



Opportunities for blue carbon restoration projects in degraded agricultural land of the coastal zone in Queensland, Australia

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Abstract

Restoring degraded agricultural lands to their original coastal wetland cover is an approach for enhancing blue carbon storage. This approach enhances carbon sequestration in biomass and soils whilst reducing greenhouse gas emissions and delivering other conservation benefits such as enhancing biodiversity, improving water quality, and protecting coastlines from sea level rise. In Queensland, Australia, tens of thousands of hectares of coastal land have been converted to agriculture since the 1900s, often through drainage. We evaluated the characteristics of degraded agricultural land to identify opportunities for blue carbon restoration projects. Degraded agricultural land was identified through visual inspection of satellite imagery. Our analysis revealed degradation was associated with historical *Melaleuca*-dominated wetland vegetation and current land uses other than intensive agriculture. Field sampling of a subset of paired degraded and non-degraded sites found that water content, organic carbon, and electrical conductivity were significantly higher at degraded sites. We also observed standing water, drainage structures, and dead trees at degraded sites. From our analyses, we inferred land degradation is likely caused by waterlogging, salinisation, and land management choices. Degraded land historically vegetated by *Melaleuca*-dominated wetlands could be targeted for blue carbon restoration projects that protect remaining soil carbon and enhance carbon storage, restore ecosystem services, and provide new income streams for landowners. Further characterisation of the distribution of degraded lands may contribute to prioritisation of sites suitable for restoration.

Keywords Blue carbon · Coastal wetland · *Melaleuca* · Restoration · Land degradation · Coastal agriculture

Introduction

Coastal wetlands, which are called blue carbon ecosystems, are important for climate change mitigation and adaptation on a national scale (Taillardat et al. 2018) because they store and sequester high amounts of organic carbon in their biomass and accumulated sediments (Lovelock and Duarte 2019; Serrano et al. 2019). Coastal wetland ecosystems include seagrass meadows, saltmarshes, mangrove forests, and brackish swamp forests dominated by *Melaleuca* and *Casuarina* species (Macreadie et al. 2017; Lovelock and Duarte 2019). These ecosystems also provide other ecosystem services such as coastal protection, biodiversity

enhancement, and the maintenance of water quality (Barbier 2011).

Australia has the largest area of blue carbon ecosystems of any country and thus one of the largest blue carbon storage capacities (Serrano et al. 2019). Of all states, Queensland has among the highest biomass and soil carbon stocks per hectare (Macreadie et al. 2017; Serrano et al. 2019). However, large areas of Queensland's coastal wetlands have been cleared and degraded through conversion to agriculture (Creighton et al. 2015; Hossain et al. 2020), which has led to past CO₂ emissions (Serrano et al. 2019; Kelleway et al. 2020). These converted lands may provide an opportunity for restoration of these ecosystems and enhancement of carbon sequestration (Duarte de Paula Costa et al. 2022). However, identifying and characterising agricultural land degradation is a key step in understanding the scale and scope of this opportunity.

Land degradation involves a reduction or loss of soil quality due to chemical alteration and/or physical degradation of the soil structure that inevitably reduces biological

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and/or economic productivity (Stewart 2015; Webb et al. 2017). Land degradation in coastal agricultural areas can be caused by (1) seawater intrusion and the salinisation of soil resources (Butcher et al. 2016; White and Kaplan 2017; Tully et al. 2019); (2) inadequate drainage of irrigated fields and associated rising groundwater causing secondary salinity (Elgabaly 1977; Szabolcs 1989; Oliveras-Berrocales et al. 2017); (3) coastal storms that destroy crops and result in tidal surges that cause long-term damage to fields (Tully et al. 2019; Gould et al. 2020; Hasan and Kumar 2020) and erosion (Webb et al. 2017; Van Pelt et al. 2017; Hos-sain et al. 2020); (4) changes in the chemical properties of soils associated with poorly planned or maintained coastal engineering infrastructure, especially barrages, bund walls, and culverts (Wolanski and Cassagne 2000; Khanom 2016; Central Water Commission 2017); and (5) the acidification of soils in response to land-use changes or poor land management (Fanning and Burch 2000; Fanning et al. 2017). In particular, soil acidification, erosion, and salinisation are major sources of agricultural land degradation in Australia (Gretton and Salma 1996). In Queensland, acidified soils are common and a major cause of agricultural degradation because of extensive drainage for agriculture (Sammut et al. 1996; White et al. 1997; Powell and Martens 2005); erosion is a cause of degradation in the Murray-Darling basin (Gretton and Salma 1996); and saline soils with high water content are known to co-occur with historical storm damage (Mushtaq et al. 2018) or hydrological modifications that alter flooding regimes (e.g. bunds, drains, or tidal gates) (Abbott et al. 2020). These key threats to agricultural soil quality are likely exacerbated by climate change, including rising sea levels, rising atmospheric temperatures, increased intensity of storm events, deterioration of coastal management infrastructure, and urbanisation (Beresford 2004; Olsson et al. 2019).

When agricultural fields become degraded, landholders can ameliorate or abandon them. Due to high costs associated with amelioration, many landholders opt to abandon agricultural practices on degraded fields (Gates and Grismer 1989; Ghassemi et al. 1995; Hussain 2005; Abdalla et al. 2015). These degraded agricultural fields may be useful for blue carbon restoration as coastal wetland plants can grow in saline soils and in saturated soils, although tidal flows are also important (Lewis et al. 2016). However, few studies have considered restoring degraded agricultural land to blue carbon ecosystems (Aye and Takeda 2020; Iram et al. 2022). Coastal wetland restoration has been identified as an effective climate change mitigation strategy that can be used to avoid, reduce, or reverse land degradation (Olsson et al. 2019; Kelleway et al. 2020). Therefore, degraded agricultural areas in coastal zones, particularly in Queensland, have the potential for coastal wetland restoration and enhancing blue carbon storage without trade-offs to agricultural

production (Gates and Grismer 1989; White and Kaplan 2017; Tully et al. 2019; Hagger et al. 2022).

The restoration of degraded farmland can easily be incorporated into existing farming systems and can enhance overall productivity. For example, a study in Myanmar found that successful colonisation and growth of planted mangroves on abandoned rice paddies benefited adjacent agricultural areas with increased productivity through improved soil conditions and weed suppression (Aye and Takeda 2020). An additional benefit for landholders from restoring degraded farmland to blue carbon ecosystems is the potential income source and economic benefits associated with voluntary carbon markets, and in Australia using the blue carbon method of the Emission Reduction Fund (Runting et al. 2017; Macreadie et al. 2017; Kelleway et al. 2020; Adame et al. 2020; Hagger et al. 2022; Lovelock et al. 2022). For this reason, characterising the distribution of degraded agricultural lands in coastal Queensland is important for identifying restoration opportunities.

To explore the potential of degraded coastal agricultural land in Queensland for coastal wetland restoration, this study aimed to (1) identify agricultural land degradation in coastal Queensland and discover the factors influencing degradation (e.g. soil acidification, salinisation, coastal flooding, etc.); and (2) characterise degraded soils based on soil properties and land management practices. Our goal was to provide information to elaborate the opportunities for blue carbon restoration in the degraded agricultural fields of coastal Queensland.

