegetation types cannot reveal variation in the condition of coastal wetlands and particularly if they are in poor condition. Those coastal wetlands in poor condition may have the potential for restoration activities which may result in carbon abatement (Macreadie et al., 2017). Poor condition of coastal wetlands in Australia may be caused by disturbances of soils and vegetation (Creighton et al., 2015; Finlayson & Rea, 1999) from high densities of feral animals such as pigs, cattle, and buffalo (Waltham & Schaffer, 2021) as well as other human mediated disturbances (Creighton et al., 2015). In northern Australia, soil disturbances could be managed to enhance carbon abatement (Gehrke, 2009; Robson et al., 2013). Although there is currently no ERF method to obtain carbon credits for grazing or feral animal management, future methods may consider awarding carbon credits for these activities, that could support restoration in the Ord River and other northern regions of Australia. Our stakeholder meetings indicated there is wide-spread interest in restoring coastal wetland condition, particularly from removing cattle, pigs and buffalo. In northern

Australia, there are large opportunities these restoration activities to be Traditional Custodian-led. For example, in the Kimberley's where First Nations people have native title on 85% of the coastal land.

Our mapping of tidal inundation also does not incorporate understanding of catchment hydrology. While hydrodynamic modelling can demonstrate the tidal ingress that is possible from bund removal (Karim et al., 2021), the outcome can be much lower because of site and climate dynamics, such as varying rainfall and effects on hydrology (Abbott et al., 2020). Therefore, confirming the feasibility of blue carbon restoration sites will likely require local knowledge and understanding of site processes.

Our case study regions spanned regions with (1) both large (Fitzroy Basin and Ord River) and small tidal ranges (Peel-Harvey), (2) vastly different levels of direct modification, with the Fitzroy Basin and Peel-Harvey being highly modified by drainage and levees/bunds, while the Ord River region has been influenced by activities in the catchment but limited direct hydrological modifications of coastal wetlands, and (3) different co-benefits. Although our analyses revealed important factors that can be used to identify opportunities for blue carbon tidal restoration (and data deficits, see below), examination of other regions with more moderate tidal range, different land-uses (e.g. cropping) and different co-benefits would increase the breadth of understanding of the feasibility of tidal restoration in Australia.

Our analyses identified data deficits that limit the ability to identify opportunities for tidal restoration of coastal wetlands:

- Detailed mapping of hydrological modifications (e.g. drains and tidal exclusion structures) on floodplains is limited in many regions, but if available would enhance mapping of restoration opportunities. A consistent map of drainage lines is available in some states (e.g. Queensland), but not others (e.g. Western Australia).

- Improved mapping of the level of tidal inundation and its potential changes with sea level rise would enable greater accuracy in determining the suitability of land for tidal restoration. An example of the influence of the Peel-Harvey estuary on the HAT and tidal planes using an unstructured mesh hydrodynamic model has been developed to understand the uncertainty of the blue carbon method to varying input data (CSIRO Oceans and Atmosphere, unpublished). It is not feasible to develop hydrodynamic models everywhere, however there are opportunities to incorporate existing tidal models into a framework to identify restoration opportunities to analyse tidal processes, for example hydrodynamic modelling of the WA estuaries by University of Western Australia and WA Department of Water.

- Fine scale data of farm gross margins would improve our assessment of economic feasibility through improving knowledge of opportunity costs. Although some data on farm gross margins exists for WA, this was not available for the project and instead we used data from South Australia for a similar climate zone. But there are important differences in length of supply chains and other factors between South Australia and Western Australia that reduces confidence in the economic analyses when using data from other localities.

- Regional data on restoration costs including approval, capital, and maintenance costs for restoration activities would also improve our assessment of economic feasibility.
- Identification of cultural heritage values would allow explicit inclusion of cultural heritage in the co-benefits. Future analyses might include the presence of important food plants and animals, presence of middens, camps and ceremonial grounds. Although Indigenous land tenure mapping is being developed, this is sensitive data that may not be publicly available and in some cases land tenure determination is still being resolved. Engagement with Traditional Custodians should always be undertaken when planning blue carbon projects.
- Mapping of infrastructure, such as roads and railways, that may be altering tidal flows from causeways and culverts, may provide an opportunity for tidal restoration in addition to agricultural land-uses considered in this study. Further study of infrastructure impacts on coastal wetlands and restoration opportunities, as well as the potential of coastal wetland restoration to enhance protection of infrastructure could be considered in further analyses.
- Mapping the condition of coastal wetlands is not available, but if available would enable exploration of restoration activities to enhance condition, which would increase carbon abatement and co-benefits. Characterising linkages among coastal wetland condition, avoided emissions, carbon sequestration, and co-benefits could form the basis of new blue carbon methodologies for restoration of coastal wetlands via activities in addition to tidal restoration.

5. Recommendations – Towards a framework for identifying restoration opportunities for blue carbon ecosystems

Discovering the opportunities for blue carbon restoration goes beyond identifying biophysically suitable sites, and instead is highly dependent on a range of factors including hydrological modifications, income from land-uses, the distribution of threatened species, and land ownership. To establish a comprehensive framework for identifying blue carbon tidal restoration opportunities recommendations arising from our research include:

- Take a place-based approach to identify coastal wetland restoration sites based on local-level assessments of restoration opportunities and site knowledge, such as catchment hydrology, land tenure, and threatened species distributions and habitats. This approach would facilitate engagement with Traditional Custodians.

- Test the multifunctionality approach on field data to refine standardised and representative metrics for co-benefits based on the availability of spatial data, that can be used in an accounting framework to allow stacking of payments for ecosystem services, such as biodiversity stewardship or nutrient offsets. Coastal wetlands provide a wide range of unique co-benefits that provide important ecosystem services to coastal communities and complement the carbon abatement achieved. For example, each case study region had a nationally listed threatened species or community for which coastal wetland restoration can enhance its conservation and adaptation to sea-level rise. Use of data from other locations should always be acknowledged. The use of local data is preferable, for example local hydrology, vegetation cover and wetland condition could improve our estimations of the provision of DIN removal services.

- Development of an Indigenous blue carbon strategy that is action-orientated and regionally specific to ensure collaboration with Traditional Custodians in planning of blue carbon restoration programs. Our project revealed high levels of interest of Traditional Custodians in restoration of coastal wetlands, but limited potential for the use of the tidal restoration blue carbon method in some regions.

- Data on restoration opportunities from regional assessments should be made available to Traditional Custodians. Additionally, support to Indigenous groups, the NRM groups, and industry partners is recommended to help develop work packages and bundling multiple restoration projects to enable Traditional Custodian-led blue carbon programs.

- Increase the range of case studies to include regions with different tidal ranges, land-uses and levels of biodiversity in order to uncover the full range of factors that should be considered in selecting sites for blue carbon tidal restoration and further opportunities to assess potential trade-offs among carbon abatement and co-benefits. Develop a national data set of economic returns from different land uses.

