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Contrasting effects in tidal inundation under varying sea levels on the ecological structure and functions of tropical marsh ecosystems

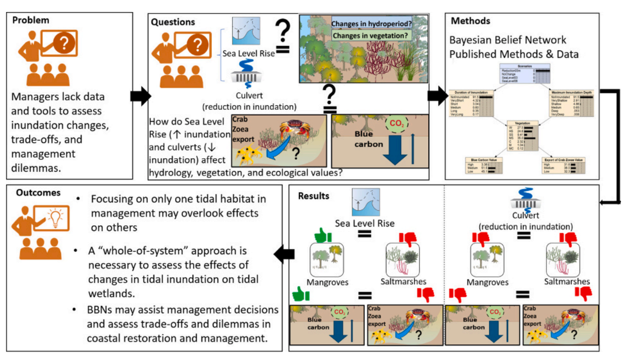
Cécile Vulliet *, Jack Koci, Marcus Sheaves, Nathan Waltham

TropWATER, Centre for Tropical Water and Aquatic Ecosystem Research, James Cook University, Bebegu Yumba, Townsville, QLD 4814, Australia
College of Science and Engineering, James Cook University, Bebegu Yumba, Townsville, QLD 4814, Australia

HIGHLIGHTS

- Coastal managers need simple management tools to assess future climate change likely consequences.
- Modeling found future sea levels may increase blue carbon value but reduce crab zoea export on saltmarshes.
- Landward expansion of mangroves at the loss of saltmarshes will reduce both blue carbon and crab zoea export.
- A whole-of-system approach is necessary to assess impacts on coastal ecosystems.

GRAPHICAL ABSTRACT



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ABSTRACT

Coastal managers continue to be confronted with making management decisions with few data available and insight of the outcomes. Practical tools that can be used to inform on the effects of different scenarios of changes are particularly important to assist decision-making. This study has applied a Bayesian Belief Network (BBN) to investigate the contrasting effects of Sea Level Rise (SLR) scenarios and a reduction in tidal inundation on a tropical tidal wetland mosaic including saltmarshes, mangroves, and intertidal mudflats. We investigated: 1) the habitability of the study site for tidal vegetation under different scenarios associated with changes in inundation; and 2) the probability that the ecological values of export of crab zoea and blue carbon be supported under the scenarios. The study highlights that, without the ability to adjust to future SLR scenarios, saltmarshes here are likely to be lost to mangroves, and open water, under a scenario of 0.8 m SLR. Tidal inundation reduction decreased mangrove cover but increased habitability for terrestrial vegetation and subtidal herbaceous saltmarshes. SLR is likely to positively affect the blue carbon value but decreases the likelihood of the site holding high crab zoeae export values in saltmarsh areas. In contrast, a reduction of tidal inundation declined the likelihood of the site holding both high blue carbon and crab zoeae export values. The findings highlight the importance of "whole-of-system" approach to assessing the effects of different scenario changes in tidal inundation. Focusing only on one tidal wetland habitat and a single targeted value may affect the structure and functions of other habitat components of the coastal ecosystem mosaic. BBNs are useful tools to summarise

* Corresponding author at: College of Science and Engineering, James Cook University, Bebegu Yumba, Townsville, QLD 4814, Australia.
E-mail address: cecile.vulliet@my.jcu.edu.au (C. Vulliet).

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preliminary assessments of the potential effects of tidal inundation changes on wetland ecosystems, which may assist managers to make the most informed decision to conserve and restore coastal transitional areas.

1. Introduction

The direct effects of human activities on ecosystems are exacerbated by rapidly changing climate and sea-level rise (SLR) (Boon et al., 2018). Those changes particularly threaten coastal ecosystems (Roy et al., 2023), such as saltmarshes (Adams, 2020; Raw et al., 2021; Rolando et al., 2023). Their position along coastlines, the most prized geographic areas in the world by humans for living, recreational activities, tourism, and commerce (Williams et al., 2022), makes them particularly vulnerable to human exploitation (Sandi et al., 2018; Martínez et al., 2024). Shifts in their distribution are already visible (Armitage et al., 2015; Raposa et al., 2017; Fagherazzi et al., 2019). This pattern is likely to continue to increase across this century (Langston et al., 2021), requiring careful management and restoration of coastline areas (Raposa et al., 2016; Schuerch et al., 2018; Nguyen et al., 2022).

Tidal wetlands such as saltmarshes provide incredible biodiversity, economic, cultural, and recreational values in many places (Barbier et al., 2011). For instance, they provide nursery and feeding grounds for fish (Litvin et al., 2018; Whitfield, 2017), support coastal fisheries by providing primary and secondary productions to coastal water (Hollingsworth and Connolly, 2006; Raoult et al., 2018; Taylor et al., 2018; Vulliet et al., 2024b), protect coastlines from erosion (Finotello et al., 2022), and improve water quality by processing nutrients (Sousa et al., 2010). Tropical saltmarshes have been the subject of few investigations compared to other coastal wetland habitats, like mangroves, but in the few studies available they are described as providing basal sources for coastal food web productivity (Sheaves et al., 2007) and providing habitat for benthic macrofauna (Reis et al., 2019; Reis and Barros, 2020). In addition, tidal connections to upper tropical tidal habitats where saltmarshes occur have been shown to export considerable quantities of crab zoea (Vulliet et al., 2024b) in a similar way to subtropical and temperate Australian saltmarshes (Saintilan and Mazumder, 2017). The synchronised release of substantial densities of crab zoeae by crabs has been seen as an important prey-pulse that links together high intertidal habitats with the remaining seascape, symbolising the importance of land-seascape connectivity (Vulliet et al., 2024b) and supporting fisheries value (Saintilan and Mazumder, 2017).

One of the most discussed values of tidal wetlands is the ability to assimilate carbon at higher rates and quantities than primary terrestrial forests, hence tidal wetland ecosystems are referred as "blue carbon ecosystems" (Lovelock and Duarte, 2019; Alongi, 2020a; Macreadie et al., 2021). The blue carbon value of coastal ecosystems makes them particularly attractive to mitigate climate change (Macreadie et al., 2021), while providing other critical services such as fisheries and storm protection (i.e., "the multifaceted nature of blue carbon ecosystems", Macreadie et al., 2019). Consequently, blue carbon ecosystems are at the centre of discussions in carbon offsets and credits (Friess et al., 2022; Zhang et al., 2024). Mangroves are often reported as having a high above-ground and below-ground carbon stock compared to saltmarshes and seagrasses (Kelleway et al., 2016; Alongi, 2020a, 2020b). For instance, reviews of carbon stocks within the first meter below the surface of mangroves found a mean of 255 Mg C ha^{-1} and a maximum of 683 Mg C ha^{-1} compared to a mean of 162 Mg C ha^{-1} and maximum of 259 Mg C ha^{-1} in saltmarsh soils (Duarte et al., 2013). In all, mangroves are considered a more valuable proposition for restoration over saltmarshes, with the focus only on blue carbon outcomes, which unfortunately ignores the range of other services that saltmarshes are known for. However, despite their lower carbon value compared to mangroves, saltmarshes still play an important role in blue carbon sequestration (Waltham et al., 2023). It is therefore necessary to assess how climate change and SLR might affect the carbon dynamics of saltmarshes, alongside the broader range of services they provide, to ensure that management efforts adequately address their ecological importance.

The intrinsic link between tidal hydroperiod (duration, depth and frequency of tidal inundation) and intertidal wetland distribution and connectivity patterns (Baker et al., 2013; Minello et al., 2012; Sheaves et al., 2024), makes increased tidal flooding associated with accelerating SLR potentially a major positive service that is offered by tidal wetlands (Raposa et al., 2016; Macreadie et al., 2017). For example, an increase in SLR may lead to the upland migration of saltmarshes at the expense of terrestrial woodland species (Vulliet et al., 2024c). Nevertheless, in the tropics and subtropics, mangroves are encroaching into saltmarshes in response to SLR (Kelleway et al., 2017), leading to a landward squeeze of saltmarshes to mangroves (Vulliet et al., 2024c; Wen and Hughes, 2022). Without elevation space and sediment availability to accommodate for this upland transgression, saltmarshes might be casualties of coastal squeeze (Willemsen et al., 2022), becoming lost naturally at the expense of mangroves or open waters.