Throughout Queensland, climate varies and includes subtropical regions, tropical humid regions, and drier monsoonal regions with highly seasonal rainfall. We therefore expected regional variation in the occurrence and causes of agricultural land degradation driven by differences in elevation, climate, hydrology, and soil conditions (see Supplementary Information 1). Specifically, land degradation caused by salinity was anticipated to occur in regions with high temperatures, low rainfall, low elevations, clay soils, and hydrological modifications restricting tidal flow. Land degradation from erosion would occur in regions with sandy soils, high rainfall, and historical storm damage. Finally, we anticipated that the distribution of degraded land with acidic soils would be concentrated around anthropogenic drainage structures (Fitzpatrick et al. 2011).

Materials and methods

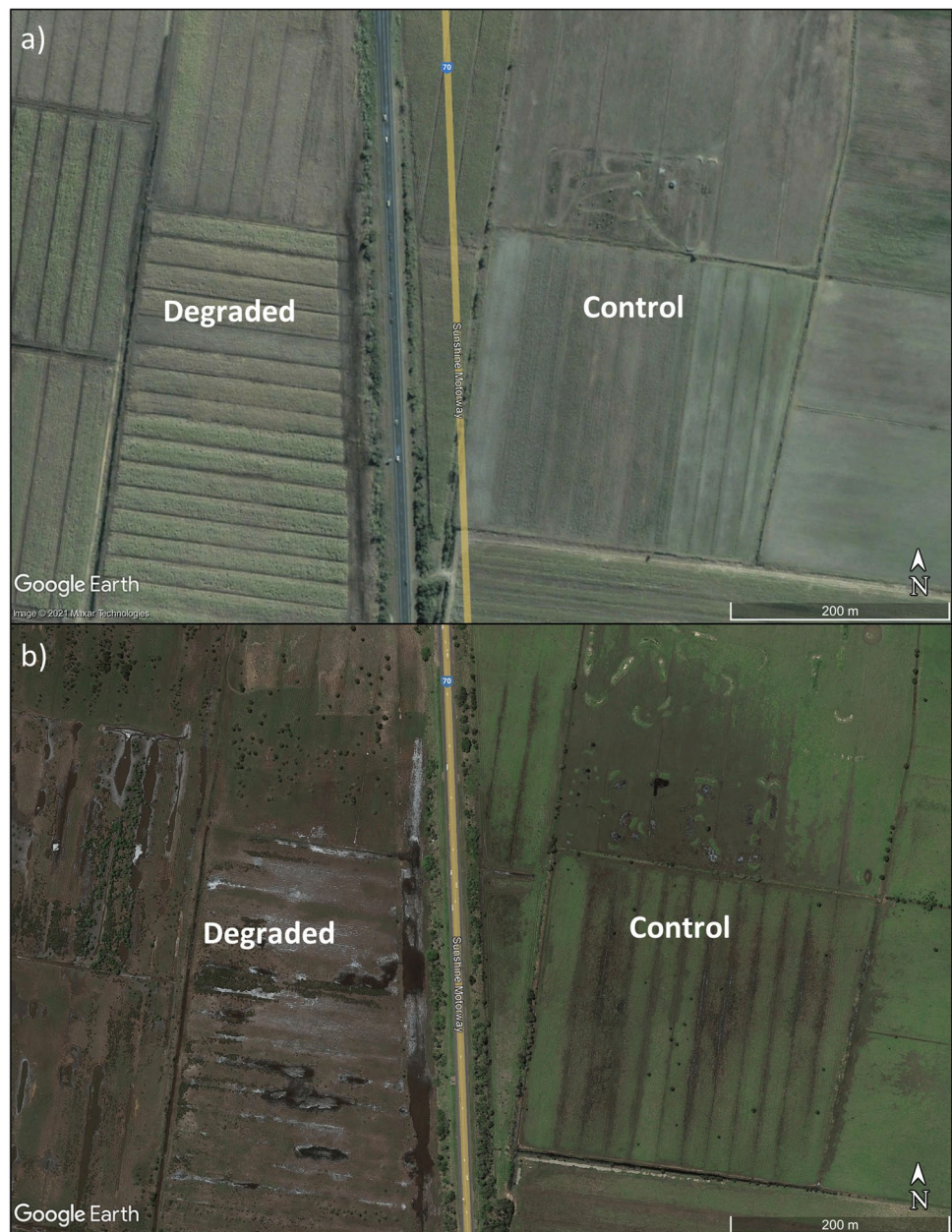
Identifying agricultural land degradation on the Queensland coast

Parcels of degraded coastal land were identified along the Queensland coast (-28.735496306 to -15.798120394°) from a combination of satellite imagery and aerial

photographs at resolutions above 30 cm (Fig. 1). Degraded sites were selected by examining sequential historical satellite images at a landscape or finer scale using Google Earth Pro Version 7.3.6. If the quality or scale of images was inadequate for the detection of degradation at specific sites, historical aerial imagery was assessed (State of Queensland 2021). Degraded sites comprised those that experienced declines in crop cover of at least 20% within 5 years with no visible recovery, or gradual declines in extent between 1991 and 2021. This was determined using a regular grid (100×100 m) spatial overlay. Non-degraded sites, herein referred to as control sites, demonstrated no visible crop cover declines over the same period but had

similar characteristics. Specifically, control sites had similar land zone classifications and elevation and were located within 5 km of the paired degraded sites. Land zone classification (Wilson and Taylor 2012) and elevation are linked to edaphic conditions that influence soil quality (Gbakima 1994; Hashim 2003). We examined sites within the Highest Astronomical Tide (HAT) contour (State of Queensland 1995). Only sites that were previously coastal wetlands were included in analyses, determined using pre-clear regional ecosystems mapping (Queensland Herbarium 2013; Eyre et al. 2015). Our definition of coastal wetlands was consistent with “Wetland Broad Vegetation Groups” listed by Neldner et al. (2019). Some pairs of supratidal sites were

Fig. 1 An example of agricultural land degradation in South-east Queensland. Panel **a** is of sugarcane fields in June 2004, and panel **b** shows degraded cane fields in October 2017. Locations that are considered “control” fields where sugarcane is still in production and degraded fields are indicated in the image. Images downloaded from Google Earth Pro at 1024×768 resolution



included if they were historically brackish freshwater wetlands. These were located within a transition zone between tidally inundated areas and areas under freshwater influence, no more than 2 km landwards of the HAT (Queensland Herbarium & Environmental Protection Agency 2005). Sites within this zone may have experienced subsidence from agricultural tillage and compaction and therefore may be tidally influenced in the future (White and Kaplan 2017). Through this process, we identified 94 degraded agricultural fields with paired control sites. These were distributed throughout 25 catchments (Fig. 2) over four biogeographic regions, including Southeast Queensland, the Brigalow Belt, the central Queensland coast, and the Wet Tropics. These regions cover the latitudinal extent of coastal agriculture in Queensland. Data on topographic and edaphic properties that are likely to affect agricultural land degradation were collected from national and state datasets (Table 1). Values for both degraded and control sites were extracted from relevant datasets in ArcGIS 10.7.1.