- Test an alternative cost-effectiveness approach for selecting sites for tidal restoration that optimises carbon abatement and co-benefits for a given cost, such as using Marxan. This could be used for regions where blue carbon abatement is not economically feasible (based on returns calculated with the non-local input parameters used such as in the Peel-Harvey).

- Investigate the feasibility of additional blue carbon methods that focus on preventing or reducing disturbance to coastal wetlands by removing non-native ungulates in collaboration with Indigenous partners to increase the opportunity for blue carbon restoration in northern Australia. Gather data on emissions and removals in degraded wetlands against natural wetlands in good condition and assess the economic feasibility given costs associated with fencing and other management of non-native ungulates.

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Appendix A Supplementary Methods

Table A1 Australian Land Use and Management Classification (ALUM) codes and hydrologically modified wetlands included in the identification of restoration sites

Study region	ALUM codes	Local hydrology modifier codes
Fitzroy Basin	3.2.2 (Grazing modified pasture – woody fodder plants) 2.1.0 (Grazing native vegetation) 4.2.0 (Grazing irrigated modified pastures) 4.2.1 (Grazing irrigated woody fodder plants) 6.2.2 (Water storage – intensive use/farm dams) 1.3.1 (Defence) 6.5.2 (Marsh/wetland production)	H2M2 H2M2a H2M2b H2M3 H2M1
Peel-Harvey	2.1 Grazing native 3.2 Grazing modified 4.2 Grazing irrigated	NA
Ord River	2.1 Grazing native 3.2 Grazing modified 3.3 Cropping 4.2 Grazing irrigated 4.3 Irrigated cropping 5.2.8 Abandoned intensive animal production	NA

Table A2 Areas of agricultural land-uses within the Highest Astronomical Tide (HAT) level of each case study region

Land-use	Fitzroy Basin		Peel-Harvey		Ord River	
	ha	prop	ha	prop	ha	prop
2.1.0 Grazing native vegetation	14171	0.39			414,727	1
2.2.0 Production native forests	120	0.003				
3.2.0 Grazing modified pastures			578.265	0.89		
3.3 Cropping	36	0.001	58.49075	0.09		
4.2.0 Grazing irrigated modified pastures			5.710714	0.009		
4.3.5 Irrigated cropping	0.04	<0.001				
4.4.1 Irrigated perennial horticulture	0.02	<0.001				
4.5.4 Irrigated seasonal horticulture	0.07	<0.001				
5.2.5, 5.2.6 Intensive animal production	9	<0.001				
5.1.1 Intensive horticulture	0.09	<0.001				
6.5.2 Marsh/wetland	21333	0.58				
6.2.2 Reservoir/dam	826	0.02				
5.2.6 Horse studs			3.768129	0.006		

Table A3 Pre-clear wetland vegetation types included in the identification of restoration sites

Study region	Vegetation descriptions	Mapping source
Fitzroy Basin	Dominant Broad Vegetation Groups (BVGs) containing wetland ecosystems in coastal areas (Neldner, 2017), Appendix 2: 35a (mangrove) 35b (saltmarsh) 34c and 34g (sedgeland on floodplains) 22a, 22b and 22c (*Melaleuca* spp. swamps and forest on streams and banks) 4a and 4b (vine forest) 15b, 16a, 16c, and 19b (*Eucalyptus* spp. forest on streams and banks) 16d and 34d (waterholes and swamps) 26a (*Acacia* spp. forest on streams and banks) 29a (wet heath) Estuary *Note: no BVG 15b, 19b, 26a and 34g in Fitzroy Basin region.*	(DES 2019a)
Peel Harvey	Vegetation descriptions containing wetland ecosystems (Beard et al., 2013): 3. Woodland: jarrah, marri, wandoo, tuart and flooded gum 6. Low forest: acacia, peppermint, coastal moort, Rottnest pine or mixed tropical forest 9. Low woodland, open low woodland: other species 10. Mangroves: low forest (Kimberley) or thicket (Pilbara) mangroves (Avicennia marina, Rhizophora stylosa, Bruguiera exaristata) 14. Thicket: wattle, casuarina and teatree (Acacia–Allocasuarina–Melaleuca alliance). 29. Short bunch-grass savanna 32. Riverine sedgeland/grassland with trees 33 Sedgeland: (mainly in the South West) Cyperaceae, Restionaceae, Juncaceae 38. Shrub-steppe 51 Salt lake, lagoon, claypan 53 Tidal mud flat Mosaic 101. Medium forest or woodland/Low woodland/Low forest or woodland Mosaic 106. Low woodland/Scrub or thicket Mosaic 107. Scrub-heath/Thicket Mosaic 116. Short bunch-grass savanna/ Grass-steppe	(DPIRD 2017)
Ord River	As Peel-Harvey and South West	(DPIRD2017)

Table A4 Average tidal ranges for each catchment in case study regions

Study region	Catchment	Place name	HAT m above AHD (average in catchment)	HAT + 0.71m SLR*
Fitzroy Basin	Boyne	Gatcombe Head	2.12 (2.28)	2.99
		South Trees Wharf	2.43	
	Calliope	Gladstone	2.49 (2.60)	3.31
		Fishermans Landing	2.7	
	Curtis Island	Graham Creek	2.73 (2.91)	3.62
		The Narrows (Boat Creek)	2.92	
		The Narrows (Ramsay Crossing)	3.16	
		Sea Hill	2.82	
	Fitzroy	Rockhampton	3.56 (3.32)	4.03
		Port Alma	3.08	
	Shoalwater	Thirsty Sound	4.12	4.83
	Styx	McEwen Islet	4.97	5.68
	Waterpark	Cape Manifold	2.79	3.47
		Port Clinton	2.76	
		Rosslyn Bay	2.72 (2.76)	
Peel-Harvey	Blackwood	Busselton	0.55	1.26
	Busselton	Busselton	0.55	1.26
	Preston	Bunbury	0.67	1.38
	Collie	Bunbury	0.67	1.38
	Harvey	Bunbury	0.67	1.38
	Murray	Fremantle	0.81	1.52
Ord River	Ord River	Wyndham	4.25	4.96

*Sea-level rise for RCP 8.5 2081-2100 – 0.71 (Oppenheimer et al., 2019)

Table A5 Methods for estimating avoided greenhouse gas emissions and removals from ceasing agricultural land use, and associated equations, emission factors (EF), accumulation rate (AR), conversion factors (CF), and global warming potentials (GWP)