Barriers to tidal inundation, such as culverts, have signficant consequences on tidal wetland vegetation distribution and extent (Mora and Burdick, 2013; Rodríguez et al., 2017). For instance, bund walls can increase the proliferation of, on the landward side of the wall, freshwater or terrestrial invasive weeds to the detriment of tidal wetland vegetation such as saltmarshes (Abbott et al., 2020; Karim et al., 2021). Artificial barriers can also change the wetting and drying of upper tidal wetland habitat (Rodríguez et al., 2017), hence changing groundwater and drainage dynamics and soil properties (Mora and Burdick, 2013), which can negatively affect saltmarsh and mangrove vegetation. Engineered barriers such as dams can also reduce sediment delivery to coastal areas, which is necessary for saltmarsh accretion (Cornacchia et al., 2024) and coastal erosion prevention (Wolanski and Hopper, 2022). The effects of man-made barriers can decrease the resilience of tidal wetlands to other anthropogenic stressors such as climate change (Rodríguez et al., 2017), further threatening the values and services provided by tidal wetlands.

Given intensifying degradation of coastal ecosystems, there have been developing interest in restoring tidal wetland connectivity and vegetation so that tidal wetland values and services are maintained or increased (Sheaves et al., 2021; Hagger et al., 2022; Raw et al., 2021; Waltham et al., 2021; Waltham et al., 2020). Simultaneously, growing interests in carbon offsets have raised public and political motivations in restoring or enhancing tidal wetland habitats to improve the blue carbon values of coastal areas. Nevertheless, managers are increasingly challenged to predict management and restoration outcomes with little quantitative data (Mishra and Farooq, 2022), and examples of successful restoration outcomes are few and far between (Bayraktarov et al., 2016; Billah et al., 2022).

The notion that restoring one value or habitat does not necessarily improve another creates trade-offs and contradictions in coastal ecosystem restoration and management (Yang et al., 2018; Yang et al., 2021; Choi et al., 2022). For instance, removal of bund walls to restore tidal connectivity and tidal wetlands so as to enhance the blue carbon values of coastal areas or to restore historical tidal vegetation (Karim et al., 2021) might be done to the detriment of freshwater habitats with nesting and bird habitat values. Conversely, while a detriment for freshwater species, this mitigation might enhance habitat values for mangrove specialist bird species (Canales-Delgadillo et al., 2019). Similarly, mangrove afforestation on bare tidal flats to enhance blue carbon sequestration and shoreline protection (Jia et al., 2018) can be performed at the expense of bare, though productive, tidal flats values (Bouma et al., 2014; Erftemeijer and In, 2000), such as decreasing the extent of critical foraging areas for threatened shorebirds (Choi et al., 2022). Another example is that an increase in mangrove encroachment due to SLR might improve the blue carbon value of coastal areas including mangroves which have a higher carbon reserve compared to saltmarshes (Kelleway et al., 2016). Yet, the effects of this encroachment are poorly understood (Kelleway et al., 2017) and might negatively affect the floristic and faunal diversity, and conservation values held by saltmarshes such as the provision of foraging grounds for threatened bats (Saintilan and Rogers, 2013) or the export of large quantities of crab zoea (Mazumder et al., 2009; Vulliet et al., 2024b).

Developing accessible modeling tools to assist scientists and practitioners observe these trade-offs and assess whether restoration measures achieve multiple outcomes while maximising ecological benefits is needed (Yang et al., 2021; Waltham et al., 2021). Bayesian Belief Networks (BBNs) are used increasingly in ecosystem management and conservation due to their ability to deal with complex systems with high degrees of uncertainty (McCann et al., 2006). While the implementation of BBNs requires practitioners with expertise in probabilistic modeling and ecological data interpretation, the "concise representation" (Coupé and Van der Gaag, 2002) of BBN and the user-friendly interface make it highly attractive for managers to inform decision-making on the outcomes and effectiveness of management or restoration measures. Marcot et al. (2023) illustrate the value of BBNs in assessing ecosystem resilience, specifically in tidal wetlands under SLR scenarios. BBNs can be structured using empirical data, expert knowledge, or a combination of both, and can be updated with new observations to improve predictive accuracy. Additionally, BBNs serve as effective tools for assessing vulnerability, identifying key uncertainties, and providing clear probability distributions to support risk analyses and decision-making under uncertainty (Marcot et al., 2023). Other examples of BBN application in coastal ecosystems include environmental risk level assessments (Jäger

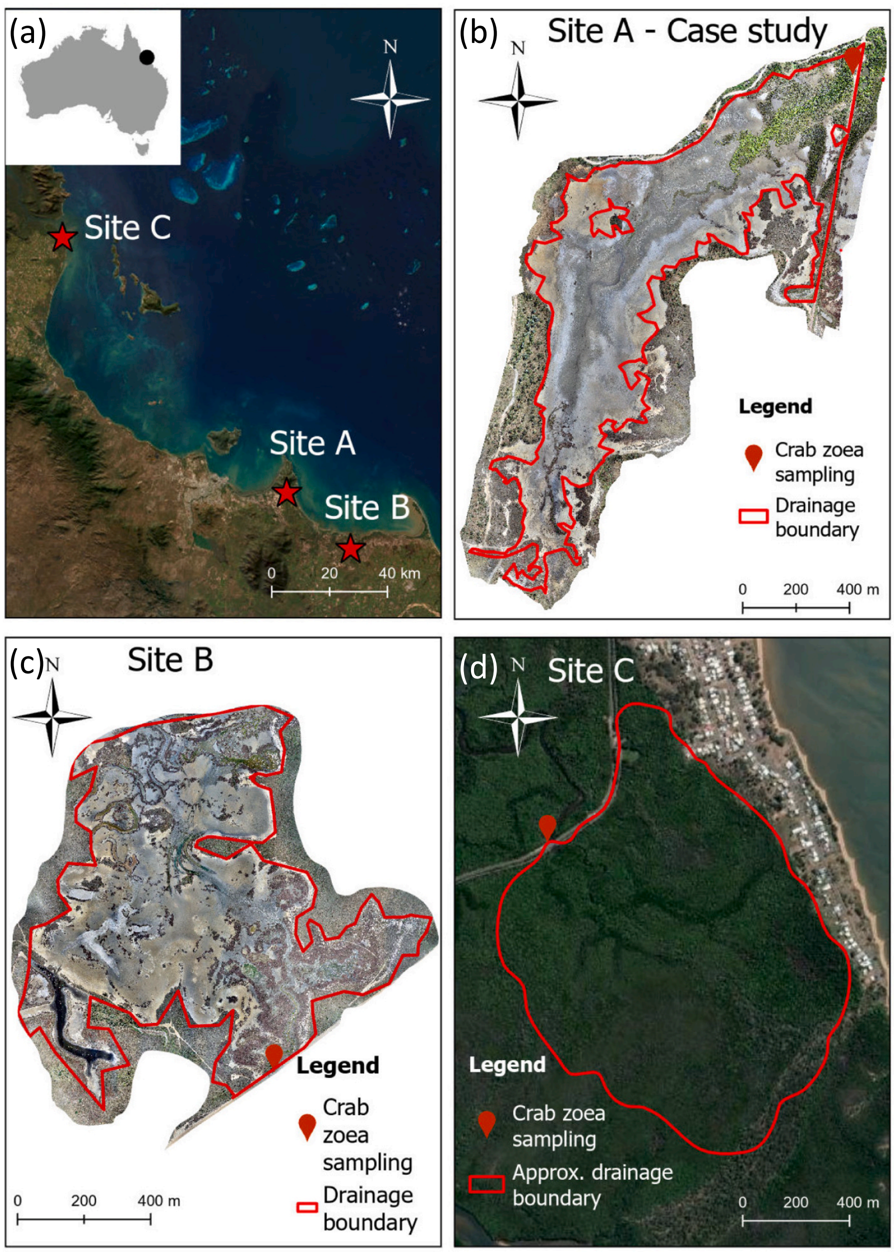

Fig. 1. Map of the study area showing (a) the geographic locations of the case study site (Study Site A) and of the two study sites where crab zoea were sampled (Site B – saltmarsh-dominated site and Site C-mangrove-dominated site) along the eastern Australian coast; (b) the orthophoto of Site A derived from UAV-SfM showing the position of the sampling of crab zoea; (c) the UAV-SfM orthophoto of Site B with the position of the sampling of crab zoea; and (d) satellite imagery of Site C showing the position of crab zoea sampling and the approximate drainage area of the site.

et al., 2018; Malekmohammadi et al., 2023), prediction of erosion due to SLR and assessment of SLR prevention measures efficiencies (Sahin et al., 2019), and the evaluation of tidal wetland values based on management strategies such as feral pig exclusion by fencing surrounding coastal habitats (Waltham et al., 2020).

