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Managed Retreat of Saline Coastal Wetlands: Challenges and Opportunities Identified from the Hunter River Estuary, Australia

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Abstract We analyse the potential impacts of sea-level rise on the management of saline coastal wetlands in the Hunter River estuary, NSW, Australia. We model two management options: leaving all floodgates open, facilitating retreat of mangrove and saltmarsh into low-lying coastal lands; and leaving floodgates closed. For both management options we modelled the potential extent of saline coastal wetland to 2100 under a low sea-level rise scenario (based on 5 % minima of SRES B1 emissions scenario) and a high sea-level rise scenario (based on 95 % maxima of SRES A1FI emissions scenario). In both instances we quantified the carbon burial benefits associated with those actions. Using a dynamic elevation model, which factored in the accretion and vertical elevation responses of mangrove and saltmarsh to rising sea levels, we projected the distribution of saline coastal wetlands, and estimated the volume of sediment and carbon burial across the estuary under each scenario. We found that the management of floodgates is the primary determinant of potential saline coastal wetland extent to 2100, with only 33 % of the potential wetland area remaining under the high sea-level rise scenario, with floodgates closed, and with a 127 % expansion of potential wetland extent with floodgates open and levees breached. Carbon burial was an additional benefit of accommodating landward retreat of wetlands, with an additional 280,000 tonnes of carbon buried under the high sea-level rise scenario with floodgates open (775,075 tonnes with floodgates open and 490,280 tonnes with floodgates closed). Nearly all of the Hunter Wetlands National Park, a Ramsar wetland, will be lost under the high sea-level rise scenario, while there is potential for expansion of the wetland area by 35 % under the low sea-level rise scenario, regardless of floodgate management. We recommend that National Parks, Reserves, Ramsar sites and other static conservation mechanisms employed to protect significant coastal wetlands must begin to employ dynamic buffers to accommodate sea-level rise change impacts, which will likely require land purchase or other agreements with private landholders. The costs of facilitating adaptation may be offset by carbon sequestration gains.

Keywords Saline coastal wetland · Sea-level rise · Carbon sequestration · Elevation model · Sediment accretion · Threshold

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Introduction

The ecological and environmental services provided by saline coastal wetlands have been recognized for some time (Costanza et al. 1998; Costanza et al. 1989). Depending on their situation, these wetlands may act to dampen flood waters, recycle nutrients, provide timber and stabilize banks (Ewel et al. 1998; Lee et al. 2006). They provide essential habitat and contribute to the sustenance of a range of biota including commercially and recreationally important fish, invertebrates, waterbirds, bats and macropods (e.g. kangaroos and wallabies) (Kwak and Zedler 1997; Mazumder et al. 2006; Spencer et al. 2009). Recent reviews have documented the efficiency of saline coastal wetlands as carbon sinks, with more carbon sequestered per unit area than any other natural ecosystem type (Alongi 2012; McLeod et al. 2011). However, saline coastal wetlands (specifically mangrove and saltmarsh) throughout the world are continuing to diminish in size as a consequence of the construction of levees, drains and floodgates. The conversion of substantial areas of saline coastal wetland to aquaculture has also occurred in recent decades (see for example Primavera 2006). Australia has a long history of saline coastal wetland drainage and reclamation. In coastal New South Wales the bulk of this drainage work had been completed by 1969 and Goodrick (1970) established that there had been a loss of 41,000 ha of fresh and saline wetlands principally due to drainage. Over 4,200 structures with the potential to impede tidal exchange have been identified in this area (Williams and Watford 1997) and these structures have promoted the establishment of pasture grasses and macrophytes more suited to brackish or freshwater conditions. These brackish and freshwater communities do not support coastal fisheries, are likely to be less efficient in the retention of carbon (Poffenbarger et al. 2011) and have been sources of poor water quality that has caused fish kills from acidification (Sammut et al. 1995) and deoxygenation (Walsh et al. 2004).

The Convention on Wetlands (Ramsar) has provided a framework for contracting parties (162 national governments) to list wetlands of international significance and voluntarily commit to their preservation. Other saline coastal wetlands are dependent upon national and local planning and legal frameworks, such as no net loss (USA), and local protected area legislation. In Australia, with the exception of wetlands listed as internationally important under the Ramsar Convention, wetland protection largely occurs at a state level through the acquisition of land as part of the reserve system, such as National Parks, or filters down to local and regional levels through legislation and policy.

Unfortunately, in Australia neither the Ramsar framework nor State policies deal well with the preservation of dynamic landscapes. Wetlands listed under the Ramsar Convention have official management boundaries defined as polygons drawn at the time of listing. Similarly, wetlands within reserves or protection zones are frequently managed according to gazetted boundaries. These boundaries may be derived from photography captured in the past and in the time since gazettal, boundaries may change, and have the potential to change at an accelerating rate as sea levels rise. While these planning and conservation instruments have served saline coastal wetlands well in the past, their utility in the conservation of critically important coastal ecosystems will diminish over time if they fail to accommodate the retreat of wetlands at a local scale.