Removal or emission	Land use	Method	Equation	Accumulation rate (Mg C ha^{-1} yr^{-1})	Emission factor (kg ha^{-1} yr^{-1}) or default stock change factors	CF⁺	GWP⁺	Assumptions
CO_2 soil carbon loss	Grazing	IPCC Tier 1/2 approach Vol. 4 Ch. 6.2.3, using default stock change factors (Lasco et al., 2006) and soil organic carbon (SOC) stocks within top 30cm of soil calculated from the Australian baseline map of SOC in in ArcGIS (Viscarra Rossell et al. 2014), as adopted in Australian Blue Carbon method (Lovelock et al., In Review).	$CO_2e(MgYr^{-1}) = (SOC\ stock\ (Mg\ site^{-1}) - (SOC\ stock\ (Mg\ site^{-1}) \times F_{LU} \times F_{MG} \times F_I)/\ 20)) \times CF \times GWP$		F_{LU} = 1 (permanent grassland), F_{MG} = 0.97 (tropical moderately degraded grassland), F_I = 1 (no additional improvements)	3.67	1	Moderately degraded management regime assumed as 28.3% of grazing land in Fitzroy was under best management practice (GBR Report Card 2017-2018).
CH_4 from flooded agricultural land, managed wet meadow or pasture	Grazing	IPCC Tier 2 approach using Australian specific emission factor and area mapped as flooded agricultural land using the QLD wetlands data (hydrological modified wetlands H2M2, H2M3 and H2M5) for Fitzroy Basin (DES 2020c). There was no data available to assess this for Peel-Harvey.	$CO_2e(MgYr^{-1}) = (Area(ha) \times EF/1000) \times GWP$		325	1.33	28	
N_2O from flooded agricultural land, managed	Grazing	IPCC Tier 2 approach using Australian specific emission factor (Lovelock et al., In Review) and area mapped as flooded agricultural land calculated	$CO_2e(MgYr^{-1}) = (Area(ha) \times EF/1000) \times GWP$		14	1.33	28	

Removal or emission	Land use	Method	Equation	Accumulation rate (Mg C ha^{-1} yr^{-1})	Emission factor (kg ha^{-1} yr^{-1}) or default stock change factors	CF⁺	GWP⁺	Assumptions
wet meadow or pasture		using the QLD wetlands data (hydrological modified wetlands H2M2, H2M3 and H2M5) for Fitzroy Basin (DES 2020c). There was no data available to assess this for Peel-Harvey.						
CH_4 from ponds and other constructed water bodies	Grazing	IPCC Tier 2 approach using Australian specific emission factor (Lovelock et al. 2021) and area mapped as farm ponds and dams using a combination of QLD water storage points and reservoir data (Department of Resources, 2021d, 2021e) for Fitzroy Basin, and polygon data (DPIRD 2020a) for Peel-Harvey.	$CO_2e(MgYr^{-1}) = (Area(ha) \times EF/1000) \times GWP$		226.3	1.33	28	
Soil carbon accumulation in hydrologically disturbed mangrove, saltmarsh, and herbaceous settings	Disturbed wetlands	IPCC Tier 2 approach using Australia specific emission factor and area mapped as degraded wetlands identified from regrowth and remnant vegetation (DES 2019a, 2020a) for Fitzroy Basin and remnant vegetation (DPIRD 2020b) for Peel-Harvey.	$CO_2e(MgYr^{-1}) = Area\ (ha) \times AR \times GWP$	0.47 (mangrove or saltmarsh) 0.61 (supratidal)		3.67	1	

Table A6 Methods for estimating CO_2 removals and greenhouse gas emissions in coastal wetland ecosystems, and associated equations, carbon accumulation rates (AR), emission factors (EF), conversion factors (CF) and global warming potential (GWP) applied

Removal or emission	Wetland type	Methods	Equation	Accumulation rate (Mg C ha^{-1} yr^{-1})	Emission Factor (kg CH_4 ha^{-1} yr^{-1})	CF	GWP
Removals							
CO_2 above-ground biomass	Mangrove	AGB accumulation rates were modelled for years 1 to 100 using a logistic growth equation: $\mathrm{AGB(t)} = a \times (\exp(-k/\mathrm{age}))$	$CO_2e(MgYr^{-1}) = Area(ha) \times AR \times CF \times GWP$	Growth curve	NA	3.67	1
	Supratidal forest	where *a* is the mature AGB, *k* is the rate of biomass increase over time (Lovelock et al., In Review). The AGB maximum carbon stocks were 167 Mg C ha^{-1} for mangroves in tropical Australia (Serrano et al., 2019), and 192 Mg C ha^{-1} and 178 Mg C ha^{-1} for supratidal forests in tropical and temperate Australia respectively (Adame et al., 2020). There were no pre-European mangroves mapped in the Peel region. *k* was 29.6 derived from published and unpublished data on mangrove aboveground biomass carbon development (Lovelock et al., In Review). The rate of increase of biomass for supratidal forest was assumed to be similar to that described for mangroves (*k* = 29.6).		Growth curve	NA	3.67	1
	Saltmarsh ≤1 year	Saltmarsh AGB was assumed to accumulate to maturity within 1 year (Lovelock et al., In Review). The accumulation rate for the first year was estimated from the AGB carbon stock for subtropical Australia of 1.36 Mg C ha^{-1}*, as there is no data for tropical Australia and 7.89 Mg C ha^{-1} for temperate Australia (Serrano et al., 2019).		1.36 (tropical), 7.89 (temperate)^	NA	3.67	1

Removal or emission	Wetland type	Methods	Equation	Accumulation rate (Mg C ha^{-1} yr^{-1})	Emission Factor (kg CH_4 ha^{-1} yr^{-1})	CF	GWP
Removals							
Elevation Ranges	Scrub mangrove	A multiplier of 0.7 should be applied to account for reduced AGB accumulation in shrub mangroves in tropical regions in the 0.32 – 1 Standard Tidal Position Index (STPI) (Lovelock et al., In Review).	NA	NA	NA	NA	NA
CO_2 Below-ground biomass	Mangrove	In mangroves the proportion of aboveground to belowground biomass (root shoot ratio, R:S) was assumed to be 0.32 based on studies from Australia (Castaneda-Moya, 2011).	$CO_2e(MgYr^{-1}) = AGBt \times R{:}S \times CF \times GWP$	NA	NA	3.67	1
	Supratidal forest	A R:S of 0.27 was used based on values reported for tropical trees (Mokany et al., 2006).		NA	NA	3.67	1
	Saltmarsh	In saltmarsh, the root biomass is included within the soil carbon accumulation and therefore belowground biomass is zero (Lovelock et al., In Review).		NA	NA	NA	NA
CO_2 Soil carbon	Mangrove	Default values for soil carbon accumulation rates in different ecosystem types have been derived based on a national collation of blue carbon data (Serrano et al., 2019), updated to include recently published and unpublished datasets (Lovelock et al., In Review). There is one national value for each ecosystem, as no significant differences in soil carbon accumulation rates were found among climatic zones.	$CO_2e(MgYr^{-1}) = AR \times CF \times GWP$	0.95	NA	3.67	1
	Saltmarsh			0.48	NA	3.67	1
	Supratidal forest			0.61	NA	3.67	1
Emissions							
CH_4 flooding of wetlands	Mangrove	IPCC Tier 2 approach for CH_4 and N_2O emissions from coastal wetlands following flooding. Emission factors based on median values have been derived for different climate zones in Australian coastal wetlands from published and unpublished data (Lovelock et al.,	$CO_2e(MgYr^{-1}) = Area(ha) \times (EF/1000) \times GWP$	NA	2.19 (tropical humid)	1.33	28
	Saltmarsh			NA	0.11 (tropical humid)	1.33	28
	Supratidal forest			NA	-2.19 (tropical humid)	1.33	28