The aim of this study was to use the BBN platform to illustrate the contrasting effects of SLR and a reduction in tidal inundation on a tropical tidal wetland complex that includes a mosaic of habitats including saltmarshes, mangroves, and mudflats. More specifically, we investigate the potential impacts of sea level rise scenarios (+0.3 m and +0.8 m) and reduction in tidal inundation (−0.3 m) on the habitability of the study site to tidal wetland vegetation (i.e., "habitability" from Cockell et al. (2016) which refers to the probability or ability of an environment to support or not support an organism). In addition, we observed how these changes may affect the blue carbon prospects of the site and its value in providing for trophic subsidies in estuaries via the export of crab zoea. The main objectives were to assess: 1) the habitability of the study site for tidal wetland vegetation under different scenarios of change inferred by maximum inundation depth and duration of inundation; 2) the probability that the values of export of crab zoea and blue carbon be supported under the different scenarios computed in (1); and 3) the potential of BBN to identify trade-offs or contrasting outcomes of hydrological changes such as SLR.

2. Methods

2.1. Case study area

The case study site is in the north part of Blacksoil Creek (Site A) in north Queensland, Australia (−19.297, 147.021) (Fig. 1). The site is within the Bowling Green Bay National Park, a Ramsar internationally important wetland complex adjacent to the Great Barrier Reef. Detailed descriptions of the site are found in Vulliet et al. (2024a). Briefly, the site represents a dry tropical wetland complex composed of mangroves dominated by the red mangrove, *Rhizophora stylosa*, at low elevation near channels and seaward edge of the site followed by the grey mangrove, *Avicennia marina* and the yellow mangrove, *Ceriops tagal* and *Ceriops australis* at higher elevations. The site is then composed of mudflats and saltpans including succulent saltmarshes (including the bead weed *Sarcocornia quinqueflora*) and herbaceous saltmarsh (dominated by the salt couch, *Sporobolus virginicus*) below and near the highest astronomical tide (HAT). Additional details on this study site and vegetation are found in Vulliet et al. (2024a) and Vulliet et al. (2024c).

Two other sites were used to obtain data on crab zoea export and to inform the values of tidal wetlands in the BBN (Fig. 1). The Jerona site (Site B) (−19.450, 147.228, 89.21 ha) is found south of Site A and has the same climate and tidal characteristics as Site A. This site also lies within the Bowling Green Bay National Park and is listed under the Ramsar Wetland Convention. The site also has an intersecting road with a small (1 m) 1-pipe culvert. The mangrove community is dominated by *A. marina*, along the channel and at the saltmarsh-landward margin. The Blind-Your-Eye Mangrove, *Excoecaria agallocha*, is another species found at the terrestrial margin at Site B. The site is mostly dominated by saltmarsh vegetation. The saltmarsh vegetation community is dominated by succulent marsh, notably the bead weed, *Sarcocornia quinqueflora*, with isolated patches of the glasswort, *Tectinoria* spp., the grey samphire; *Tecticornia australasica*, the pigweed; *Portulaca* spp., the pigface; *Carpobrotus glaucescens*, the prickly saltwort; *Salsola australis*; and the seablite; *Suaeda australis*. The herbaceous saltmarsh community is composed of the saltcouch, *Sporobolus virginicus* with green couch, *Cynodon dactylon*, and jointed rush, *Juncus kraussi*. The saltmarsh vegetation at Site B is less visibly affected by human activities than at Site A, although vehicle damage is apparent at some locations. In addition, there are signs of erosion of herbaceous saltmarsh at tidal-terrestrial margin.

The Lucinda Site (Site C) (−18.542, 146.330) is mangrove-dominated (Table 1). The site is in the Wet Tropics near the Lucinda township. The site is mostly dominated by mangrove vegetation, although some small patches of succulent saltmarshes were found on the landward side of mangroves (pers. observations). The site has a mesotidal and semi-diurnal regime. The highest astronomical tide is 1.15 m above AHD (Boswood et al., 2007). The site is also bisected with a road with a large multi-pipe culvert. The culvert was where crab zoea sampling was conducted (see below for details on crab zoea sampling).

2.2. Bayesian Belief Network

2.2.1. Development

Bayesian Belief Networks (BBNs) are a graphical modeling framework (represented by a directed acyclic graph-DAG) that illustrates causal relationships between variables based on probabilities computed using the Theorem of Bayees (for detailed methods, description and examples of applications of BBNs, see Jensen and Nielsen (2007), Henriksen and Barlebo (2008), Dlamini (2010), and Loftin et al. (2018)).

A BBN is first built qualitatively by listing all the variables potentially influencing the targeted study system (Marcot et al., 2006). In the case of the effects of different sea levels on tidal wetland vegetation distribution, these variables are summarised in Fig. 2 and include the complex feedback and interactions of surface hydrology, local tidal dynamics, topography, microtopography, temperature, accretion rates, vegetation type, estuarine and coastal geomorphology, meteorological conditions, climatic events, edaphic variables, nutrients inputs and underwater hydrology, as well as biological interactions such as predation, facilitation, competition and diseases (see reviews from Friess et al., 2012; Saintilan et al., 2019; Rogers and Krauss, 2019). Here, the study has used field and modelled data only based on the hydrodynamic model computed for Site A (Vulliet et al., 2024a; Vulliet et al., 2024c) to inform the BBN on the habitability of the site under different SLR scenarios and reduction of tidal inundation. Hence, the model consisted of only two parent nodes: inundation maximum depth and inundation duration that were informed using field data. The BBN was built with Netica 6.09 (Norsys Software Corporation 2021).

2.2.2. Cases and data inputs

2.2.2.1. Duration of inundation and maximum depth. Outputs from a hydrodynamic model of average-tide scenarios computed in Vulliet et al. (2024c) were used to inform the BBN on the duration of inundation and maximum inundation depth at the study site (only the area above the main concrete road was included here to represent the area drained by the zooplankton sampling pump, see Section 2.2.2.3). The rasters were reclassified to obtain discrete states for the duration of inundation and maximum depth (Table 2) to populate the probability tables. The

Table 1
Wetland cover at the case study site (Site A), Site B, and Site C used to inform the crab zoea value node in the BBN. The land covers at Site A and Site B were calculated using the UAV-derived orthophotos whose methods are described in Vulliet et al. (a, b). Land cover at Site C was estimated to be predominately mangroves (based on on-site visual and satellite imagery assessments) with some saltmarsh patches (on-site observations as the patches could not be seen via satellite imagery).