Despite planning for saline coastal wetland protection in Australia being based on policy and legislation at state jurisdictions, planning actions that promote in situ adaptation and migration of saline coastal wetlands with sea-level rise is largely undertaken at the local scale (see for example Burley et al. 2012; Shoo et al. 2013). However, the need for effective coastal wetland legislation, policy and management is becoming even more imperative. The Intergovernmental Panel on Climate Change (IPCC) projected sea-level rise of between 18 and 59 cm above present levels by the end of the twenty-first century (Meehl et al. 2007). However, recent research indicates that this may underestimate sea-level rise, with current observations tracking the highest scenarios (Nicholls et al. 2011) and revised estimates of sea-level rise at the end of the century in the order of 47 to 100 cm (Horton et al. 2008), 30 to 215 cm (Grinsted et al. 2010) and 59 to 180 cm (Jevrejeva et al. 2010).

Saline coastal wetlands may have some capacity to adapt to accelerated sea-level rise and hydrological alterations through in situ processes that maintain their elevation relative to water levels. There has been considerable research into these processes over recent years, with results stressing the importance of sedimentation (Oliver et al. 2012; Rogers et al. 2012), and also the accumulation of root mass (McKee 2011; McKee et al. 2007; Nyman et al. 2006) as strategies by which saline coastal wetlands may adapt to sea-level rise. The latter has attracted interest as a mechanism for the burial of atmospheric carbon, with the associated benefit of offsetting emissions (Chmura et al. 2003; Duarte et al. 2005). As sea-level rise is likely to impact the resilience of these valuable ecosystems it is timely that consideration be given to appropriate management actions that will promote in situ adaptation of saline coastal wetlands, and their associated benefits.

In this paper we model sea-level rise impacts and adaptation strategies for an internationally important wetland. The Hunter River, NSW, Australia, supports an extensive temperate saline coastal wetland system comprising mangrove, saltmarsh and mixed mangrove–saltmarsh habitats. A large portion of these wetlands (4,257 ha) are formally protected within the Hunter Wetlands National Park and this includes over 2,926 ha listed as internationally important under the Ramsar Convention in 1984. The distribution of these wetlands within the lower Hunter has been shaped by the legacy of a network of 176 levees, culverts and floodgates that were constructed in the period 1950–1980 to create land suitable for agriculture and industry and to protect this land from flooding under the Hunter Valley Flood Mitigation Scheme (Williams and Watford 1997; Winning and Saintilan 2009). Wetland areas behind levee banks have transitioned from saltmarsh to brackish reedswamps dominated by *Schoenoplectus subulatus*, *Typha orientalis* and *Phragmites australis* (Winning and Saintilan 2009). Part of this estuary is heavily developed, being the site of the city of Newcastle, as well as one of the largest coal-loading facilities in the world. The lower Hunter therefore presents a useful case study in the management of internationally important and highly dynamic wetlands in the context of strong development pressure and anticipated sea-level rise.

We apply sea level projections provided by the IPCC (Meehl et al. 2007) and Hunter (2010) to the accretion/elevation model developed by Rogers et al. (2012) for the Hunter River to explore the adaptation capacity, resilience and mitigation opportunities for saline coastal wetlands. In doing so, we test the effect of flood mitigation structures on the distribution of saline coastal wetlands under low and high sea-level rise scenarios, and consider the differing carbon burial benefits of options relating to the management of floodgates and levees. This modelling allows for the exploration of the costs and benefits of planning and management actions that can be implemented at both state and local scales that promote the maintenance of wetland extent.

This is the first paper to quantify carbon burial benefits associated with contrasting sea-level rise adaptation options for saline coastal wetlands. We model two management options: (1) opening all floodgates and levees, facilitating retreat of mangrove and saltmarsh into low-lying coastal lands; and (2) leaving floodgates closed and levees elevated to protect coastal lowlands. For both management options we modelled the extent of saline coastal wetland to 2100 under a low sea-level rise scenario and a high sea-level rise scenario. For this we applied a dynamic elevation model (Rogers et al. 2012), which factored in the accretion and vertical elevation responses of mangrove and saltmarsh to rising sea levels, using locally derived data on these processes. We then calculated the volume of sediment accumulated across the estuary under the two management options for each sea-level rise scenario, and converted this to carbon accumulated using published estimates for the same wetland system (from Howe et al. 2009).

Methods

Study Site

The Hunter River (151°48′ E, 32°55′ S) has a length of approximately 300 km and drains a relatively large catchment area of 22,000 km^2 (Fig. 1). The Hunter River forms a permanently open barrier estuary with a semi-diurnal tidal regime and a tidal range of 1.9 m at the river mouth. In geomorphological terms, the Hunter estuary is in a mature evolutionary stage (Roy et al. 2001). Saline coastal wetlands on the Hunter