Characterising degraded soils in relation to soil and land management properties

Paired field sites from each of the four study regions were randomly selected for field visits where we obtained permission from landholders to conduct field sampling. A classical power calculation revealed 16 independent replicates were necessary for 80% experimental power (McCrum-Gardner 2010). We used 20 replicates, compensating for a potentially higher standard deviation or smaller effect size than expected. This equated to five paired sites within each of the four bioregions or a total of 40 sites (Fig. 2).

Stratified random sampling was used to collect six topsoil (0–30 cm) samples with a small “push” corer (cylindrical corer 5.2 cm diameter and 40 cm length) from each site. Specifically, the GPS locations of three 30 m transects per site were chosen at random. Two soil samples of known volume and with an approximate mass of 50 g were collected from random distances along each transect, culminating in

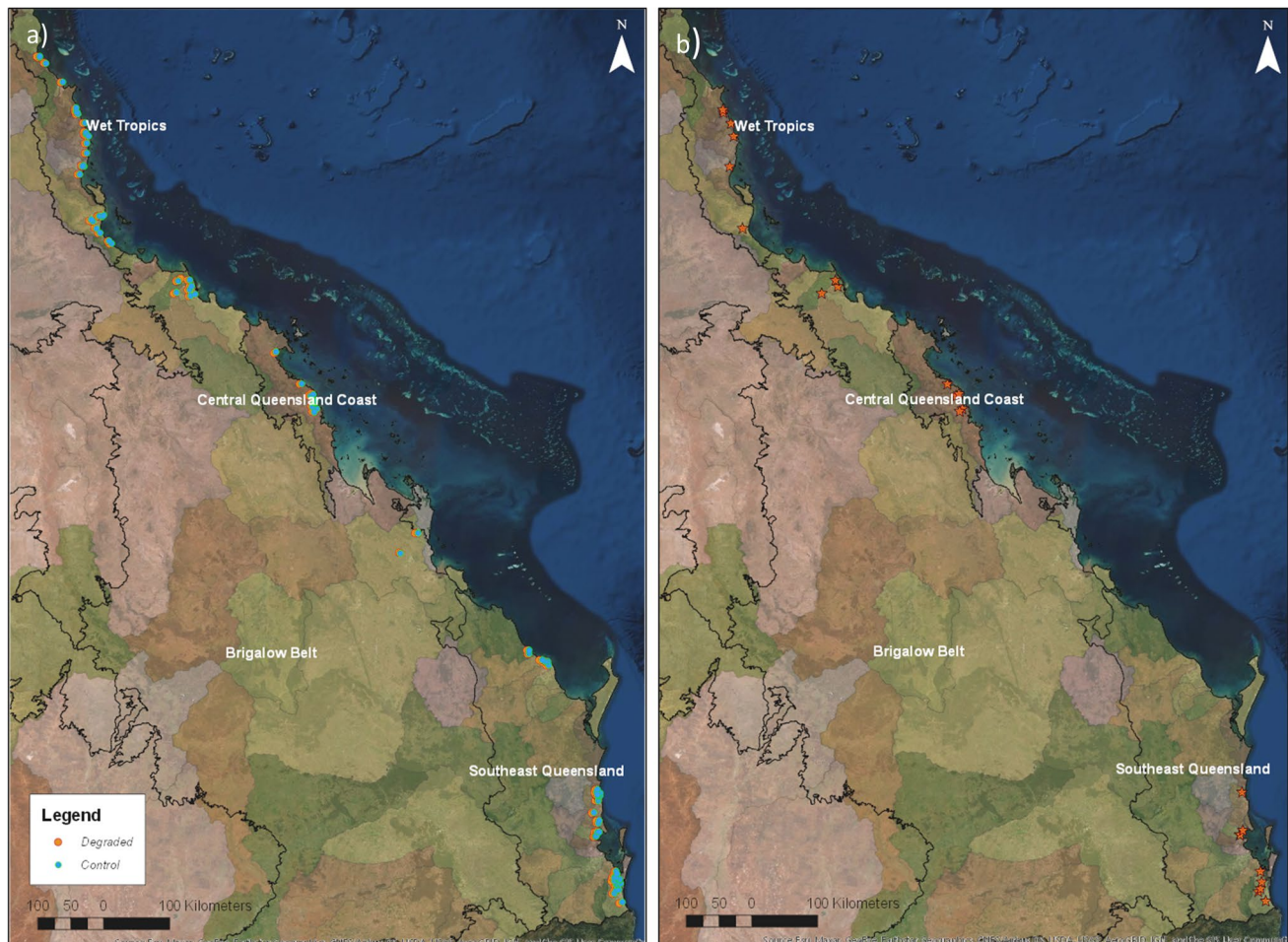


Fig. 2 The distribution of 94 degraded (orange) sites with adjacent control sites (blue) used to assess the characteristics of degraded and control land (a); and the distribution of the 20 paired sites used for

field sampling (b). In both panels, catchments are differentiated in a transparent colour layer with biogeographic regions outlined in black and labelled in white. This figure was created using ArcGIS 10.7.1

Table 1 Climatic, topographic, hydrological, and edaphic variables extracted from spatial datasets to assess differences among degraded and undegraded (control) sites identified in this study. The method of extraction and rationale for extraction are provided

Type	Variable/s	Dataset name	Data	Description	Source	Method of extraction	Rationale
Topographic	Land use	Land use mapping—1999 to Current—Queensland	Categorical	Digital map of the most recent land uses of QLD according to the Australian Land Use and Management Classification	State of Queensland (2019a)	“Spatial Join” tool, ArcGIS Desktop 10.7.1	To test whether land degradation can be linked to specific land uses including cropping and grazing land
	Elevation	30 arc-second DEM of Australia	Continuous	Elevation data of sites in Australia extracted from the EROS GTOPO30 digital elevation model	EROS (1996)	“Extract Multi Values to Points” tool, ArcGIS Desktop 10.7.1	To test the association of land degradation with low-elevation land
	1. Permanent hydrology feature 2. Proximity to feature	Surface Hydrology Polygons (Regional)	1. Categorical 2. Continuous	Contains geometric representations of hydrographic landscape features (natural and artificial)	Crossman and Li (2015)	“Proximity” tool, ArcGIS Desktop 10.7.1	To test an association between land degradation and drainage, indicated by hydrographic landmarks
	1. Estuary classification Proximity to estuary	OzEstuaries Online GIS (now Ozcoasts)	Categorical	Geomorphic habitat map of Australian estuaries	Beard et al. (2003)	“Spatial Join” tool, ArcGIS Desktop 10.7.1	To test whether degraded sites were associated with estuaries
	Pre-clearing Broad Vegetation Group (BVG)	Queensland Regional Ecosystems	Categorical	Map of pre-clearing regional ecosystems based on vegetation, soil, and geology	Queensland Herbarium (2013)	“Spatial Join” tool, ArcGIS Desktop 10.7.1	Used for site selection and to test for association of degradation with specific BVGs. Only sites historically classed as wetlands were included
	1. Proximity to canal 2. Canal density	Canal areas—Queensland	1. Continuous 2. Continuous	Contains manmade open channels designed for drainage of irrigated land, coastal management, or canal estates	State of Queensland (2019b)	“Proximity” and “Line Density” tools, ArcGIS Desktop 10.7.1	To test an association between degradation and drainage canals, which indicate management responses to inundation
	Local hydrology modifier	Wetland areas	Categorical	Provides information on activities being conducted for hydrological modification of wetlands (‘Extraction’ or ‘Addition’)	State of Queensland (2020)	“Spatial Join” tool, ArcGIS Desktop 10.7.1	To test whether degradation cooccurs with hydrological modifications that restrict tidal flow