Land cover	Site A Per cent of total wetland area (%)	Site B Per cent of total wetland area (%)	Site C Per cent of total wetland area (%)
Mudflat/saltpan	65.87	61.01	5
Herbaceous saltmarsh	16.41	9.28	0.1
Succulent saltmarsh	8.56	29.17	0.1
Ceriops spp.	6.24	0	0
Mangroves	2.53	0.31	93.25
Main channel	0.37	0.22	0.75

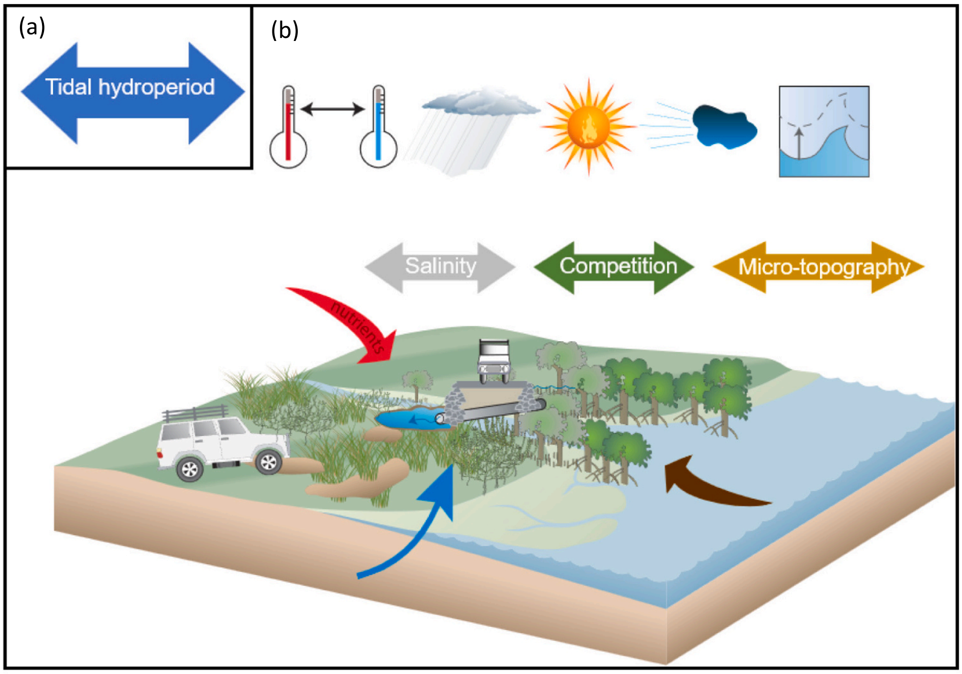

Fig. 2. Conceptual diagram of (a) the processes considered in the present study by opposition to (b) the likely interacting processes, together with tidal hydroperiod (b) that shape saltmarsh and mangrove dynamics in tropical seascapes and influence their vulnerability to sea-level rise and human interventions.

Table 2
Discretisation of the hydroperiod continuous values (maximum inundation depth and duration of inundation) used to inform the parent nodes in the BBN.

Parent node	Range of values	Discrete state
Duration of inundation (hours)	None = 0	Not inundated
	[0;12]	Very short
	[12;48]	Short
	[48;163]	Medium
	[163;560]	Long
	≥560	Very long
Maximum inundation depth (m)	None = 0	Not inundated
	[0;0.1]	Very shallow
	[0.1;0.4]	Shallow
	[0.4;0.8]	Medium
	[0.8;1.5]	Deep
	≥1.5	Very deep

outputs rasters were discretised into six states from no inundation (or depth) to very long duration of inundation (or very deep maximum depth) (Table 2). Following Loftin et al. (2018), the rasters were resampled to 0.1 m resolution and converted to points. The value to point was extracted to create a database with the number of pixels (calculated from the points) for each state. The percent probability fed into the tables was calculated by dividing the number of pixels for each state by the total number of pixels of the study site. This was repeated for the SLR scenarios of 0.3 m and 0.8 m, and a reduction in inundation of 0.3 m.

2.2.2.2. Case study 1: habitability of the site for tidal wetland vegetation type. In this case, the probability of finding a wetland type based on tidal inundation conditions (described in Section 2.2.2.1) influenced by scenarios of changes was assessed. The probability was considered as a measure of habitability. The child node "Vegetation" was divided into seven states representing land cover type (W = woodland/Grassland; HS = herbaceous saltmarsh; SS = succulent saltmarsh; C = Ceriops mangrove species (*Ceriops tagal* and *Ceriops australis*); M = *A. marina/ R. stylosa* (abbreviated as "M" for other mangroves than *Ceriops* spp.); MS = mudflat/saltpans; and MC = main channel).

The 3-cm resolution land cover data derived in Vulliet et al. (2024a) and Vulliet et al. (2024c) using unattended-aerial-vehicle structure-from-motion (UAV-SfM) photogrammetry to obtain information on the distribution of tidal wetland vegetation (herbaceous saltmarsh, succulent saltmarsh, *Ceriops* spp., and other mangroves), unvegetated flats (mudflats/saltpans), and terrestrial vegetation (woodland/grass). Their distribution was expressed by the number of pixels per parent node state combinations (i.e., inundation duration and maximum inundation depth) (using the same geoprocessing workflow for the hydroperiod variables above) (Fig. 3). To populate the Computational probability tables (CPTs), percent probabilities were calculated by dividing the number of pixels extracted for the land cover type and for the combination of parent node states (e.g., number of pixels with succulent saltmarsh having very short inundation and shallow maximum depth) by the sum of pixels of the given combination of the parent nodes across all land cover type (Fig. 3).

2.2.2.3. Case study 2 - values and outcomes: likely probability of crab zoea export. The export of crab zoea by saltmarsh crab's is seen as a key value supporting the need to ensure, frequent, inundation of saltmarshes in temperate (Mazumder et al., 2009) and subtropical Australia (Hollingsworth and Connolly, 2006). By sampling zooplankton outputs in relation to tidal height over multiple days, Vulliet et al. (2024b) described a predictable pattern in the export of crab zoea in relation to tidal height at the case study site (Site A). This study was the first to describe the same patterns in crab zoea export in relation to tidal hydroperiod in tropical tidal wetlands than that of temperate and subtropical saltmarshes (Mazumder et al., 2009; Saintilan and Mazumder, 2017). Nevertheless, although crab zoea export was related to the extent of succulent saltmarsh inundation it was also related to the extent of *Ceriops* spp. and saltpan/mudflats inundation. This makes it difficult to clearly associate crab zoea export as a value that is more importantly attributed to saltmarshes compared to values presented for temperate and subtropical Australian saltmarshes (Mazumder et al., 2009; Saintilan and Mazumder, 2017). Therefore, additional unpublished data was used to further inform the BBN on the crab zoea export values of tidal wetlands.

(a)

Scenarios	NotInnund...	VeryShallow	Shallow	Medium	Deep	VeryDeep
Reduction03m	75.904	9.51	12.699	1.71	0.14	0.0374
NoChange	54.381	12.897	27.573	4.846	0.256	0.048
SeaLevel03	34.626	6.629	39.205	18.279	1.1	0.161
SeaLevel08	16.655	3.291	13.343	42.611	23.905	0.196

(b)

Scenarios	NotInnund...	VeryShort	Short	Medium	Long	VeryLong
Reduction03m	75.904	11.306	7.94	2.597	1.527	0.727
NoChange	54.381	20.905	14.704	6.914	2.541	0.554
SeaLevel03	34.626	13.5	19.61	21.675	8.155	2.434
SeaLevel08	16.655	7.008	7.758	18.395	42.71	7.475

(c)

MaximumDepth	DurationOfInnundation	W	HS	SS	MS	C	M	MC
NotInnundated	NotInnundated	30.03	26.356	9.859	31.754	1.288	0.713	7.40e-6
NotInnundated	VeryShort	x	x	x	x	x	x	x
NotInnundated	Short	x	x	x	x	x	x	x
NotInnundated	Medium	x	x	x	x	x	x	x
NotInnundated	Long	x	x	x	x	x	x	x
NotInnundated	VeryLong	x	x	x	x	x	x	x
VeryShallow	NotInnundated	x	x	x	x	x	x	x
VeryShallow	VeryShort	0.0948	0.779	5.548	91.19	2.059	0.329	4.01e-5
VeryShallow	Short	0.0476	0.313	4.126	87.362	7.059	1.091	1.52e-5
VeryShallow	Medium	x	x	x	x	x	x	x
VeryShallow	Long	x	x	x	x	x	x	x
VeryShallow	VeryLong	x	x	x	x	x	x	x
Shallow	NotInnundated	x	x	x	x	x	x	x
Shallow	VeryShort	8.70e-3	13.828	6.114	64.16	15.884	5.80e-3	0
Shallow	Short	3.33e-3	0	4.862	85.256	9.057	0.822	8.73e-4
Shallow	Medium	4.13e-4	0.0844	1.896	15.836	68.399	13.775	9.08e-3
Shallow	Long	0	0	0	0	0	70.588	29.412
Shallow	VeryLong	0	100	0	0	0	0	0
Medium	NotInnundated	x	x	x	x	x	x	x
Medium	VeryShort	1.528	0	2.248	65.86	28.108	2.255	0
Medium	Short	0.0434	0	1.181	96.24	0.221	2.245	0.0696
Medium	Medium	2.23e-3	0	0.824	11.669	56.528	30.94	0.0373
Medium	Long	0	0	0	0.117	24.072	74.775	1.036
Medium	VeryLong	0	0	0	0	0	0	100
Deep	NotInnundated	x	x	x	x	x	x	x
Deep	VeryShort	0.569	0	7.032	54.338	37.682	0.379	0
Deep	Short	0.334	0	1.969	53.252	40.639	3.715	0.0899
Deep	Medium	0.0392	0	0.147	42.116	12.048	39.11	6.54
Deep	Long	2.03e-3	0	0	0.457	6.164	52.041	41.336
Deep	VeryLong	0	0	0	0.0231	0	3.985	95.992
VeryDeep	NotInnundated	x	x	x	x	x	x	x
VeryDeep	VeryShort	x	x	x	x	x	x	x
VeryDeep	Short	0	0	1.456	72.491	26.053	0	0
VeryDeep	Medium	0	0	0.0219	25.487	10.587	48.821	15.083
VeryDeep	Long	0.135	0	0	3.231	6.369	12.988	77.278
VeryDeep	VeryLong	0	0	0	0	0.027	2.116	97.857