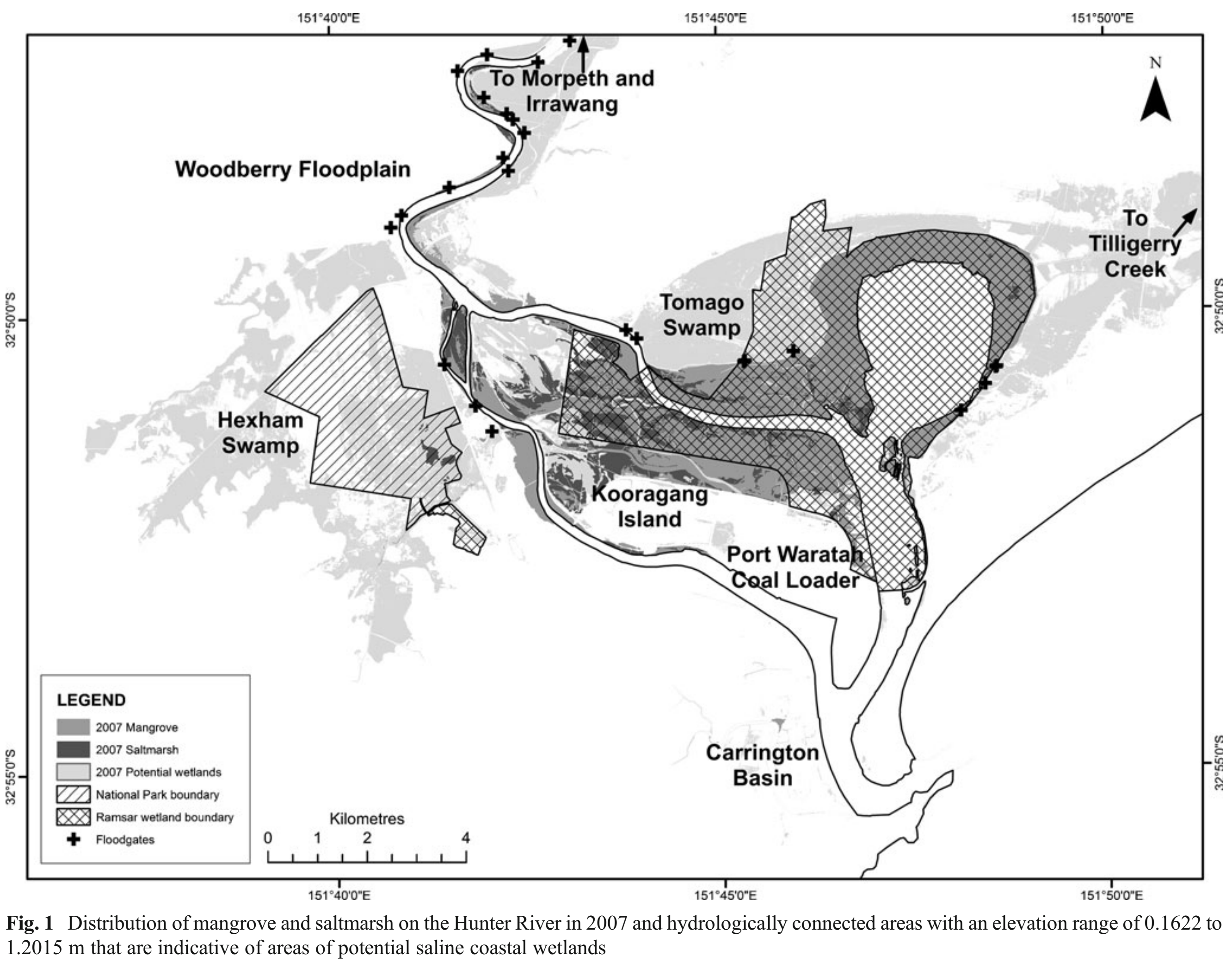

Fig. 1 Distribution of mangrove and saltmarsh on the Hunter River in 2007 and hydrologically connected areas with an elevation range of 0.1622 to 1.2015 m that are indicative of areas of potential saline coastal wetlands

River have undergone significant structural and morphological change since European occupation due to urban development as well as works associated with flood mitigation and drainage. Research by Rogers et al. (2006) indicates that incursion of mangroves into saltmarsh can be explained by relative sea-level rise in the region.

This study focuses on saline coastal wetlands located within the spatial extent of a light detection and ranging (LiDAR)-derived digital elevation model. Consequently, the analysis excludes saline coastal wetlands extending northeast from Fullerton Cove along Tilligerry Creek, and to the northwest on the Woodberry Floodplain, and at Morpeth and Irrawang.

Water Level Data

We used the maximum monthly water level model developed by Rogers et al. (2012) as a baseline for water level projections. This model was developed by time series analysis of monthly maximum water levels obtained from the Tomaree ocean level gauge using ARIMA modelling techniques. The model was selected as it maintained the upward trend in water level and accounted for seasonal trends associated with high spring tides in June/July and December/January. The maximum monthly water level predicted by the model for January 2010 was 2.9851 m and this was used as a baseline for water level projections.

Hunter (2010) provides minima (5th percentile) and maxima (95th percentile) projections of sea-level rise according to SRES scenarios from the IPCC fourth assessment report (Meehl et al. 2007) at decadal resolution to 2100. Two sea level projections from Hunter (2010) were selected for application to the landscape elevation model so as to project the full range of IPCC scenarios:

- Projections of sea level for 5th percentile minima according to the B1 emissions scenario, which has a projected sea-level rise between 1990 and 2100 of 18.5 cm.
- Projections of sea level for 95th percentile maxima according to the A1FI emissions scenario, which has a projected sea-level rise between 1990 and 2100 of 81.9 cm.

These scenarios were selected to capture the range of possible sea-level rise projections presented by the IPCC. These scenarios were disaggregated further to annual resolution by applying a quadratic curve of fit to each trend. Predicted values for each annual increment were added to the baseline water level value for January 2010 to generate projected maximum monthly water level values for the Tomaree water level gauge that corresponded to the 5th percentile minima according to the B1 emissions scenario and 95th percentile maxima according to the A1FI emissions scenario.