Table 1 (continued)

Type	Variable/s	Dataset name	Data	Description	Source	Method of extraction	Rationale
Edaphic	Soil type (Categorised)	Digital Atlas of Australian Soils	Categorical	Provides a description of Australian soils according to soil type, which are defined by specific soil characteristics	Bureau of Rural Sciences (2009)	“Spatial Join” tool, ArcGIS Desktop 10.7.1	To test whether the proportions of clay, sand, and silt influence degradation
	Soil carbon	Baseline map of Australian soil organic carbon stocks and their uncertainty. v2	Continuous	Contains percentage values for soil organic carbon aggregated from SCaRP, NGSa and ASRIS databases	Viscarra Rossel et al. (2014)	“Extract Multi Values to Points” tool, ArcGIS Desktop 10.7.1	To test an association between low soil organic carbon and degradation

a total of 240 samples. Soil samples were sealed, labelled, and stored in a cool, dry place prior to laboratory analysis. Soil properties assessed included bulk density, soil texture, water content, pH, electrical conductivity, and organic carbon (Table 2; methods in Supplementary Information 2).

Factors known to indicate the type and severity of observed degradation include land use, vegetation structure, community composition, relief (slope), aspect, ponding or flooding, the presence of drainage structures, and the occurrence of dead trees (Fauzie 2016; Kirwan and Gedan 2019; Omuto et al. 2020). These variables were assessed and recorded in the field (Table 2). Anecdotal evidence of land degradation and land management was also collected from discussions with landowners where possible.

Statistical analysis

All statistical analyses were conducted using R Version 4.1.1. A binomial generalised linear model (GLM) was constructed to identify the main factors associated with degraded land identified from satellite imagery. The response variable was binary (0 = control and 1 = degraded), and nested within site to account for the paired data structure. Topographic and edaphic variables that differed within and between paired sites (Table 1), together with their interactions, were included as fixed effects in the model. These included soil type, soil carbon, land use, elevation, permanent hydrology features, proximity to hydrology features, pre-clearing Broad Vegetation Groups (BVGs), proximity to canals, canal density, proximity to estuaries, and local hydrology modifiers (for example bund walls and other obvious mechanical disturbances). The variables included in the

analyses were assessed for collinearity, but none were significantly correlated ($r > 0.5$, Figure S1).

Chi-squared tests were used to measure the amount of variation explained by inclusion of specific predictor variables in the GLM. This guided the selective exclusion of predictors to increase parsimoniousness in the model (Chong et al. 2016). Land use (State of Queensland 2019a) was included as a binary variable with categories including (1) intensive, which included irrigated agriculture, dryland agriculture, and intensive horticulture; and (2) low-intensity agricultural land use which was comprised of grazing. Assumptions of linearity and constant variance were validated (Figure S2). A chi-squared Analysis of Variance (ANOVA) test was used to determine which variables were significantly different between degraded and control sites. We used Tukey’s pairwise tests to assess significant differences between pre-clearing BVG classifications (mangroves/saltmarshes, *Melaleuca*-dominated forests, forests, and other).

For the data collected during field sampling, a Principal Components Analysis (PCA) was used to assess variables that distinguish degraded and control sites, and which may indicate land degradation caused by salinisation, erosion, or acidification. The PCA used relief, percentages of clay, silt, sand, and gravel, dry bulk density, water content, electrical conductivity, pH, and soil organic carbon to generate multivariate dimensions that optimally differentiated degraded and control sites (Figure S3). Other variables collected in the field in Table 2 were excluded from the PCA and analysed separately because they were not normally distributed.

The variables hypothesised to indicate land degradation, including pH, water content, soil organic carbon, and

Table 2 Variables measured in the field and laboratory to assess the characteristics of degraded and undegraded (control) sites and the techniques used to measure them

Variable	Field or laboratory	Data	Technique
Land zone	Field	Categorical	Site-specific assessment to verify spatial data
Elevation (m)	Field	Continuous	Measured for each transect using a GPS, and verified with the 'Altimeter' application on an iPhone5
Dominant crop	Field	Categorical	Any crops that were recently cultivated at each field site were recorded
Vegetation structure	Field	Categorical	Vegetation structure on and adjacent to sites was assessed using Specht's ecosystem classification (Specht 1970)
Community composition	Field	Continuous	3 × 30 m transects were laid out perpendicular to any visible agricultural furrows at each site. Every 5 m along each transect, a 1 × 1 m quadrat was used to assess the percentage cover of plant categories adapted from the QLD BioCondition Assessment Manual (Eyre et al. 2015). These include native perennial grass cover, herbs, shrubs, and leaf litter
Relief (°)	Field	Continuous	Measured using a clinometer
Aspect	Field	Categorical	Determined using a compass
Inundation	Field	Binary	Occurrence on a binary scale (0/1)
Drainage structures	Field	Categorical	Drainage structures visible at field sites were categorised (bridge culverts, culverts, swales, or none)
Drain location	Field	Categorical	The location of drainage structures visible at field sites was categorised (in canal, under road, or near stream)
Condition of canal	Field	Categorical	An indication of the appearance of artificial or artificially diverted natural canals near a study site (no problem evident, sewage litter, rubbish, eroded, or eutrophication)
Dead tree species	Field	Categorical	Any dead trees observed were identified to the genus level (e.g. <i>Avicennia</i> , <i>Melaleuca</i> , <i>Acacia</i>)
Tidal restrictions	Field	Binary	Farmers were asked if any management actions had been taken to restrict tidal flow onto their farms (i.e. floodgates, weirs, tidal gates, barrages, or strategic planting). Occurrence was recorded on a binary scale (0/1)
Proportion clay (%)	Laboratory	Continuous	Particle Size Distribution Analysis (PSDA) using Mastersizer 2000
Proportion sand (%)	Laboratory	Continuous	Particle Size Distribution Analysis (PSDA) using Mastersizer 2000
Proportion silt (%)	Laboratory	Continuous	Particle Size Distribution Analysis (PSDA) using Mastersizer 2000
Proportion gravel (> 2 mm) (%)	Laboratory	Continuous	Soil samples were sifted using a 2 mm sieve. Any particles over 2 mm were retained and weighed
Dry bulk density	Laboratory	Continuous	Dry bulk density (g/cm^3) = Dry soil weight (g) / Soil volume (cm^3)
Water content	Laboratory	Continuous	Oven-dry technique (Gardner 1986)
pH	Laboratory	Continuous	TPS WP-81 probe measurement from 1:5 soil suspension
Electrical conductivity	Laboratory	Continuous	TPS WP-81 probe measurement from 1:5 soil suspension
Soil organic carbon	Laboratory	Continuous	Estimate of soil organic carbon based on Loss on Ignition (LOI) to assess organic matter (Hoogsteen et al. 2015) and conversion of organic matter to organic carbon using a conversion factor of 0.52 (Ouyang and Lee 2020)

electrical conductivity, were a response variables in a series of linear mixed-effects models using degradation and climatic region as predictors. Samples nested within transects nested within sites were included as location-specific random effects in the model (Ritz et al. 2015). Resultant *p* values were adjusted using the Bonferroni correction to avoid multiple comparison problems.