Removal or emission	Wetland type	Methods	Equation	Accumulation rate (Mg C ha^{-1} yr^{-1})	Emission Factor (kg CH_4 ha^{-1} yr^{-1})	CF	GWP
Removals							
N_2O flooding of wetlands	Mangrove	In Review). Negative values indicate a net sink (uptake) relative to the atmosphere.		NA	0.24 (tropical humid)	1.57	265
	Saltmarsh			NA	0.13 (tropical humid)	1.57	265
	Supratidal forest			NA	0.25 (tropical humid)	1.57	265

*rate for subtropical Australia, no rate available for tropical Australia
^ only assigned for the first year

Table A7 Gross margins per adult equivalent (AE) for productivity groupings (Star et al., 2017)

Productivity grouping	Gross margin per AE AUD 2015 prices	Assumptions
1	312.55	heavier Japanese-oxen
2	241.98	Not specified
3	199.3	Not specified
4	197.43	Not specified
5	182.32	light store cattle

Table A8 Beef enterprises and stocking densities (adult equivalent per ha, AE/ha) recommended for land types in the Kunwarara area (Black Speargrass, Unknown), and productivity groupings and Grazing Land Management (GLM) land types (DAFF, 2021) assigned

Land type	**AE/ha**	**Productivity grouping**	**GLM land type and code**
Saltpan	0	NA	Estuary (AL01), water (AL09)
Marine plains	0.25	2	Marine plains (FT18), Marine plains and tidal flats (MW07)
Freshwater plains	0.49	1	modified wetland (H2M2, H2M2a, H2M2b, H2M3) from Wetlands data (DES 2019), Alluvial flats and plains (MW01), Coastal wetlands (MW05)
Teatree	0.1	5	Tea tree flats (CB12), Coastal sand dunes (FT09), Coastal tea tree plains (FT10), Coastal tea tree plains (MW04)
Blue gum flats	0.31	2	Blue gum flats (CB02), Blue gum / river red gum flats (FT02), Brigalow with melonholes (FT05), Blue gum on cracking clay (IB02), Blue gum on loam and duplex (IB04), Blue gum on alluvial plains (MO01)
Gum clay flats	0.25	2	
Box clay flats	0.21	2	Box flats (FT03), Eucalypts and bloodwood on clay (FT13), Poplar gum woodlands (MW08)
Grey ironbark flats	0.25	1	Loamy alluvials (BD13), Coastal flats with mixed eucalypts on grey clay (FT08), Coolibah floodplains (FT11)
Ironbark-bloodwood country	0.25	2	Gum-topped box (CB04), Ironbark and spotted gum on duplex and loam (CB09), Gum-topped box flats (FT16), Mixed open forests on duplex and loam (MO08), Coastal eucalypt forests and woodlands (MW02), Eucalypt hills and ranges (MW06)
Serpentine country	0.1	5	Softwood scrub (FT29)
Rosewood country	0.1	5	Lancewood - bendee – rosewood (FT17)

Table A9 Cost reduction rates applied to restoration and maintenance costs (Strassburg et al., 2019)

Restoration area (ha)	Restoration cost (USD)	Cost reduction rate
1	12000	NA
>=5	11000	-0.083333
>=10	10000	-0.166667
>=25	9250	-0.229167
>=50	8250	-0.312500
>=100	7500	-0.375000

Table A10 Ecosystem service indicators and measures for each case study region and the weightings applied under the different scenarios

EcosystemS ervice	Indicator	Fitzroy Basin	Peel-Harvey	Ord River	Scaled measure	Weight equal Fitzroy	Weight high Fitzroy	Weight equal Peel	Weight high Peel
Biodiversity	Threatened species diversity	Number of critically endangered, endangered, vulnerable or migratory species under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) and/or the Queensland *Nature Conservation Act 1992* within 1000 m of the restoration site boundary from a Wildnet search of the study region, filtered by verified records (DES, 2022b).	Number of critically endangered, endangered, vulnerable or migratory species within 1000m from Threatened and Priority Flora and Fauna databases (Department of Biodiversity Conservation and Attractions, 2020, 2021). Log+1 as the data was skewed.	Number of critically endangered, endangered, vulnerable or migratory species within 1000m from Threatened and Priority Flora and Fauna databases (Department of Biodiversity Conservation and Attractions, 2020, 2021). Log+1 as the data was skewed.	A count scaled to 0-100 with 100 the most EVNT taxa records	0.0625	0.175	0.0625	0.175
	Habitat for a threatened species or community highly valued in the region	Area of habitat for the Capricorn Yellow Chat (CYC) within each restoration site. Two convex zones encompassing the 17 known CYC sites with a 1 km buffer around them representing the northern subpopulation	Area within the restoration site and a 500m buffer around the site mapped as remnant vegetation types within the EPBC Act listed Temperate saltmarsh community (DSEWPC, 2013),	Area of habitat for the Purple-Crowned Fairy Wren (*Malurus coronatus*, PCFW) within each restoration site was calculated as the area of vegetation types that held records of the species, or matched descriptions of the species habitat from	Am area scaled to 0-100 with 100 the largest area	0.0625	0.175	0.0625	0.175

centred on Broadsound and one the southern and southeastern subpopulations encompassing the Fitzroy River delta and Curtis Island with high likelihood to contain habitat to be used or occupied by the chat (Houston unpublished data). Potential habitat for the CYC across the Fitzroy Basin region has been mapped as pre-clear RE1 or RE2 (DES, 2019a) on marine plains containing *Schoenoplectus subulatus, Sporobolus virginicus, Cyperus alopecuroides* and *Eleocharis spp.*: 8.1.3, 8.1.4, 11.1.1, 11.1.3, 12.1.2, 11.1.2b, 11.3.27x1a, 11.3.27x1b, 11.3.27x1c. Habitat for the CYC has then been mapped as all potential habitat within the population convex zones.

using pre-European vegetation types and remnant vegetation mapping (Department of Primary Industries and Regional Development, 2017, 2020b). Vegetation types 42, 44, 47, 50, 29 and 33 contain elements described in the EPBC listing. Vegetation type 42 is not coastal, so was excluded. The majority of area was vegetation type 50 ('Samphire: Tecticornia spp. communities in saline areas') and type 33 ('Sedgelands: Cyperaceae, Restionacae, Juncacae').