(d)

Vegetation	High	Medium	Low
W	0	0	100
HS	0	100	0
SS	0	100	0
MS	0	50	50
C	100	0	0
M	100	0	0
MC	0	0	100

(e)

Vegetation	High	Medium	Low
W	0	0	100
HS	35.983	63.629	0.388
SS	69.897	29.756	0.347
MS	46.261	49.948	3.791
C	0.323	2.633	97.044
M	0.323	2.633	97.044
MC	0	0	100

Fig. 3. Computational probability tables (CPTs) of (a) the parent nodes "Duration of inundation" and (b) Maximum depth; (c) the child node "Vegetation" showing the cominations of the states of the parent nodes; (d) the child node "Blue carbon value"; and (e) the child node "Crab zoea export value". "HS" Herbaceous saltmarsh; "SS" = Succulent saltmarsh; "MS" = Mudflats/Saltpans; "C" = *Ceriops* spp.; "M" = *R. stylosa/A. marina* (abbreviated as "M" for mangroves other than *Ceriops* spp.).

The same method and sampling apparatus deployed at Site A were used at Site B (dominated by succulent saltmarshes) (over one week in February–March and one week in June 2022, simultaneously sampled with Site A) and Site C (dominated by mangroves, over one week in July 2022 as geographic constraint prevented simultaneous deployment with Sites A and B (Fig. 1)). By using the same sample processing methods described in Vulliet et al. (2024b), it was found that the succulent saltmarsh-dominated site (Site B) provided the highest densities of crab zoea per m^3 (Fig. 4).

This information was used to inform the child node "Export of Crab Zoea Value" (Fig. 5). The states of the BBN node "Export of Crab Zoea Value" were divided into High, Medium and Low. According to the field data (Fig. 4), each of the states were assigned one of the values: Site B (saltmarsh-dominated site) had the highest value as export of crab zoea and was therefore qualified as "High", while Site A (case study site) was assigned to "Medium", and Site C ("mangrove-dominated site) to "Low". To populate the probability table, we related the crab zoea value to the percentage of land cover found at each site (Table 1). For instance, succulent saltmarsh at Site B was 20.1 % compared to 9.2 % at Site A, and 0.1 % at Site C. Therefore, the row of succulent saltmarsh in the CPT was set to 68.3 for High, 31.3 % for Medium, and 0.3 % for Low (Fig. 3) (calculated as the percentage of the given land cover at the given site (e.g., 20.1 % for Site B) divided by total percentage of cover for the given land cover category across the three sites (here 29.4 % for succulent saltmarsh).

For Sites A and B, the proportion of each land cover was derived from the 3-cm UAV-SfM orthomosaic maps created for each site following the methods described in Vulliet et al. (2024a) and Vulliet et al. (2024c). As no UAV-SfM orthomosaic map was derived for Site C, the land cover type at Site C was broadly estimated using GoogleEarth and was assigned as predominantly mangroves (Table 1).

2.2.2.4. Case study 3 - values and outcomes: likely probability of blue carbon value. The values of blue carbon between tidal wetland types were investigated in the literature to inform the BBN. The carbon storage capacity was used as a proxy to inform the blue carbon value node, although it is acknowledged that saltmarsh and mangrove contribution to carbon flow is known to vary substantially (Alongi, 2020a). For instance, mangroves have a higher carbon storage capacity compared to saltmarshes, but rates of C_{org} burial is greater in saltmarsh soils (although not statistically significantly greater) (Alongi, 2020a, 2020b). Nevertheless, carbon storage was used here to simplify the BBN.

Based on the literature examined (Alongi, 2020a; Brown et al., 2021; Kelleway et al., 2016; Murray et al., 2011; Perera et al., 2022; Phang et al., 2015; Rogers et al., 2016; Ruiz-Fernández et al., 2018), mangroves had an overall higher blue carbon value compared to saltmarshes. In addition, several studies have highlighted the likely positive effects of SLR on coastal blue carbon capacities as mangroves encroachment becomes more important (Kelleway et al., 2016). Hence, mangroves and *Ceriops* spp. were assigned as having high value ("High"), while succulent and herbaceous saltmarsh were assigned a medium value ("Medium"). Mudflats/saltpans were assigned to both "Medium" and "Low", and woodland/terrestrial and main channel was set to "Low".

2.2.2.5. Sensitivity analysis and model validation assessments. Sensitivity analyses are commonly used in the validation process of BBN (Coupé and Van der Gaag, 2002). Sensitivity analyses test how much a finding in one variable influences the beliefs of another variable. Sensitivity analysis results can then be used to refine or obtain more accurate information on targeted input parameters (Rohmer and Gehl, 2020). It is also used as a measure of model output robustness. Model sensitivity was analysed using the in-built function "Sensitivity to Findings" in the Netica software. Sensitivity was calculated on the vegetation, crab zoea value, and blue carbon value node for each scenario. In addition, the root mean square of error (RMSE), mean error, and coefficient of correlation R^2 were calculated between the observed (calculated from the land cover) and simulated (calculated by the BBN in the child node "Vegetation") land cover percentage to assess the accuracy of the BBN in