Chi-squared tests were used to assess differences in the frequency of degraded and control sites among categorical variables collected in the field. These include vegetation structure, dominant crop, aspect, drainage structures, drain

location, drain appearance, condition of canals, the occurrence of dead trees, and dead tree species.

Variation in vegetation community composition between degraded and control sites was assessed using a GLM with the following predictors: average percent cover of native perennial grasses, herbs, shrubs, organic litter, invasive plants, crop plants, and dead vegetation. No collinearity was detected between predictors (Figure S4). Location-specific random effects for transect nested within sites were also included.

Results

Identifying agricultural land degradation on the Queensland coast

The pre-clearing vegetation of degraded sites were significantly different than those of control sites ($p < 0.05$, $df = 12$). Specifically, there was a high frequency of historic mangrove forests among degraded sites compared to other pre-clearing vegetation categories ($p < 0.005$, $df = 3$). Eighteen percent of degraded sites identified in this study were historically mangrove forests, compared to 5% of control sites. However, almost half (47%) were historically *Melaleuca*-dominated forests and woodlands. A similarly high proportion of control sites (51%) were once *Melaleuca*-dominated. Our analysis also found that land uses categorised as intensive agriculture were significantly more common at control sites than degraded sites ($z = 2.458$, $p < 0.05$). Soil type, mapped soil carbon, land elevation, permanent hydrology features, proximity to hydrology features, proximity to canals, canal

density, proximity to estuaries, and local hydrology modifiers were not different between degraded and control sites (Table S1).

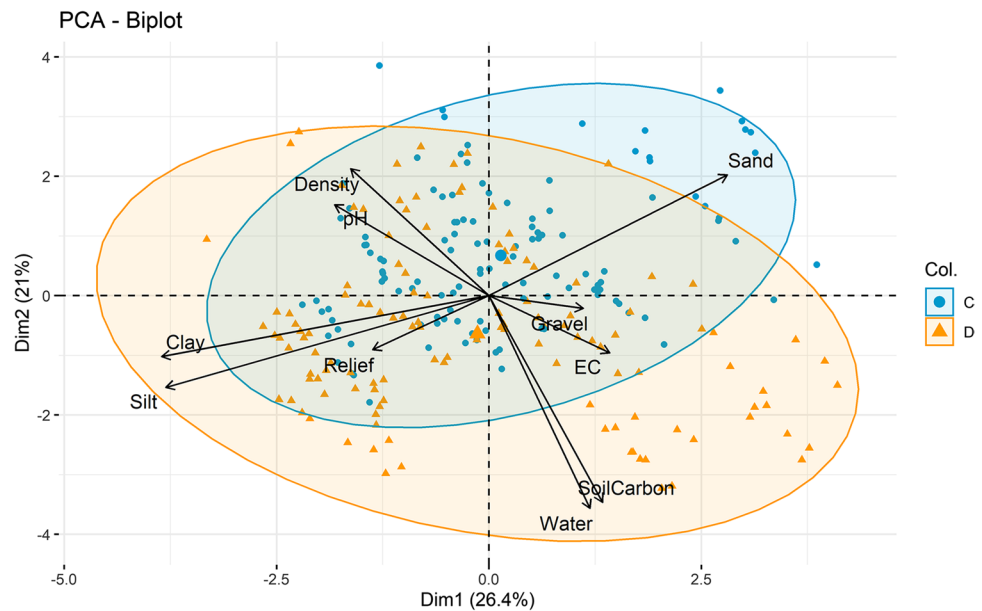
Characterising degraded soils in relation to soil and land management properties

Degraded and control sites differed significantly in soil and land properties (Table 3; Fig. 3). The PCA revealed the first two eigenvectors explained 26.4 and 21.0% of variation, respectively. Degraded sites were associated with high soil water content ($p < 0.0001$, $df = 60$) (Fig. 4; Table 3), which was positively correlated with soil organic carbon ($r = 0.888$, $p < 0.001$) (Figure S5). Degraded sites had significantly higher electrical conductivity values compared to control sites ($p < 0.05$, $df = 60$) (Fig. 4; Table 3). Soil texture was also significantly different between degraded and control sites (Fig. 4; Table 3). Degraded soils were associated with significantly lower sand content ($p < 0.0001$, $df = 60$), higher silt content ($p < 0.0001$, $df = 60$), and higher clay content ($p < 0.0001$, $df = 60$), although

Table 3 Average and standard error values calculated for land and soil characteristics of control and degraded sites per bioregion, which were included in the Principal Components Analysis

Variable	Southeast Queensland		Brigalow Belt		Central Queensland Coast		Wet Tropics		All	
	Control	Degraded	Control	Degraded	Control	Degraded	Control	Degraded	Control	Degraded
Water content (%)	20.1 ± 1.4	32.8 ± 1.8	5.9 ± 0.5	9.6 ± 0.8	13.4 ± 0.7	20.3 ± 1.8	16.2 ± 1.3	26.8 ± 2.4	15.8 ± 0.7	25.9 ± 1.2
Soil organic carbon (%)	4.6 ± 0.6	5.8 ± 0.7	5.8 ± 0.2	9.7 ± 1.0	8.8 ± 0.6	15.5 ± 1.3	5.7 ± 0.5	13.1 ± 2.0	3.5 ± 0.2	6.5 ± 0.5
EC (µS/cm)	537 ± 78	2,710 ± 500	238 ± 26	269 ± 22	260 ± 24	2,120 ± 469	156 ± 18	315 ± 42	325 ± 32	1,653 ± 241
Bulk density (g/cm ³)	1.2 ± 0.1	1.0 ± 0.0	1.0 ± 0.0	1.0 ± 0.1	1.1 ± 0.1	1.3 ± 0.1	1.4 ± 0.0	1.0 ± 0.0	1.2 ± 0.0	1.1 ± 0.0
pH	5.6 ± 0.1	5.4 ± 0.1	6.9 ± 0.1	6.2 ± 0.1	5.0 ± 0.1	6.2 ± 0.2	5.4 ± 0.1	5.3 ± 0.1	5.5 ± 0.1	5.6 ± 0.1
Proportion clay (%)	0.4 ± 0.04	0.3 ± 0.02	0.4 ± 0.03	0.5 ± 0.08	0.4 ± 0.03	1.0 ± 0.06	0.5 ± 0.05	0.6 ± 0.06	0.4 ± 0.02	0.6 ± 0.03
Proportion silt (%)	26.7 ± 1.42	28.4 ± 1.00	38.4 ± 2.50	43.2 ± 2.75	32.5 ± 2.01	46.9 ± 1.53	40.4 ± 2.78	44.7 ± 2.98	33.4 ± 1.23	38.9 ± 1.26
Proportion sand (%)	60.1 ± 2.04	59.9 ± 0.87	61.2 ± 2.54	56.3 ± 2.82	59.2 ± 1.22	41.5 ± 2.49	53.9 ± 2.75	49.6 ± 2.74	58.1 ± 1.18	52.2 ± 1.24
Proportion > 2 mm (%)	12.8 ± 2.39	11.4 ± 1.59	0.0 ± 0.00	0.0 ± 0.00	8.0 ± 2.10	10.6 ± 1.81	5.2 ± 0.84	5.1 ± 0.94	8.0 ± 1.08	8.3 ± 0.85
Relief (°)	-0.5 ± 0.3	0.3 ± 0.4	0.0 ± 0.0	0.0 ± 0.0	-0.6 ± 0.2	0.0 ± 0.0	-0.4 ± 0.1	0.0 ± 0.0	-0.4 ± 0.1	0.1 ± 0.2