the EPBC Act profile (Department of Agriculture Water and the Environment, 2022) or literature (Skroblin, 2010, 2012), or were within 1 km of a perennial waterway. Vegetation types included were 10 (Mangroves, one record 2018), 23 (Bunch grasslands with grey-box, one record 1981), 24 (Bunch grassland with bloodwood, one record 1981), 26 (Bunch grassland with Eucalyptus terminalis, one record 2000), 27 (Grassland, several records pre-dam), 29 (Saltwater grasslands, one record 2000), 35 (Bunch grasslands with bloodwood and snappy gum, two records), and 36 (Bunch grasslands with bloodwood and snappy gum, two records). All sites contained or were within a catchment which there are PCFW records, though some have confirmed

				extirpations. We assume an extirpated catchment could be restored to support the species again (van Doorn et al., 2015).					
	Connectivity with Ramsar wetlands	Euclidean distance of restoration site boundary to nearest Ramsar listed wetland (DAWE, 2020b).	Euclidean distance of restoration site boundary to nearest Ramsar listed wetland (DAWE, 2020b).	Euclidean distance to of restoration site boundary to nearest Ramsar listed wetland (DAWE, 2020b).	A distance scaled to 0-100 with 100 adjacent to a Ramsar site	0.0625	0.175	0.0625	0.175
	Connectivity with existing wetlands	Euclidean distance to nearest wetland classified as terrestrial, estuarine or marine using Queensland wetlands data (DES, 2020c).	Euclidean distance to nearest wetland mapped within the National register of important wetlands (Environment Australia, 2010).	Euclidean distance to nearest wetland mapped within the National register of important wetlands (Environment Australia, 2010).	A distance scaled to 0-100 with 100 adjacent to an important site	0.0625	0.175	0.0625	0.175
	Patch size (not included as correlated with other indicators)	Area of the restoration site in ha calculated using ArcGIS geometry functions.	Area of the restoration site in ha calculated using ArcGIS geometry functions.	Area of the restoration site in ha calculated using ArcGIS geometry functions.	Area scaled to 0-100 with 100 a site with the largest area	NA	NA	NA	NA
Fisheries	Nursery habitat for fisheries	The lower intertidal zone likely has two high tides daily in the Fitzroy Basin providing nursery habitat. The area of lower intertidal zone within the restoration site, determined using the DEM and the tide levels (MSL-MHWN).	The area of riverine or floodplain within the restoration site from the Western Australian hydrolines dataset applying a 100m buffer (Crossman & Li, unknown) and flood projection model (DWR, 2022). .	The area within the restoration site. Calculated using the area of lower intertidal habitat calculated as for Fitzroy plus the area of riverine habitat calculated the same as for Peel.	Area scaled to 0-100 with 100 a site with the largest area	0.083	0.23	0.125	0.35

	Connectivity to fish habitat	Third order streams and above are likely to contain fish habitat and higher fish populations (DAFF, 2013). Closest Euclidean distance of the boundary of the restoration site to the nearest third order stream and above (Department of Resources, 2021f).	Flow path distance from the restoration site boundary, along any watercourse in the Hydrolines dataset, to an opening to the ocean, and then Euclidean distance to the nearest marine reserve or national park (DAWE, 2020a).	Flow path distance from the restoration site boundary, along any watercouse in the Hydrolines dataset, to an opening to the ocean, and then Euclidean distance to the nearest marine reserve or national park (DAWE, 2020a).	Distance scaled to 0-100 with 100 a site adjoining the fish habitat.	0.083	0.23	0.125	0.35
	Connectivity to declared fish habitat area	Closest Euclidean distance of the boundary of the restoration site to a declared Fish Habitat Area from the Wetlands data (DES, 2020c).	No designated fish habitat areas	No designated fish habitat areas	Distance scaled to 0-100 with 100 a site adjoining the fish habitat.	0.083	0.23	NA	NA
Water quality	Dissolved Inorganic Nitrogen (DIN) concentration	DIN catchment loads divided by catchment flows to give concentrations (Waters et al., 2014).	TN concentrations from catchment nutrient reports (DWR, 2018) were assigned to relevant catchments. Each restoration site was assigned the identity of a catchment if it was connected to it, was within a 100m buffer of the line, or was within an estuary that the line fed. Sites on estuaries with multiple sources of TN were ascribed	Data unavailable	Concentration scaled to 0-100 with 100 a site within the catchment with the highest DIN concentration.	0.0625	0.175	0.125	0.35

		the sum of all sources.						
Total Suspended Solid (TSS) concentration	TSS catchment loads divided by catchment flows to give concentrations (Australian and Queensland Government, 2019).	Data unavailable	Data unavailable	Concentration scaled to 0-100 with 100 a site within the catchment with the lowest TSS concentration.	0.0625	0.175	NA	NA
Hydraulic efficiency	Permanent watercourses are likely to provide high inflows. Hydraulic efficiency was estimated as the area of the restoration site intersecting 3 order streams and above (Department of Resources, 2021f). Calculated from the length of the stream with a 100 m buffer.	NA	No nutrient data available, so not calculated.	Area scaled to 0-100 with 100 a site with the highest area of permanent watercourse.	0.0625	0.175	NA	NA
Estuarine water residence time	The low intertidal zone likely has two high tides daily in the Fitzroy Basin providing estuarine water inundation. The area of lower intertidal zone within the restoration site, determined using the DEM and the tide levels (MSL-MHWN).	Area of fisheries habitat within the restoration site, and then flow path distance of the site boundary along any watercourse in the Hydrolines dataset, to the point of opening of its semi-enclosed estuary. Closed estuaries were assigned an arbitrary value of 1000 then the raw data was logged for analysis.	No nutrient data available, so not calculated.	Area or distance scaled to 0-100 with 100 a site with the highest area of low intertidal zone or time/area.	0.0625	0.175	0.125	0.35

Coastal protection	Indirect protection during everyday conditions – reduce erosion from inland floods	Area of restoration site within the flood zone (100 year flood or 1% AEP) identified from the Fitzroy River Basin Flood Mapping (DNRME, 2015).	Area of pre-European saltmarsh within the restoration site buffered by 2 km that falls within the 100-year flood projection model (DWR, 2022).	No flood projection model available.		0.125	0.35	0.25	0.7
	Direct protection during storm events – wave and erosion attenuation from coastal floods	Area of restoration site likely to have mangroves (BVG 35a) from the pre-clear regional ecosystem mapping (DES, 2019).	Not applicable – these are semi-enclosed estuaries where sites are protected from coast by a dune.	Mangrove area calculated from the total area of mangrove mapped within the WA vegetation type mapping dataset and the NISB.		0.125	0.35	NA	NA
Cultural heritage	Native Title	Area of the restoration site that has Native Title, an Indigenous Land Use Agreement (ILUA) or Future Act Notices identified from the National native title tribunal (National Native Title Tribunal, 2020). Include 'Native title exists' both exclusive and non-exclusive (exclude native title does not exist and native title extinguished). Only non-exclusive The stakeholder meetings revealed that there is no Indigenous owned land in the study region.	Data unavailable as Native Title is currently in resolution (Department of the Premier and Cabinet, 2022). The stakeholder meeting revealed that there is no Indigenous owned land in the study region.	According to the Kimberley Land Council, Native Title in this area has been resolved. However no sites were found on the National Native Title Tribunal register, and no information regarding Indigenous land tenure was available.	NA	NA	NA	NA	NA