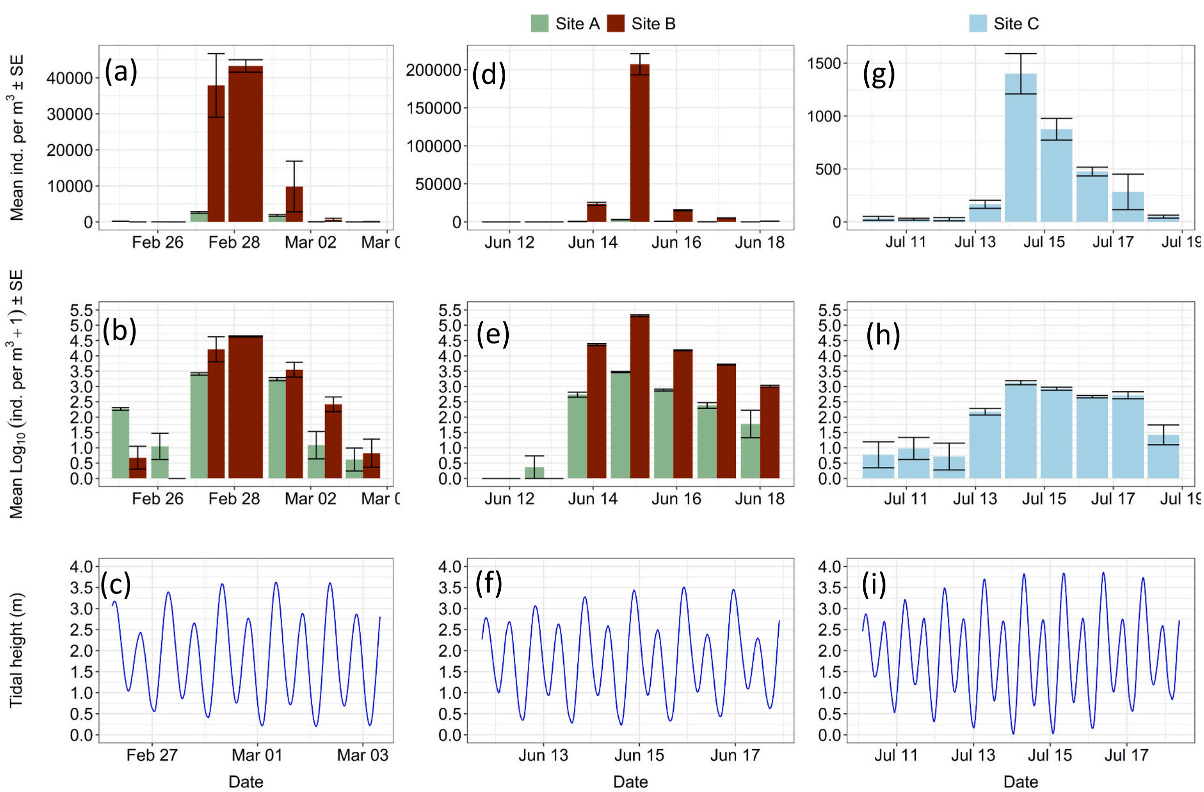

Fig. 4. Mean crab zoea densities per sampling day (i.e., per samples of 5 replicates of 6 ml – the mean and standard error (SE) are the sample mean and standard error) sampled at Site A (Blacksoil Creek, the case study site site), Site B (Jerona, the saltmarsh-dominated site), and Site C (Lucinda Site, the mangrove-dominated site) showing (a, d, g) mean log10(individual per m^3 + 1) ± SE, (b, e, h) mean individual per m^3 ± SE, and (c, f, i) corresponding observed water height (HW) recorded at Cape Ferguson tidal gauge for Site A and Site B and at the Cardwell tidal gauge for Site C. Note that for visualisation purposes, the scale of the y-axis of a, d, g are different on each graph.

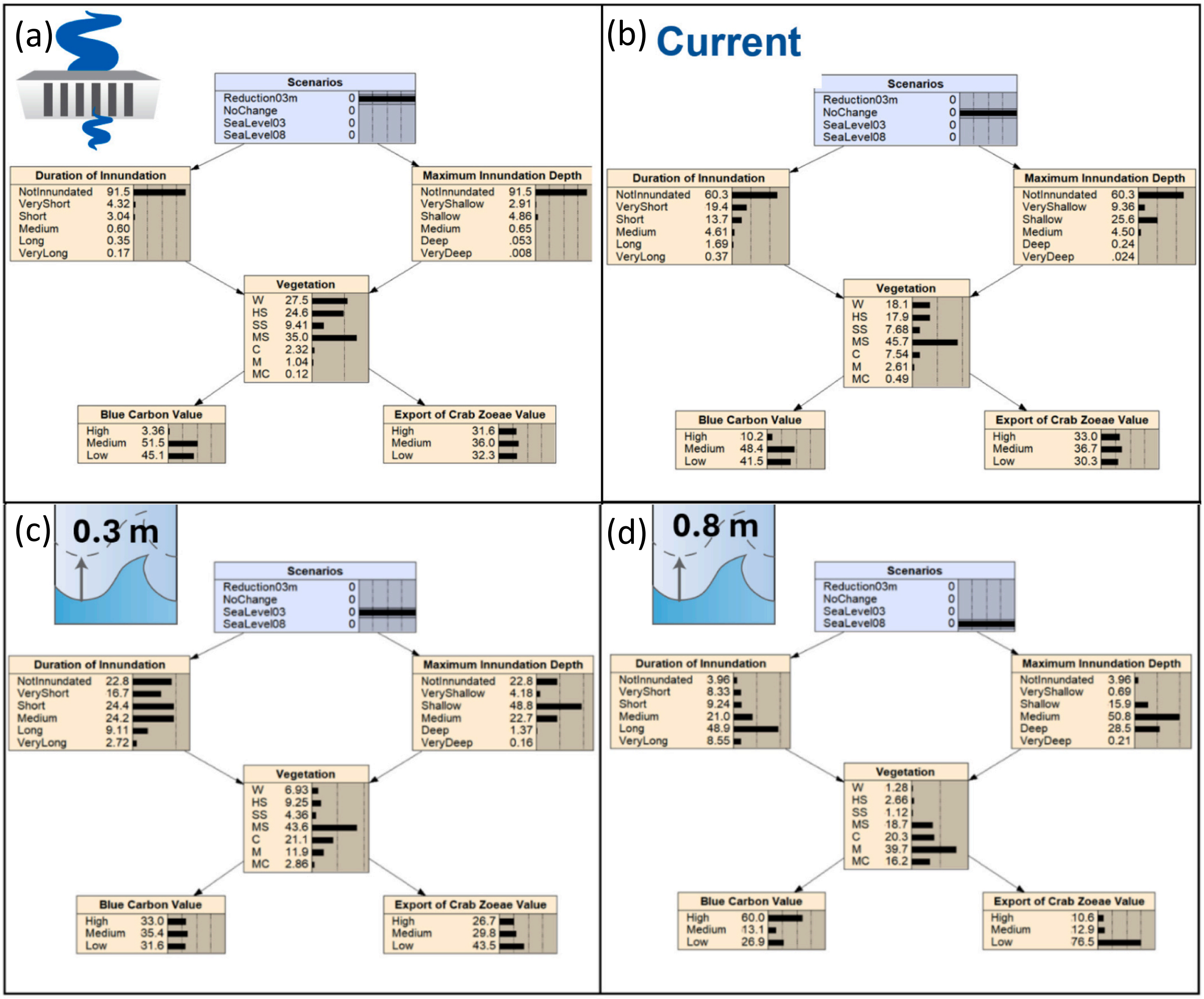

Fig. 5. Bayesian belief networks predicting changes in the habitability of the study site to vegetation, and to hold blue carbon and crab zoea export value under (a) a reduction of 0.3 m in tidal inundation from the current average tide scenario; (b) current average tidal inundation conditions expressed as duration of inundation and maximum duration depth (see Table 1 for explanation of the discrete states); (c) 0.3 m sea-level rise (SLR) scenario; and (d) 0.8 m SLR. "HS" = Herbaceous saltmarsh; "SS" = Succulent saltmarsh; "MS" = Mudflats/Saltpans; "C" = *Ceriops* spp.; "M" = *R. stylosa/A. marina* (abbreviated as "M" for mangroves other than *Ceriops* spp.).

predicting vegetation cover from the parent nodes.

2.2.3. Assumptions

The BBN model developed here assumes that inundation duration and maximum depth are the primary drivers of tidal wetland vegetation distribution under SLR scenarios. However, other factors, such as groundwater dynamics, human disturbances, biotic interactions, and past rainfall patterns, are likely important drivers of wetland dynamics at the study site (Vulliet et al., 2024c) but were not included due to data limitations. Sediment dynamics and potential upward migration were also excluded due to a lack of data, resulting in a 'coastal squeeze' scenario where wetland retreat is constrained (Borchert et al., 2018; Vulliet et al., 2024c). Probabilities for each state were derived from hydrodynamic model outputs and field observations, assuming stationarity in environmental conditions over the modelled period - a limitation discussed in Vulliet et al. (2024a, 2024b, 2024c).

3. Results

3.1. Effects of scenarios on the duration of inundation and maximum depth

At the current sea level, the hydrological conditions of the study site were principally characterised by no inundation (54.4 of the 0.1 m pixels) (Fig. 3). Inundation duration and maximum depth were predominantly very short (0 to 12 h during the entire simulation time (i.e., one month) (20.9 % of the pixels) and shallow (0.1–0.4 m) (27.6 % of the pixels). An increase in sea level of 0.3 m shifted the tidal hydrological conditions of the site towards medium (48–163 h, 21.7 % of the pixels) and short inundation (12–48 h, 19.6 % of the pixels), but shallow maximum depth (0.1–0.4 m, 39.2 % of the pixels). Longer inundation times (163–560 h) became notably more predominant (42.7 % of the pixels) under a 0.8 m SLR scenario, while the probability of deep maximum depth (0.8–1.5 m) increases (23.9 % of the pixels), although medium depth (0.4–0.8 m) was the most likely maximum depth condition (42.6 % of the pixels). A reduction in inundation of 0.3 m from the current scenario shifted the study site to a smaller inundation area (with 75.9 % of the pixels being not inundated).