Fig. 3 A biplot using the multivariate axes from a Principal Components Analysis that optimally differentiate degraded agricultural land and control, undegraded land. The blue circles are the control (C) sites, and the orange triangles are degraded (D) sites. Arrows show the strength and direction of leverage contributed by specific variables, labelled in black. Only variables with significant leverage were included in the plot. Ellipses show clustering of degraded (orange) and control (blue) sites. Degraded sites gravitated towards the left side of the biplot, whereas control sites were clustered on the right. 'EC' refers to Electrical Conductivity



the percentage of gravel (particles > 2 mm) was not significantly different between degraded and control soils.

Soil pH was similar between degraded and control sites ($p = 0.508$, $df = 60$) (Fig. 4; Table 3). However, values were generally low (acidic), with a mean of $5.77 (\pm 0.0533)$ across all samples. The difference in dry bulk density between degraded and control sites was also non-significant ($p = 1.0$, $df = 60$) (Fig. 4; Table 3).

Degraded sites had more standing water than control sites ($p < 0.0001$, $df = 60$). The frequency of different drainage structures (categorised as bridge culverts, culverts, swales, or none) differed significantly between degraded and control sites ($p < 0.005$, $df = 3$). Culverts were exclusively found at degraded sites, and it was more common for control sites to have no visible drainage structures.

Vegetation community composition was highly variable between degraded and control sites (Fig. 5). Native perennial grass cover was significantly higher at degraded sites compared to control sites ($z = -2.30$, $p < 0.05$), but there was on average a higher proportion of invasive species present at degraded sites ($z = -0.02$, $p > 0.05$).

Discussion

Characteristics of degraded coastal land

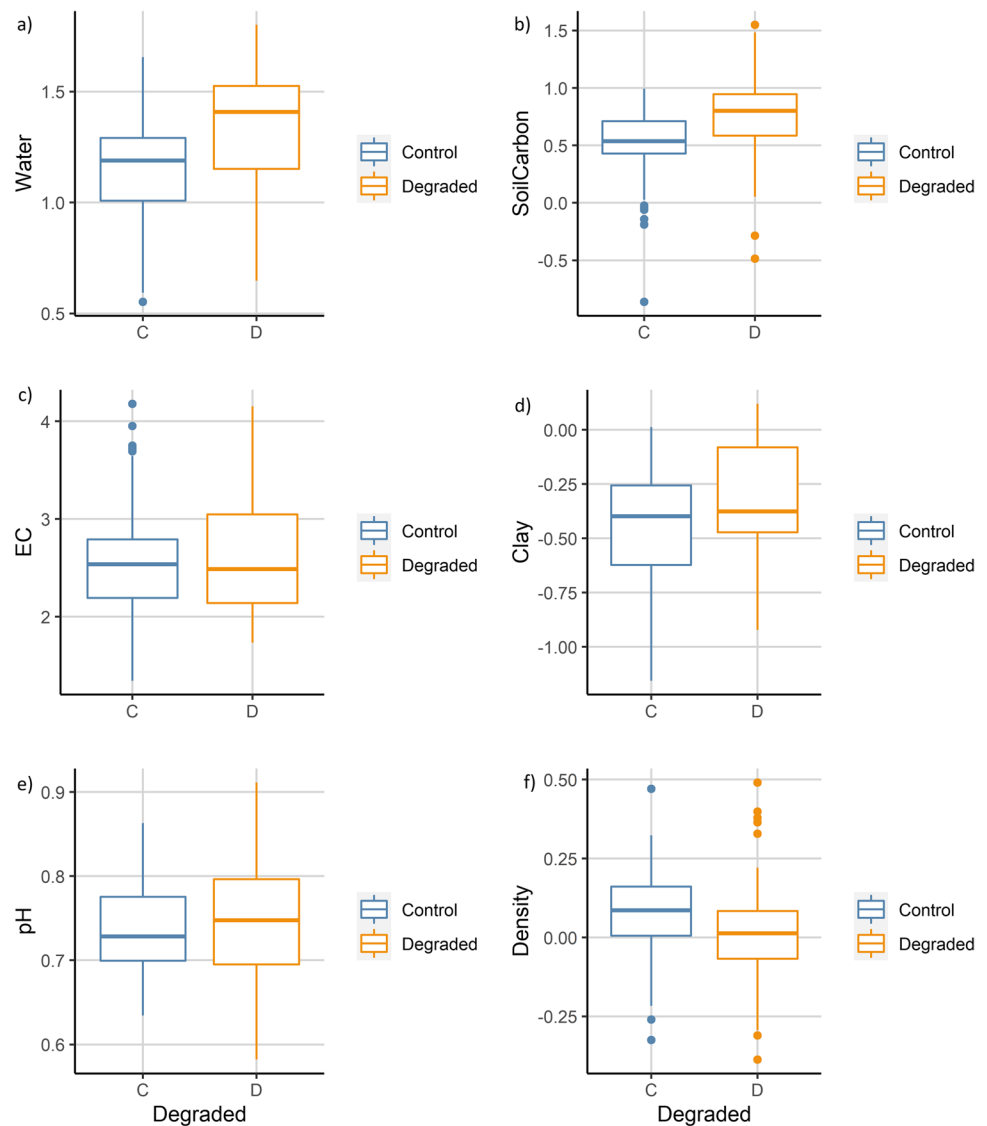
Agricultural land degradation in coastal Queensland was associated with land that was *Melaleuca*-dominated vegetation prior to clearing for agriculture and current land uses that are categorised as intensive agriculture. The physiochemical soil properties of *Melaleuca*-dominated wetlands are not ideal for agricultural production and may lead to land

degradation following grazing or cropping (Wignynosukarto 2013; Sulaiman et al. 2019). *Melaleuca*-dominated wetlands often have seasonally or permanently wet soils, classed as hydrosols or aquic podosols (Bureau of Rural Sciences 2009; Neldner et al. 2019). Ineffective drainage or tidal intrusion into land with these soil types may contribute to coastal agricultural land degradation in Queensland (White and Kaplan 2017; Griggs 2018). Our field assessment confirmed that degraded land was associated with high soil water content, electrical conductivity, and soil textures richer in clay, supporting a role of waterlogging, potentially resulting from ineffective drainage of tidal water, surface water, and rainwater, in agricultural land degradation.

Ineffective drainage of wetland soils, waterlogged soils, and associated agricultural land degradation is potentially widespread in coastal Queensland. For example, 25% of the Herbert River sugarcane area is affected by waterlogging after storm events (Tait 1995; Mitchell 2005). In this region, cane yields have declined by up to 30% since the 1990s, most likely due to drainage failure (Garside et al. 2014). A study in the Central sugarcane area suggested that the impact of rainfall and waterlogging on Mackay sugarcane yields has increased since the 1990s, and that impacts were exacerbated by industry expansion onto marginal (probably waterlogged) soils (Salter and Schroeder 2012). Although the density of drains was not significant in our analysis of the factors that affect the distribution of degraded land, our field assessment found the observed frequency of drainage structures was significantly higher at degraded sites, which were likely erected to manage drainage problems.