Potential Native Title	Area of the site that is lease hold, state or crown land identified from the Qld cadastral data (Department of Resources, 2021a).	Cadastral data unavailable for project.	Cadastral data unavailable for project.	NA	NA	NA
Cultural heritage parties	Area of the site that is within a registered cultural heritage party boundary (Department of Resources, 2022). Relevant parties in Fitzroy Basin include: 1. Barada Barna, Kabalbara and Yetimarla people, 2. Durumbal people, 3. Bailai, Gurang, Gooreng Gooreng, Taribelang Bunda.	Noongar Standard Heritage Agreement Resources and templates - South West Native Title Settlement (Department of the Premier and Cabinet, 2022). Parties in case study region: Whadjuk people, South West Boojarah #2, Gnaala Karla Booja.	Kimberley Land Council Native Title database available via their website.			
Aboriginal sites, objects and ancestral remains	Sites on the QLD cultural heritage database and register (DATSIP, 2021). No sites “declared an aboriginal site under the relics act” in the study region.	Area within the restoration site that is mapped as Aboriginal Heritage Places (DPLH, 2022).		NA	NA	NA

Appendix B Supplementary Results

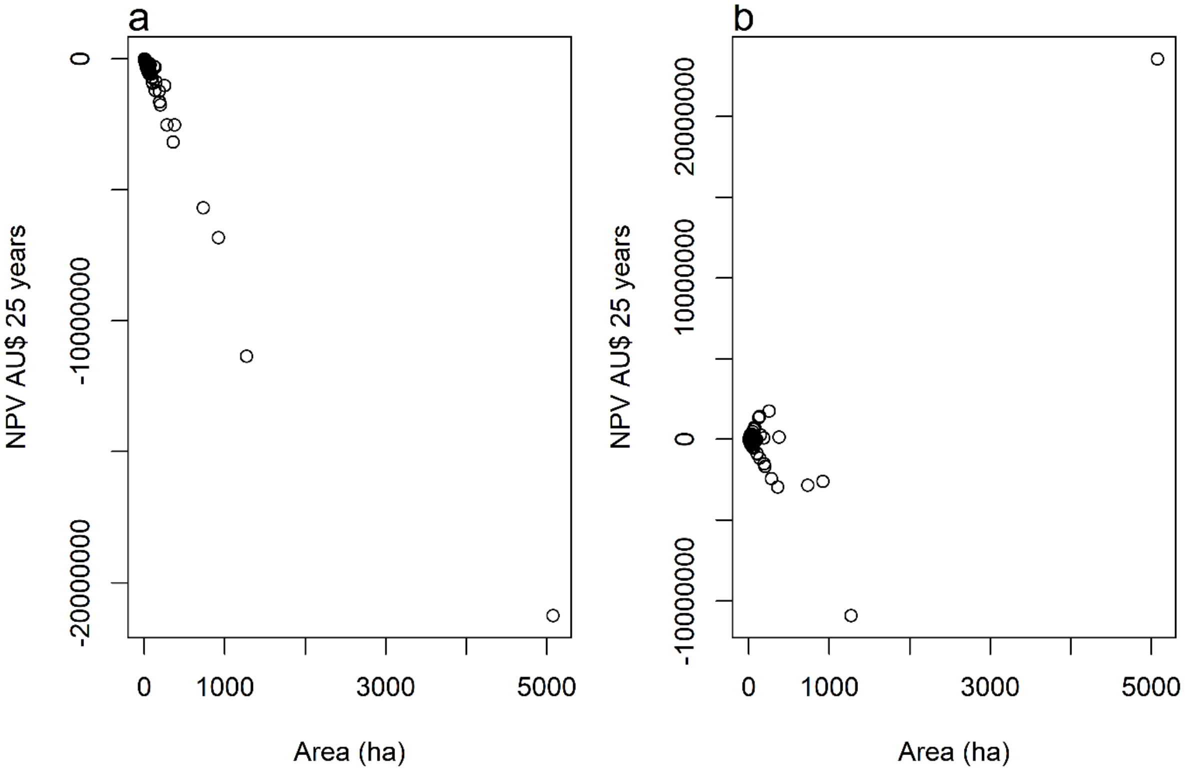

Figure B1 Fitzroy Basin case study. Net present value (NPV) plotted against size of restoration site for scenarios based on net present value at 25 years, 1% discount rate, and lower restoration cost, with (a) current carbon price (S1) or (b) higher carbon price (S3).