Accretion and Elevation Data

A standard technique to assess the in situ vulnerability of saline coastal wetlands to sea-level rise is the Surface Elevation Table/Marker Horizon (SET/MH) technique (Cahoon et al. 2000). The SET/MH technique quantifies the rate of sedimentation within a wetland and the extent to which that sedimentation contributes to raising the elevation of the wetland in response to sea-level rise. A network of SETs and MHs was established on Kooragang Island, Hunter River, in 2000 to investigate the vulnerability of these wetlands to sea-level rise. Data from the network was used to develop an accretion/elevation model that was driven by catchment scale processes such as water level and rainfall and local factors such as elevation and distance to waterways (Rogers et al. 2012). This model can be used to explore management options and planning guidance that facilitates adaptation of mangrove and saltmarsh on the Hunter River to sea-level rise. Increasingly, SET data are also being used to estimate the rate of carbon sequestration in saline coastal wetlands, and estimates of the rate of carbon burial in mangrove and saltmarsh from the Hunter SET dataset have recently been published (Howe et al. 2009).

Landscape Elevation Model

A digital elevation model (DEM) based on light detection and ranging data collected in January 2007, which reportedly has an accuracy of ±15 cm (Fugro Spatial Solutions, Pty, Ltd 2007), was used as the basis for the landscape model and to provide an understanding of the relationship between saline coastal wetland elevation and distribution. Distance to main channel was generated as a layer for input into the model.

Rogers et al. (2012) employed empirical wetland surface elevation and accretion data collected from mangrove and saltmarsh communities on the Hunter River to develop the following expression ($r^2=0.8643$, assumption of normality true) to explain changes in accretion occurring on the Hunter River estuary between January 2002 and May 2010:

$$A = -18.0118\,V + 0.0037\,W + 33.6177\,X - 0.0034\,Y + 0.0095\,Z - 0.0123\,(V-0.6958)\,(W-1880.8519) - 59.4570\,(V-0.6958)\,(X-2.9220) + 0.0114\,(W-1880.8519)\,(X-2.9220) - 0.1735\,(V-0.6958)\,(W-1880.8519)(X-2.9220) - 80.8973$$

Where:

A	Vertical accretion (mm)
V	Elevation
W	Time
X	6-month average water level
Y	Distance
Z	Rainfall intensity

The model developed by Rogers et al. (2012) is a relatively simple equilibrium model, whereby sediment volume (both organic and mineral components) is considered to be a function of relative elevation, which is influenced by a range of processes operating within catchments and estuaries, such as rainfall and water level. While Rogers et al. (2012) recognize that increasingly more sophisticated (and complex) models are available, they intentionally developed a simple model that: (1) utilized accessible information; and (2) could be used to assess planning and management scenarios.

This model assumes that accretion varies in proportion to a number of drivers that could be readily projected into the future; namely water level, distance from the main channel and rainfall intensity, which was defined as the total monthly rainfall from four rainfall gauges within the Hunter River catchment; Cessnock Airport (151.34° E, 32.79° S), Williamtown RAAF (151.84° E, 32.79° S), Jerrys Plains Post Office (150.91° E, 32.50° S) and Scone Airport (150.83° E, 32.03° S) gauges. Elevation was subsequently estimated by the addition of accretion to the previous elevation model. The model can be employed spatially within ARCGIS (ESRI® ARCMAP™ 10.0), with the LiDAR-derived DEM used as the benchmark elevation model. Rogers et al. (2012) justify excluding the contribution of fetch, wind shear and wave action due to the geomorphic setting of the wetlands within the Hunter estuary. This model also assumed that accretion was negligible at the landward wetland boundary, rainfall intensity is constant over the study period and equal to the long-term average rainfall for the region; and the distance to the sediment source is constant over the study period (Rogers et al. 2012).

In ARCGIS, we applied projected water level data to the model expression developed by Rogers et al. (2012) to generate landscape models of accretion and elevation according to the two selected scenarios; 5th percentile minima according to the B1 emissions scenario and 95th percentile maxima according to the A1FI emissions scenario. These scenarios are henceforth termed the low sea-level rise scenario (5 % minima of SRES B1 emissions scenario) and high sea-level rise scenario (95 % maxima of SRES A1FI emissions scenario). As elevation was a driver within the accretion model and water level was projected at annual increments, models of accretion and elevation were developed at annual increments to 2100. In accordance with Rogers et al. (2012), the accretion model was limited to the area of inundation and was assumed to be negligible in terrestrial areas.

Rogers et al. (2012) grouped mangrove and saltmarsh elevation distributions due to the overlapping elevation range of these communities in 2007 and estimated that mangrove and saltmarsh on the Hunter River occurred at a DEM elevation range of between 0.1622 and 1.2015 m. This accounts for 95 % of DEM cells within the mapped distribution of saline coastal wetlands. Approximately 2,442 ha of saline coastal wetland were delineated on the Hunter River in 2006. An additional 6,970 ha occurs within elevation ranges suitable for saline coastal wetlands (Fig. 1); however, it is important to note that the delineation of the potential extent of wetlands was limited by the northeast and northwest extent of the DEM; hence, the potential extent of wetlands is likely to be greater than that projected on the DEM. These additional areas do not currently support saline coastal wetlands due to management structures, such as flood levees and watercourse floodgates, which limit tidal flows to these areas of essentially private land. The DEM elevation range of 0.1622 to 1.2015 m was used to project the potential distribution of saline coastal wetlands on the lower Hunter River at 2100 according to the selected sea-level rise scenarios. Potential wetland extent was excluded from built-up areas where future expansion would be limited, namely Carrington Basin and the Port Waratah Coal loader; and a 26 ha section of the Tomago floodplain which has been infilled since the original LiDAR survey was undertaken in 2007. This approach assumes that inundation is the primary requirement for future wetland establishment and does not account for other limiting factors. Factors limiting mangrove and saltmarsh establishment have been considered elsewhere (see for example Erfanzadeh et al. 2010; Krauss et al. 2008), and the re-establishment of efficient carbon burial has been shown to be dependent upon time (Osland et al. 2012).