Between the 1930s and 1960s, there was widespread drainage and clearing of marine and freshwater wetlands to expand sugarcane production on the Queensland coast

Fig. 4 Boxplots demonstrating the differences in log-transformed measurements of **a** water content (%), **b** soil carbon (%), **c** electrical conductivity ($\mu\text{S}/\text{cm}$), **d** pH, **e** dry bulk density (g/cm^3), and **f** clay content (%) between degraded and control sites. Control sites (C) are depicted in blue and degraded sites (D) are depicted in orange. Dots represent outliers



(Griggs 2018). The clearing and drainage of wetlands has been linked to the formation of acid sulphate soils, where oxidation of hydrogen sulphide when soils are drained results in the production of sulphuric acid (Sammut et al. 1996; Nielsen et al. 2006; Brodersen et al. 2019). For example, cleared and drained sugarcane land led to pH values between two and four in the wet tropical region (Luke et al. 2017). Some soil pH values measured during our field analysis were within this range. However, pH was not significantly different between degraded and control sites and thus land that was previously *Melaleuca*-dominated wetland was likely to be more strongly influenced by high soil water content and inadequate drainage than soil acidity. Our field assessment revealed that soil carbon was higher at degraded sites than control sites. There are multiple potential explanations for this pattern including (1) carbon stocks in coastal wetlands vary and degraded sites may have historically higher soil

carbon stocks (Adame et al. 2020); (2) soil carbon may be protected from loss during agricultural production because of greater aggregate stability associated with higher clay contents (Dominy et al. 2002); or (3) because agricultural land use at degraded sites was less intensive, which perhaps limited declines in soil organic carbon loss caused by agricultural land use. Intensively managed agricultural sites in Europe have been shown to lose an average of $1190 \text{ kg C ha}^{-1}\text{yr}^{-1}$ (Poyda et al. 2019) and soils under sugarcane in Brazil lost 36% of organic carbon over 23 years of cultivation (Franco et al. 2015). Because of their higher soil carbon stocks, degraded sites may be expected to lose soil carbon over time, and thus restoration of pre-clearing wetland vegetation could protect remaining soil carbon stocks and enhance carbon sequestration in both soils and biomass (Lovell and Duarte 2019; Serrano et al. 2019; Duarte de Paula Costa et al. 2022).

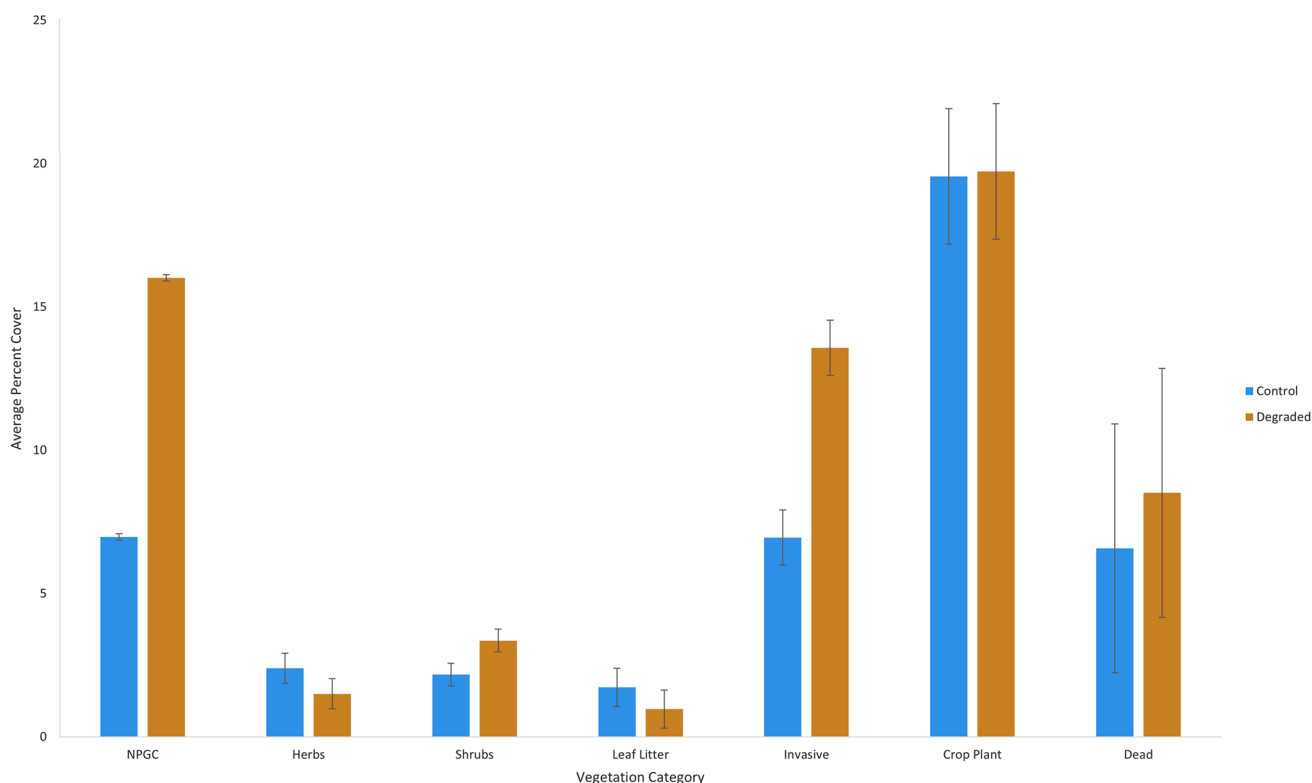


Fig. 5 Clustered bar plot demonstrating the difference in the average percent cover of various vegetation categories between degraded (orange) and control (blue) sites. Error bars represent standard error values. The 'Grass' category refers to native perennial grass cover (NPGC)

Agricultural land degradation on the Queensland coast may also be the result of seawater intrusion, which could be linked to the clearing of wetland vegetation. Functional mangrove forests or marshes have been found to prevent seawater intrusion by attenuating storm surge (Sahu 2015; White and Kaplan 2017). It has been reported that Queensland's irrigated coastal agricultural areas are the most vulnerable to seawater intrusion of all Australian states, especially with projected sea level rise (Bureau of Rural Sciences, unpublished 2008 report). High-risk areas include the Pioneer Valley and Lower Burdekin on the central Queensland coast, and the Burnett basin in Southeast Queensland (Narayan et al. 2007; Liu et al. 2006; Werner and Gallagher 2006). Therefore, seawater intrusion may explain the significantly higher electrical conductivity values measured at degraded sites compared to control sites in these regions. However, this study may also have detected instances of soil salinity caused by rising groundwater (Elgabaly 1977; Szabolcs 1989; Oliveras-Berrocales et al. 2017). Dead trees are known indicators of salinity-affected soils (Kirwan and Gedan 2019), and these were significantly more frequent at degraded compared to control sites. Salinisation of agricultural soils through seawater intrusion or rising groundwater tables may therefore be indicators of opportunity for restoration of coastal wetlands.