3.2. Effects of scenarios on site habitability to tidal wetland vegetation

Under the current scenario, mudflat/saltpans were the dominant land cover (45.7 % of pixels), followed by woodland (18.1 %), herbaceous saltmarsh (17.9 %), succulent saltmarsh (7.7 %), and *Ceriops* spp. (7.5 %). *A. marina/R. stylosa* represent 2.6 % of the pixels (Fig. 5). With an increase in SLR of 0.3 m, the system will likely shift towards a mangrove-dominated system with an increasing habitability of 21.1 % for *Ceriops* spp. and 11.9 % for mangroves. The extent of the main channel also increases from 0.5 % of the pixels under the current situation to 2.9 % under SLR +0.3 m. The habitability for saltmarsh vegetation decreased to 4.4 % for succulent saltmarsh and to 9.2 % for herbaceous saltmarsh. The site habitability for saltmarsh vegetation is predicted to be negligible under 0.8 m SLR with 1.12 % for succulent saltmarsh and 2.7 % for herbaceous saltmarsh. Habitability for *A. marina/R. stylosa* increased to 39.7 %, although its habitability for *Ceriops* spp. decrease slightly from the 0.3 SLR scenario (20.3 %). The main channel is predicted to occupy a considerably larger extent of the site, with 16.2 % of pixels predicted to be the main channel under the 0.8 m SLR, which is 6 times and 33 times, respectively, greater than in the 0.3 m SLR and current scenario.

A reduction in inundation of 0.3 m from the current scenario will likely result in woodland/terrestrial vegetation increasing from 17.9 % to 27.5 %. Herbaceous saltmarsh remains the dominant vegetated tidal wetland (24.6 %) and the habitability for succulent saltmarsh increases slightly (9.4 %). The habitability for *A. marina/R. stylosa* and *Ceriops* spp. decreased sharply to 1.04 % and 2.3 %, respectively.

3.3. Effects of scenarios on crab zoeae export and blue carbon values

Under current conditions, the likely probabilities of the site holding the export of crab zoea and blue carbon values are 36.7 % and 48.4 % respectively, charaterising a medium value (Fig. 5). Under a 0.3 m SLR scenario, the probability of the site holding crab zoeae value decreases (low, 40.9 %). Conversely, the probability of having a high blue carbon value increases from 10.2 % to 33.0 %. The 0.8 m SLR has important consequences for the ability of the site to hold a high crab zoea export value (10.6 %), which becomes more likely low category (76.5 %). On the other hand, the blue carbon value of the site becomes mainly high (60.0 %) under a 0.8 m SLR.

A reduction of inundation of 0.3 m is predicted to be slightly less promising for holding high crab export zoea value (31.6 %), with a slight increase in the probability (low category, 32.3 %) from the current conditions. A reduction in inundation markedly decreases the probability of having a high blue carbon value (3.4 %).

3.4. Sensitivity analysis and model assessment

Sensitivity analyses were run separately on each child node for the current scenario. Vegetation was most sensitive to duration of inundation (23.1 %) and maximum depth (16.4 %). Blue carbon value was sensitive to vegetation (66.6 %) but least sensitive to duration of inundation (13.2 %) and maximum depth (7.6 %). The export of crab zoea was most sensitive to vegetation (48.7 %), and only slightly to duration of inundation (6.2 %) and maximum inundation depth (1.4 %).

The model reliably predicted the land cover percentage according to duration of inundation and maximum depth, with a RMSE between observed and simulated land cover percentage for the current scenario of 0.87 %, a mean error between observed and simulated cover of 0.0043 % and a coefficient of correlation R^2 of 0.99.

4. Discussion

4.1. Habitability to tidal wetland vegetation under likely SLR scenarios

The study predicts a decline in herbaceous and succulent saltmarsh with the expansion of mangroves and an increased main channel area, with near total loss of saltmarshes under the projected 0.8 m SLR scenario likely to eventuate by 2100 in the study region (Queensland, 2019). These predictions align with studies reporting losses in saltmarsh cover in response to mangrove encroachment associated with SLR (Krauss et al., 2011; Oliver et al., 2012; Saintilan and Rogers, 2013; Sandi et al., 2018). However, the present results contradict other studies which highlight that SLR may not lead to the disappearance of saltmarsh when local factors such as sediment dynamics and availability of space for saltmarsh retreat landward as well as tidal range, local topography, geomorphology, and vegetation type allow for saltmarsh adaptation (Alizad et al., 2018; Schuerch et al., 2018). For instance, Kumbier et al. (2022) identified mangrove encroachment into saltmarshes in a microtidal south-eastern Australian estuary under a 0.4-m and 0.9-m SLR scenario using an eco-morphodynamics model, but saltmarshes were able to extend landward and their cover increased slightly. Here, the approach only considered tidal wetland duration and inundation as predictors of the site habitability to tidal wetland vegetation under SLR, and did not consider contextual and dynamic eco-geomorphological feedbacks (Kirwan et al., 2016), and the possibility for saltmarshes and terrestrial vegetation to migrate landward (Schuerch et al., 2018). Therefore, the modelled scenarios here are likely more representative of a "coastal squeeze" situation, where saltmarshes cannot migrate further inland (up elevation, horizontal migration) and adjust their elevation by sediment and biological accretion (vertical migration) (e.g., Borchert et al., 2018).

The present findings are important because they highlight that saltmarshes will be almost totally lost at the study site by the end of the century without the ability to migrate landward in response to SLR. This suggests that management efforts need to address the potential adaptation strategies (e.g., Sheaves et al., 2016; Wigand et al., 2017; Leo et al., 2019), that need to be implemented to alleviate the loss of saltmarsh ecosystems to SLR. If passive responses are chosen (e.g., no action, self-adaption, or abandon, Sheaves et al., 2016), it is important to carefully address the trade-offs associated with an increase in the likelihood of a shift to a mangrove-dominated system, so that potential social, economical, and ecological consequences are anticipated.

4.2. Habitability to tidal wetland vegetation under a reduction in tidal inundation

A reduction in tidal height of 0.3 m has slightly positive effects on the habitability of the study site to saltmarsh, which were likely to expand at the expense of unvegetated flats and mangroves. This finding aligns with previous studies that indicate an increase in saltmarsh cover and reduced mangrove encroachment when using automated controlled tidal gates to counteract SLR and preserve saltmarshes in Australian tidal wetlands (Sadat-Noori et al., 2021; Rankin et al., 2023). Nevertheless, the type of saltmarsh vegetation advantaged by a reduction in inundation here contrasted with those studies. Specifically, the habitability of the site to *S. virginicus* increased more importantly (7 %) compared to succulent saltmarsh (2 %), contrasting with Rankin et al. (2023) where those authors observed an increase in succulent saltmarsh (*S. quinqueflora*) in the medium and high marsh, with little increase in herbaceous saltmarsh cover (*S. sporobolus*) under a 0.45 m tidal height reduction. In addition, the BBN predictions suggest a 10 % increase in the habitability of the study site to terrestrial vegetation under a 0.3 m decrease in tidal height, while Rankin et al. (2023) observed minimal changes in terrestrial vegetation cover. These contrasting results highlight that apparently "similar" habitats may respond differently to a reduction in tidal inundation across different locations. While this shift to terrestrial vegetation could be influenced by the limited number of study sites and scenarios, it also underscores the role of site-specific factors in influencing vegetation responses. This highlights the need for careful consideration of site-specific nuances in saltmarsh areas (Waltham et al., 2021), including the extent and distribution of tidal wetland vegetation and their tidal inundation characteristics, when planning management strategies or urban development that involve altering tidal inundation, such as roads and culverts.