We estimated the volume of sediment required to facilitate wetland elevation adjustment at 2100 by subtraction of the benchmark DEM from the generated landscape elevation models using the Cut Fill volumetric analysis tool within the Spatial Analysis extension. We also explored a 'floodgates closed' management scenario where floodgates were maintained with no management action taken to reinstate tidal exchange (by opening floodgates) to low-lying areas thereby arresting the retreat of saline coastal wetlands into areas occupied by freshwater wetlands and farmland. As wetland extent is currently maintained through a network of floodgates that control tidal flows, we assumed that these floodgates would remain viable under higher sea levels and would continue to inhibit tidal exchange in the future. We used the landward limit of the mapped saline coastal wetland extent in 2007 as a surrogate for the extent of wetlands at 2100 under the floodgates closed management scenario. This extent was used to extract the volume of sediment required for in situ adaptation of wetlands under a floodgates closed management scenario and low and high sea-level rise scenarios.

While there is evidence that rates of carbon burial may vary with the age of regenerating wetlands (Osland et al. 2012), we assumed that carbon burial was constant over time. We also focussed on carbon burial and did not include other components of the carbon cycle that may offset carbon burial, such as gas efflux occurring in response to carbon decomposition (Adams et al. 2012; Kirwan and Mudd 2012). Hence, we assumed that carbon burial is proportional to accretion, and converted the volume of sediment into carbon content using estimates of total carbon density (Mg m^{-3}) provided by Howe et al. (2009) for the Hunter saline coastal wetlands. These estimates distinguished between natural wetlands and disturbed wetlands, the disturbed category consisting of floodplains modified by agricultural activity into which tidal flows had been reinstated. These wetlands were found to have lower carbon content by approximately 50 %, and we used these estimates as they better represented the wetlands likely to be created under the "floodgates open" scenarios. Howe et al. (2009) estimated carbon density at 0.0301 Mg m^{-3} for mangrove and 0.0420 Mg m^{-3} for saltmarsh in disturbed sites. As we did not distinguish between mangrove and saltmarsh in our modelling we used an average figure of 0.03605 Mg m^{-3} for total carbon density.

Results

Landscape Elevation Model

Based on projected sea-level rise under the low sea-level rise scenario the landscape elevation model projected a 20 % increase in the area that may support mangrove and saltmarsh by 2100, henceforth identified as projected potential saline coastal wetland area, with all floodgates open (Table 1, Fig. 2). This expansion occurred incrementally at both the upper and lower boundaries of the projected potential saline coastal wetland area (Fig. 4). The model also indicated an expansion in the area of potential saline coastal wetland within the Ramsar area of 566 ha and within the Hunter Wetlands National Park of 1,108 ha. The increase in elevation of the wetland surface by 2100 equates to a soil volume (both autochthonous and allochthonous) of 1.67×10^7 m^3, corresponding to 602,035 tonnes of carbon buried.

Under the high sea-level rise scenario the model projected a decrease in potential saline coastal wetland area in the order of 56 % by 2100 with all floodgates open (Fig. 3). Approximately 2.15×10^7 m^3 of soil volume is required to facilitate wetland elevation adjustment by 2100 under the high sea-level rise scenario, corresponding to 775,075 tonnes of carbon sequestered. The limited capacity of saline coastal wetlands to respond in situ to sea-level rise is evident under a high sea-level rise scenario. Specifically, we found that the projected potential saline coastal wetland area increased incrementally at both the upper and lower boundaries of the projected saline coastal wetland extent until 2060; after 2060 marked submergence of lower intertidal environments was evident. By 2100, projected potential saline coastal wetlands primarily occurred in areas that did not support saline coastal wetlands in 2007. The model indicated that only 202 ha of potential saline coastal wetland will occur within the Ramsar area, a decrease of 1,393 ha from the present extent. Similarly, only 618 ha would remain in the national park, representing a decrease of 992 ha from the present extent.

The majority of wetland conversion by 2100 requires management actions that facilitate the reinstatement of tidal exchange and the subsequent conversion of generally privately owned grazing land to wetland. Based on the floodgates closed management scenario we estimated that approximately 1.08×10^7 m^3 of material, comprising both mineral and organic matter, would be required to facilitate wetland elevation adjustment under the low sea-level rise scenario, while approximately 1.36×10^7 m^3 of sediment is required under the high sea-level rise scenario. This corresponds to carbon burial of 389,349 and 490,280 tonnes, respectively.