Degraded sites were more often found to be used for land uses other than intensive agriculture than control sites. It is possible that degraded lands were first cleared for intensive agriculture (Griggs 2018) and were then abandoned following declines in productivity caused by failure of drainage infrastructure (Hashim 2003; Stewart 2015; Razaq and Wahbi 2018). Degraded sites had a high frequency of agricultural weed species including declared weeds, which landholders are obligated to control. Therefore, one of the benefits of restoration may include control of weeds.

In addition to waterlogging and high salinity, soil compaction can also reduce agricultural productivity (USDA 2001; Jones et al. 2012; Spinelli et al. 2019). Although not measured directly, compaction in floodplain soils can be identified from dry bulk densities over 1.63 g/cm³ (USDA 2001). Approximately 15% of the sites we examined were compacted by this classification, primarily in the central Queensland coast region. However, dry bulk density was not significantly different between degraded and control sites in our study.

Limitations

We used satellite and historical aerial imagery to discover degraded sites on which our analysis was based. Whilst we

have provided indications of the characteristics of degraded agricultural land, in the future machine-learning approaches using remote sensing may offer improvements in the detection of agricultural land affected by waterlogging and salinity (Prajapati et al. 2021). The broader scope of a remote-sensing approach would also permit assessment of regional and management effects on land degradation, which could verify anecdotal evidence of reducing land degradation through improved agricultural practices (e.g. crop rotation). We focused our assessment on soil water content, salinity, pH, organic matter, and particle size, but other soil properties may give further insights to causes of degradation. For example, overuse of fertilisers could also contribute to degradation (Abdalla et al. 2015). Assumptions about reduced soil carbon losses from degraded areas could be verified with future research comparing carbon emissions from intensively managed cropland and degraded or abandoned agricultural areas. Finally, because of the importance of soil water content and salinity in degraded sites, further research could focus on characterising the state of drainage infrastructure at degraded sites.

Opportunities for blue carbon restoration of degraded land

The restoration of degraded agricultural lands to mangrove forests, saltmarshes, and *Melaleuca* wetlands has been recognised as a promising approach to climate change mitigation in Australia (White and Kaplan 2017; Macreadie et al. 2017; Adame et al. 2020; Hagger et al. 2022). This is primarily because of a high capacity for carbon sequestration in the soils and biomass of wetland plants (Macreadie et al. 2017; Adame et al. 2020), with a recent study highlighting the effectiveness of converting abandoned sugarcane fields for reducing greenhouse gas emissions (Iram et al. 2022). However, restoring pre-clearing wetlands on degraded agricultural lands may also prevent coastal erosion, provide critical habitat, and provide a suite of other ecosystem services (Barbier 2011; Hagger et al. 2022). Adopting this strategy in Queensland aligns with several Sustainable Development Goals as well as other national and local targets for biodiversity enhancement, water quality improvements, and climate change mitigation and adaptation (Canning et al. 2021).

In addition to being significant for biodiversity, *Melaleuca* forests, which have been extensively cleared in the region, are globally significant carbon sinks and sequester higher levels of organic carbon than many other types of coastal wetlands (Tran 2015; Adame et al. 2020) a worthwhile target for restoration. There are emerging examples of successful restoration of

Melaleuca wetlands as well as mangroves and saltmarshes in Queensland for their climate change mitigation values (Iram et al. 2022). Other examples of successful restoration of Australian agricultural land to mangrove and saltmarsh habitats indicate that restoration is highly feasible (Howe et al. 2010; Luke et al. 2017; Abbott et al. 2020).

The feasibility of blue carbon restoration projects depends on whether ecosystem benefits from restored vegetation outweigh opportunity costs associated with ceasing an existing land use (Naucmér and Enkvist 2009; Mills et al. 2014; Hagger et al. 2022). For example, a recent study concluded that the conversion of 34km² of sugarcane and grazing land to coastal wetlands in the Wet Tropics region was economically feasible and that 45.34km² would potentially be feasible if Australia's carbon price increased from AUD \$13.85 to \$25/tonne CO₂-e (Hagger et al. 2022). Economic feasibility is likely to have increased with Australian Carbon Credit Unit spot prices exceeding AUD \$50/tonne CO₂-e in December 2021 (Clean Energy Regulator 2022). Many degraded agricultural areas have relatively low opportunity costs because they cannot be used for profitable food production and are in rural or remote areas. Therefore, degraded areas could be prioritised for restoration over productive agricultural areas. Restoration of productive agricultural land is unlikely to be economically feasible due to competing land uses like beef production and cropping (Hagger et al., unpublished 2022 report). Predicted sea level rise will likely expand the areas suitable for restoration due to seawater intrusion and salinisation of overlying soils (Beresford 2004; Olsson et al. 2019; Tully et al. 2019), whilst increasing opportunity costs associated with adjacent higher elevation land (Runting et al. 2017).

Transaction and implementation costs are likely high for coastal wetland restoration projects but may be lower with natural regeneration from adjacent fringing wetland vegetation (Supplementary Figure S6 shows examples from our field study) (Naucmér and Enkvist 2009; Birch et al. 2010; Hagger et al. 2022). The remediation of waterlogged and salt-affected soils is likely expensive because it necessitates the repair or construction of new drainage infrastructure or high chemical inputs such as gypsum and ash (Gates and Grismer 1989; Ghassemi et al. 1995; Luke et al. 2017). Coastal wetland restoration for blue carbon may represent a viable alternative for landholders, particularly with the recent development of Australia's Emissions Reduction Fund blue carbon method, which is a consistent approach to the economic valuation of carbon abatement by tidal restoration projects (Australian Government (2021), Lovelock et al. 2022). However, site-specific cost-benefit analyses comparing opportunity and restoration costs as well as landholder consultation will be needed to assess the viability of restoring blue carbon ecosystems at specific locations.

Conclusions

Understanding the characteristics and causes of degradation in the coastal agricultural land of Queensland is important because it helps to elaborate the potential of degraded lands to be restored to their original wetland vegetation and in identifying trade-offs to agricultural production and land management. This study identified high levels of agricultural land degradation along the Queensland coast in areas that were previously *Melaleuca*-dominated forested wetlands, saltmarshes, mangroves, or other wetland ecosystems. Restoring pre-clearing wetlands in degraded agricultural lands could enhance blue carbon storage by protecting remaining soil carbon stocks, reducing greenhouse gas emissions, and restarting carbon sequestration in soils and biomass, and restore other pre-clearing ecosystem services. We argue that waterlogging and salinisation in these lands are barriers to agricultural production and may provide opportunities for coastal wetland restoration as an alternative land use that is more economically feasible than the restoration of productive agricultural land assuming low opportunity costs and landholder exploitation of carbon markets. Future work could focus on mapping degraded agricultural land across the region, quantifying carbon emissions in degraded and intensively cropped areas, and further understanding the costs of remediating agricultural land and the benefits of coastal wetland restoration, particularly costs of maintenance of drainage infrastructure and weed management, and benefits of carbon sequestration, enhancing biodiversity and increasing water quality in catchments of the Great Barrier Reef (Adame et al. 2020).

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