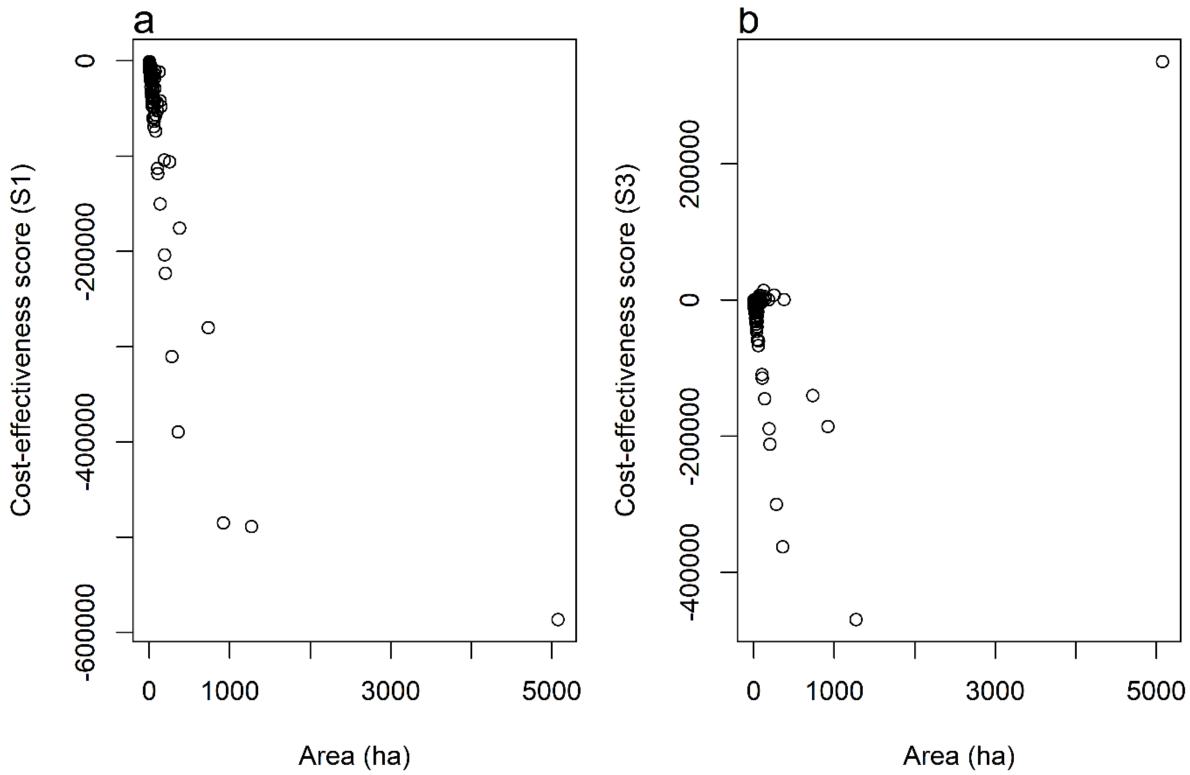

Figure B2 Fitzroy Basin case study. Cost-effectiveness (CE) score plotted against size of restoration site for scenarios based on net present value at 25 years, 1% discount rate, and lower restoration cost, with (a) current carbon price (S1) or (b) higher carbon price (S3)

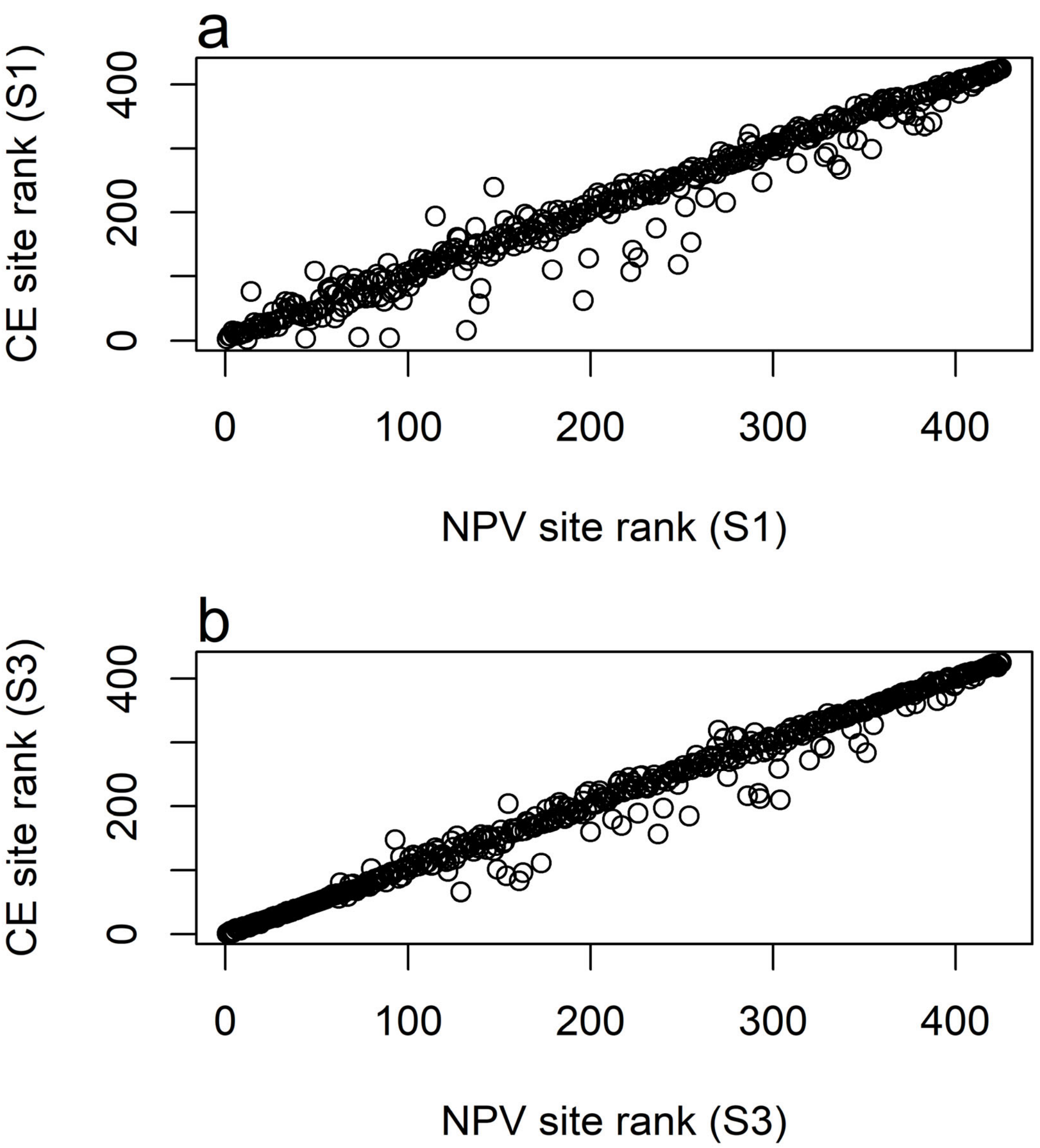

Figure B3 Fitzroy Basin case study. Relationship between site rankings for the net present value (NPV) prioritisation and the cost-effectiveness (CE) prioritisation for (a) scenario 1 (25 years, 1% discount rate, lower restoration cost [$7,174 per ha] and current carbon price [AU$16]), and (b) scenario 3 (25 years, 1% discount rate, lower restoration cost and higher carbon price [AU$40]) Plotting the site rankings of the base scenario (S1) NPV and CE were strongly positively correlated.