Discussion

Wetland Adaptation and Resilience

We applied a low sea-level rise scenario to a model developed using empirical values of accretion and elevation. This demonstrated that saline coastal wetlands may have some capacity to build elevation at rates equivalent to sea-level rise (Fig. 2), in this case sea-level rise ranging from 1.2–2.2 mm-year^{-1}. According to the low sea-level rise scenario the model projected landward retreat of potential saline coastal wetland areas at rates approximately equivalent to sea-level rise. The seaward boundary exhibited very little change with minimal expansion into estuarine channels. Sources of material contributing to accretion at lower elevations that enable build-up of the wetland surface may include:

I. Mineral and organic matter from terrigenous sources that are transported to the estuary (or directly to wetlands) by overland flow are delivered to saline coastal wetlands by tidal flow and deposited when tidal flow diminishes.
II. Maintenance of lower elevation saline coastal wetlands may relate to 'roll-over' of the estuary morphology

Table 1 Estimated wetland area and carbon burried to 2100 in the lower Hunter estuary under two emissions and two management scenarios

	High emissions, floodgates open	High emissions, floodgates closed	Low emissions, floodgates open	Low emissions, floodgates closed
Wetland area (ha)	2,780	498	8,056	3,030
Carbon burial (tonnes)	775,075	490,280	602,035	389,349

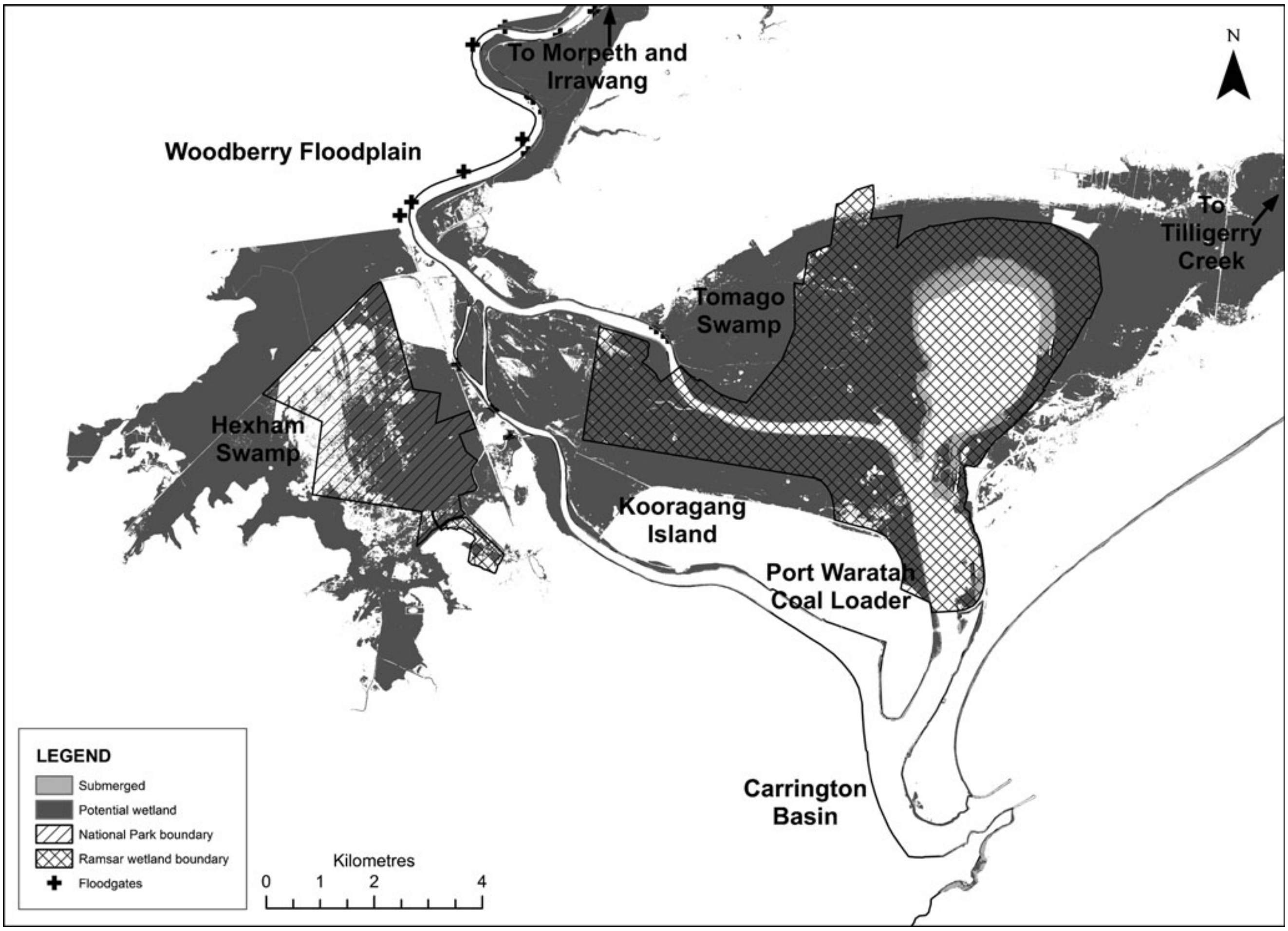

Fig. 2 Potential saline coastal wetlands areas at 2100 based on a low sea-level rise scenario (floodgates open). Saline coastal wetlands projection based on modelled hydrologically connected areas with an elevation between 0.1622 and 1.2015 m at 2100. The sea-level rise scenario employed in the model was based on a low sea-level rise projection derived from the 5th percentile minima sea-level rise projection under the IPCC B1 emissions scenario

upwards and landwards, similar to the model proposed by Allen (1990) for the Severn Estuary, UK and advocated by Pethick (2001). In this model, internal exchange and movement of fine sediment facilitates the removal of sediment from older deposits in the lower estuary to newer wetland areas so that wetlands maintain their position in the energy frame.