4.3. Probability of holding blue carbon and crab zoeae export values under SLR scenarios

The findings demonstrate that the impacts of SLR have contrasting effects depending on the components and values of the coastal ecosystem mosaic considered. The BBN predictions indicate that SLR scenarios and associated expansion in mangrove cover strongly enhanced the blue carbon value of the site, aligning with global trends that show expanding mangrove cover has positive effects on carbon storage at the expense of saltmarsh loss (Kelleway et al., 2016; Rogers et al., 2019; Simpson et al., 2019). The increase in the probability of the site holding a high blue carbon value in the 0.3 m SLR scenario (21 %) is notably similar to the measured increase in the carbon storage capacity (22 %) of a Floridian mangrove-saltmarsh ecotone after ten years of encroachment (Doughty et al., 2016). By contrast, SLR negatively impacted the ability of the site to maintain a high crab zoea export value, especially under the 0.8 m SLR scenario where succulent saltmarsh loss considerably reduced the probability of the site holding a high crab zoea export value.

The decline in crab zoea export value under SLR scenarios raises concerns about the potential negative consequences of losing this functional process on coastal ecosystem productivity and intertidal biodiversity (e.g., saltmarsh crab diversity, Mazumder, 2009). The export of crab zoeae from saltmarshes is a unique feeding opportunity for small zooplanktivorous fish (Hollingsworth and Connolly, 2006; McPhee et al., 2015), and the trophic relay that exists between saltmarsh production and coastal food webs (Saintilan and Mazumder, 2017). In addition, coastal productivity and functions are usually not linked to a segregated, unique component of the coastal ecosystem mosaic (Weinstein and Litvin, 2016). Rather, it is the synergies within the diverse components of the coastal ecosystem mosaic and their heterogeneity that support key coastal ecosystem functions and values, from supporting commercially important species (Taylor et al., 2018; Scapin et al., 2022) to enhancing carbon and nitrogen sequestration (Saavedra-Hortua et al., 2023) and coastal protection (Marin-Diaz et al., 2023). Therefore, it is possible that the predicted reduction in the heterogeneity of the coastal ecosystem mosaic under rising sea levels may negatively impact overall coastal productivity, with consequences such as a reduction in the provision of coastal ecosystem services (Gilby et al., 2021) and resilience to climate and anthropogenic changes (Aguilera et al., 2020; Li et al., 2021; Wang, 2021). It is therefore important that coastal management strategies consider future SLR projections and direct anthropogenic alterations in tidal inundation on these less "obvious" synergies within the coastal ecosystem mosaic - such as marsh trophic subsidies (Weinstein et al., 2014) and prey pulses (Vulliet et al., 2024b) - so that unexpected management and restoration outcomes can also be anticipated (Sheaves et al., 2021).

4.4. Probability of holding blue carbon and crab zoeae export values under a reduction in tidal inundation

The findings show the negative effects of a reduction in tidal inundation on both the crab export and blue carbon values, underlining the importance of maintaining tidal connectivity to support critical tidal wetland ecosystem functions and processes. Even what may appear as small reductions in tidal inundation (0.3 m reduction in tidal height and 25 % of the study site being less inundated), could lead to important shifts in the ecological structure of tidal wetlands (e.g., loss of mangrove vegetation), which may subsequently alter the values and functions of tidal wetlands (Abbott et al., 2020; Gilby et al., 2021). Coastal development and management decisions need to assess these broader implications when implementing barriers to tidal flow. By contrast, the increase in blue carbon and crab export values from the reduction in tidal inundation scenario and current scenario support that removing barriers to tidal flow may increase the blue carbon propositions (Kelleway et al., 2017) and fisheries value (Abbott et al., 2020) of tidal wetlands under future SLR projections.

4.5. Implications for management and future studies

This study illustrates that a "whole-of-system" approach to understanding the components and processes in coastal ecosystems is necessary when considering the potential outcomes of direct (e.g. human disturbances) and indirect changes (e.g. climate change and hydrology changes). The study illustrates that looking at a particular component of a system (e.g., mangroves) separately from the others could lead to unforeseen consequences on other system components and their functions (Choi et al., 2022). Management and restoration strategies will therefore need a holistic approach that evaluate (and test the sensitivity) the potential outcomes of different scenarios and management strategies on multiple ecological values, such as those developed here or elsewhere, rather than focusing on one targeted ecosystem or outcome (e.g., planting mangroves over mudflats to increase carbon sequestration) (Choi et al., 2022). For instance, in the case of the study site, balancing multiple ecological benefits (e.g., crab zoea export and blue carbon) may require maintaining current tidal inundation patterns or allocating space for saltmarsh upward migration. This would result in keeping principally medium and high probabilities of holding both crab zoeae export and blue carbon values at the study site without considerably reducing one or the other. While we have illustrated this juxtaposition using SLR and a reduction in tidal inundation, this approach could be applied with other scenarios, such as planting mangroves over mudflats (e.g., Choi et al., 2022) or reconverting freshwater wetlands to intertidal ecosystems (Karim et al., 2021). In addition, adding information on other key values provided by tidal wetland ecosystems, such as bird breeding and foraging grounds (e.g., Spencer et al., 2009), and fish nursery value (e.g., Whitfield, 2017), would provide a more holistic understanding of the trade-offs and variability in outcomes given different scenarios of changes on key processes and patterns.

This study adds to the knowledge that simple BBNs are useful for preliminary assessments of the dilemmas and directions that may need to be taken when elaborating management or restoration strategies (McCann et al., 2006). This is important because one of the "proposed activities" under objective 1.3, "*Wetlands and other coastal ecosystems are managed from a "whole-of-catchment perspective*"", of the "Reef 2050 Wetlands Strategy" (Department of Environment, 2023) highlights the need to "*redevelop and encourage use of a decision support tool to improve management decisions and prioritization*" (p. 13, Queensland, 2016). BBNs are also useful for identifying thresholds and where more research should be allocated at a targeted study site. For instance, in the case of the study site, there is a need to obtain a better understanding of the additional potential important drivers of the observed vegetation patterns (e.g., groundwater flow, nutrient availability) (Vulliet et al., 2024c) as well as whether local topography and anthropogenic context enable landward migration (e.g., Borchert et al., 2018) and sediment supply and subsurface processes (e.g., plant growth, decomposition, subsistence) (Beckett et al., 2016; Chen et al., 2024) enable saltmarshes to keep up with the vertical pace of SLR. Applying a similar BBN workflow using these additional variables in the responses of tidal wetland vegetation to SLR would be necessary to decrease uncertainties in the predictions.

Contextual variably in site structure (e.g., proportion of saltmarsh and mangrove cover) and functions (e.g., Bradley et al., 2020; Ziegler et al., 2021) also imply that each site may have unique trade-offs and dilemmas. Different sites may require different management responses (Waltham et al., 2021), and some sites (e.g., Ramsar wetlands, Sadat-Noori et al., 2021; Rankin et al., 2023) may be prioritised for management actions over other sites. For instance, in the case of the three study sites, the site with the highest saltmarsh cover and crab zoeae export value (i.e., Jerona, Site B) may be prioritised for saltmarsh management over the main study site (Site A), whose saltmarsh cover was lower and affected by 4-wheel vehicle driving (Vulliet et al., 2024a). Managers could use similar BBN approaches to identify sites that should be prioritised for management and restoration programs.

5. Conclusion

Predicting future changes in the values of coastal ecosystems is challenging due to the important complexity of interacting factors shaping coastal processes together with the high uncertainties in those processes. Concurrently, managers are increasingly challenged to make management decisions with little understanding of how these decisions benefit non-targeted values. The present study has demonstrated the potential of BBN in identifying how different tidal wetland values (e.g., habitability for a certain type of vegetation) might shift due to changes in tidal wetland inundation due to SLR or human interventions. In addition, this study advocates for BBN as a decision tool that may be useful for managers to identify trade-offs when considering management strategies in response to increasing SLR over other scenarios of changes in tidal inundation and tidal wetland vegetation.

CRediT authorship contribution statement

Cécile Vulliet: Writing – original draft, Visualization, Validation, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Jack Koci:** Writing – review & editing, Validation, Supervision, Resources. **Marcus Sheaves:** Writing – review & editing, Validation, Supervision. **Nathan Waltham:** Writing – review & editing, Visualization, Validation, Supervision, Resources, Conceptualization.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

Data will be made available on request.

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