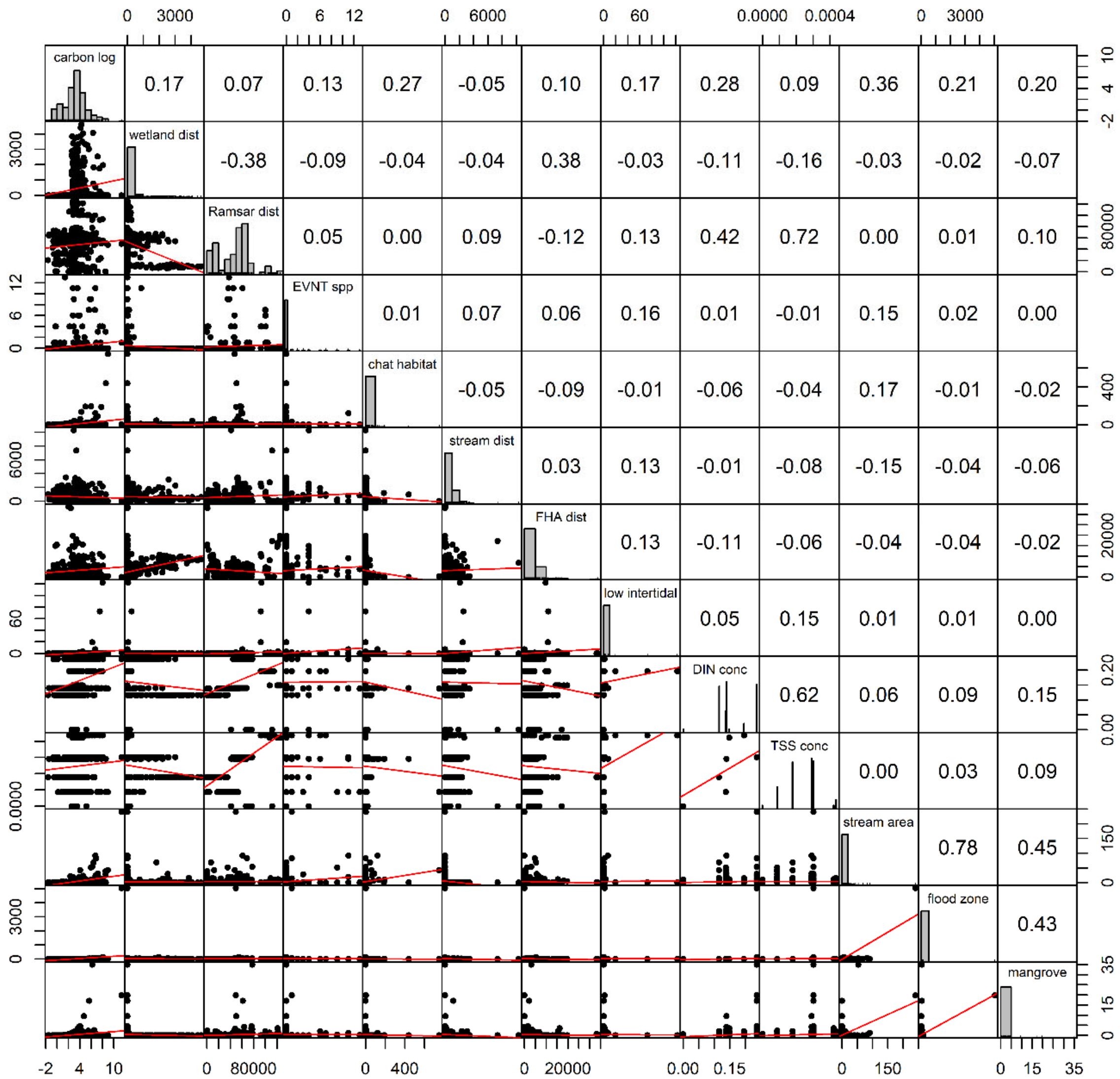

Figure B4 Pairs plots of co-benefit indicator measures and mean annual carbon abatement per Fitzroy Basin restoration site

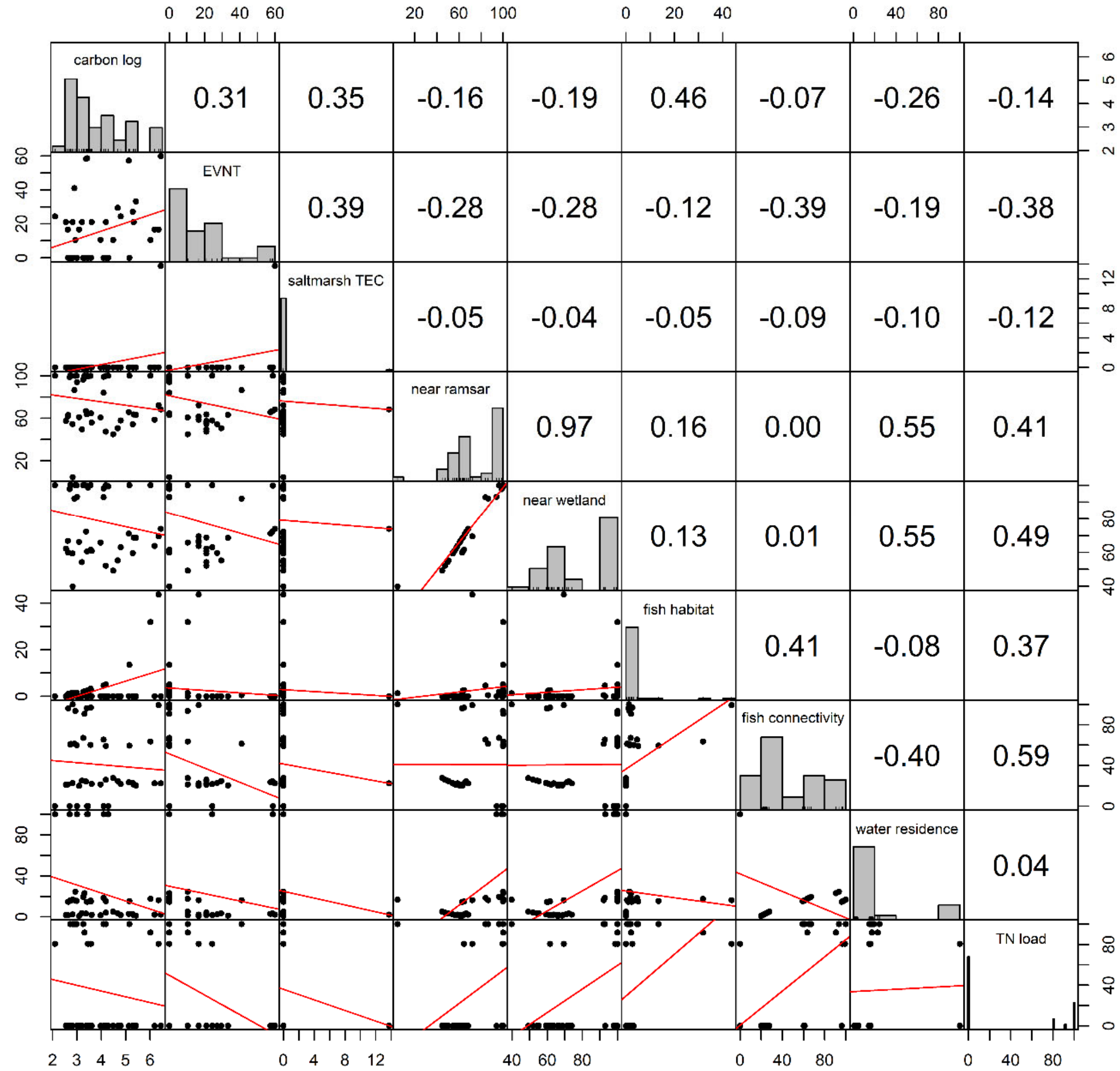

Figure B5 Pairs plots of co-benefit indicator measures and mean annual carbon abatement per restoration site for Peel Harvey

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