III. Wetland elevation gain within both mangrove and saltmarsh has been demonstrated to relate to expansion of the root zone and peat development (Bricker-Urso et al. 1989; McKee 2011; McKee et al. 2007; Nyman et al. 2006).

Our model results reflect the model applications of Kirwan et al. (2010) who demonstrated that marshes generally respond well to low rates of sea-level rise, but survival at rates exceeding 20 mm year^{-1} was dependent upon tidal range exceeding 3 m and tidal flows with high suspended sediment concentrations of 30–100 mg/L. Persistence of saline coastal wetlands at high rates of sea level-rise is not unprecedented; Woodroffe (1990) demonstrated that mangrove shorelines persisted at rates of sea-level rise in the order of 10–15 mm year^{-1} during the Holocene, and it has been demonstrated elsewhere that wetland accretion kept pace with sea-level rise over the Holocene (Alongi 2008; Fujimoto et al. 1996; Hashimoto et al. 2006; Krauss et al. 2003; Lynch et al. 1989; Miyagi et al. 1999).

The projected decline of potential saline coastal wetlands at higher rates of sea-level rise is substantial; with rates of sea-level rise exceeding the capacity of the wetlands to proportionally build elevation. This is consistent with the modelling of Traill et al. (2011) who demonstrated similar losses of seaward mangroves and landward expansion of wetlands. According to the high sea-level rise scenario the model projected extensive submergence of saline coastal wetlands, commencing around 2060 (Fig. 3) when rates of

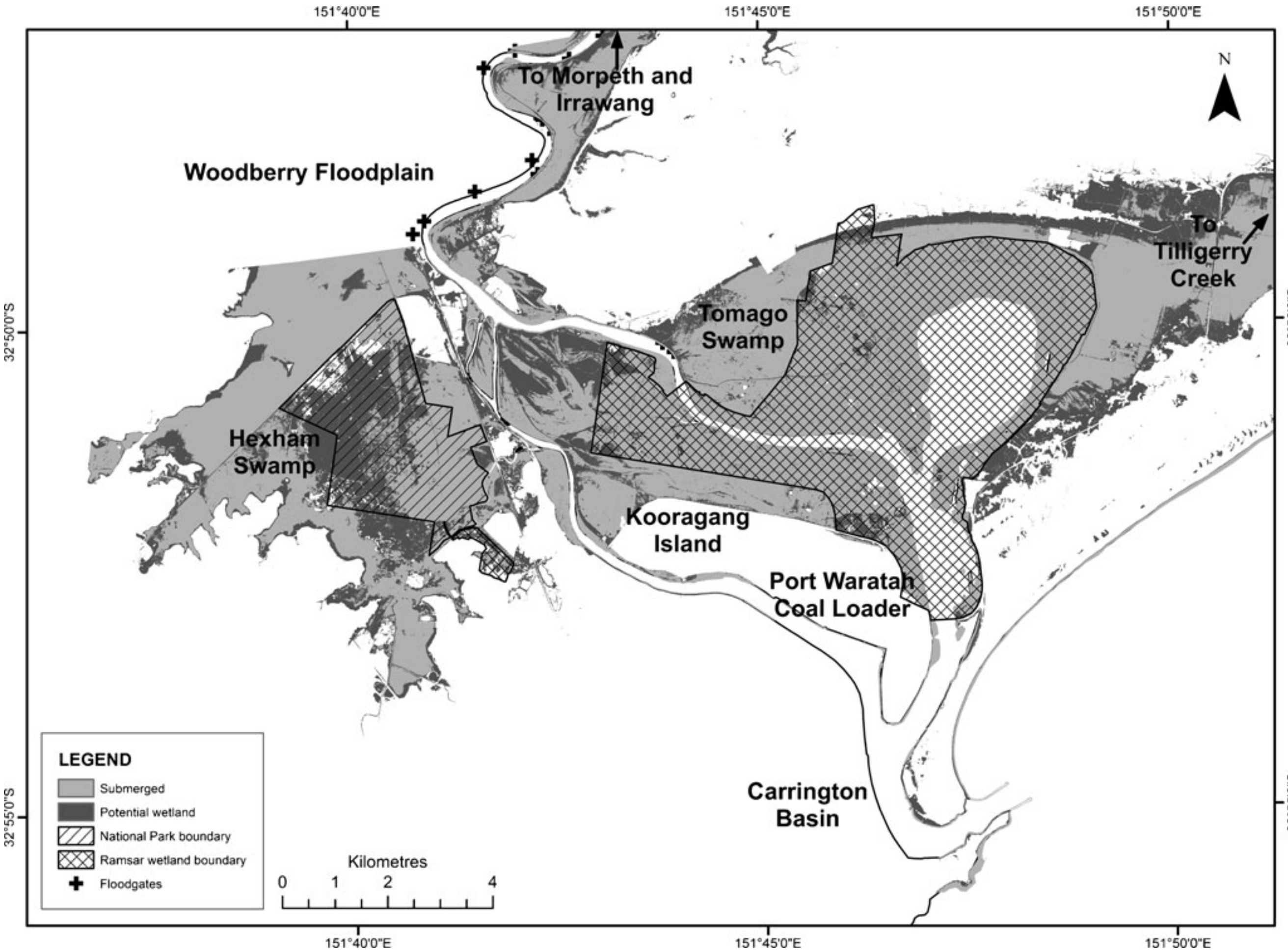

Fig. 3 Potential saline coastal wetlands areas at 2100 based on a high sea-level rise scenario (floodgates open). Saline coastal wetlands projection based on modelled hydrologically connected areas with an elevation between 0.1622 and 1.2015 m at 2100. The sea-level rise scenario employed in the model was based on a high sea-level rise projection derived from the 95th percentile maxima sea-level rise projection under the IPCC A1FI emissions scenario

sea-level rise began to exceed 9 mm year^{-1}. Based upon a high sea-level rise scenario our model demonstrates that when adaptation is facilitated by the reinstatement of tidal exchange, wetland retreat and elevation adjustment continues despite significant submergence under the high sea-level rise scenario. However, the wetland system may cross a resilience threshold by 2070 and rapidly transition to a new dynamic state of markedly less extent over a 30-year period unless adaptation measures are implemented. Though, we also recognize that the model presented here largely focusses on wetland geomorphology, and there may be some capability for biological responses to counteract geomorphological responses and inhibit (or slow) transition to new states. Upward adjustment of marsh surfaces facilitated by increased plant productivity and associated with increasing rates of relative sea-level rise have been demonstrated (Morris et al. 2002).

While our model application demonstrated the resilience of saline coastal wetlands, extended persistence beyond 2100 under high rates of sea-level rise may require significant mitigation measures to reduce the rate of sea-level rise and adaptation to reduce vulnerability. Active management of floodgate opening may be used to control tidal inundation and limit submergence in vulnerable areas located behind floodgates. Adaptation actions that manipulate accretion and wetland elevation building processes may be employed. Manipulation of elevation building processes through the addition of sediment (Ford et al. 1999) has been successfully trialled elsewhere. The success of nutrient addition in promoting plant growth and building wetland elevations is variable; McKee et al. (2007) observed increased root accumulation and elevation gain following nutrient addition of mangroves in the Caribbean region, while Lovelock et al. (2011) observed significant subsidence in nutrient-enriched mangroves and saltmarsh in Moreton Bay.

Accommodating wetland retreat has the additional benefit of carbon sequestration, as saline coastal wetlands are particularly efficient in the capture and retention of carbon (Alongi 2012; McLeod et al. 2011). The volume of carbon buried resulting from wetland accommodation of sea-level rise is a function of lateral extension and vertical accretion. Vertical accretion will be driven by geomorphic and biological responses to sea-level rise, at least until the point of submergence, while lateral extension will be a function of hinterland topography and opportunities for migration afforded by the presence or absence of hard structures. Our modelling, incorporating the carbon density data of Howe et al. (2009), show that the scenario with the greatest rate of carbon burial is the high sea-level rise scenario with all floodgates open (775,075 tonnes carbon buried in the lower Hunter to 2100: Table 1).

Implications for Wetland Management

This paper considers two management options: leaving all floodgates open, facilitating retreat of mangrove and saltmarsh into low-lying coastal lands; and leaving floodgates closed. We modelled a significant decline in wetland extent under the floodgates closed option, which would induce a significant decline in ecosystem service (Costanza et al. 1998). Under a floodgates open management option, our modelling indicated that wetlands may have some capacity to retreat without impediment. The gains in ecosystem service under this management option are likely to be substantial. In particular, carbon burial gains may be approximately 30 % greater under a floodgates open management option (Fig. 4).

Unlike many terrestrial carbon sinks, coastal wetlands are particularly efficient at sequestering carbon as mangrove and saltmarsh soils do not become carbon saturated and continue to accrete sediments over long periods of time; anoxic saline conditions slow down the rate of decay of organic material and inhibit the release of greenhouse gases; and saline soil conditions, particularly the presence of sulphates, minimize the release of methane to the atmosphere (Duarte et al. 2005; Magenheimer et al. 1996; McLeod et al. 2011). Carbon sequestration has been a recognized forest management strategy for some time (Silver et al. 2000), but has yet to be fully explored as a strategy for coastal wetland management. In a 'carbon economy', the economic cost of wetland retreat and conversion of grazing land to coastal wetland may be off-set by the economic gains achieved through carbon sequestration. In this case, the decision to keep floodgates closed would forgo approximately 210,000 and 280,000 tonnes of carbon burial under the low and high sea-level rise scenarios respectively, an opportunity cost of \$4–6 million in contemporary offset markets.

Legal and policy frameworks for wetland conservation globally are largely preservationist, grounded in the assumption that assets are stationary and can be defined in perpetuity as mappable areas. This approach is becoming increasingly discredited as a response to global environmental change (Craig 2010). Sea-level rise will diminish and in some cases obliterate the identity of relevant regulatory objects such as protected wetlands. The Hunter River case study described in this paper illustrates the limited capacity of both International (e.g. Ramsar Convention), National and State jurisdictional planning mechanisms to adequately protect critically important saline coastal wetlands in the face of climate change.

Under the high sea-level rise scenario, nearly all of the existing Ramsar wetland extent is lost by 2100. Presently, the

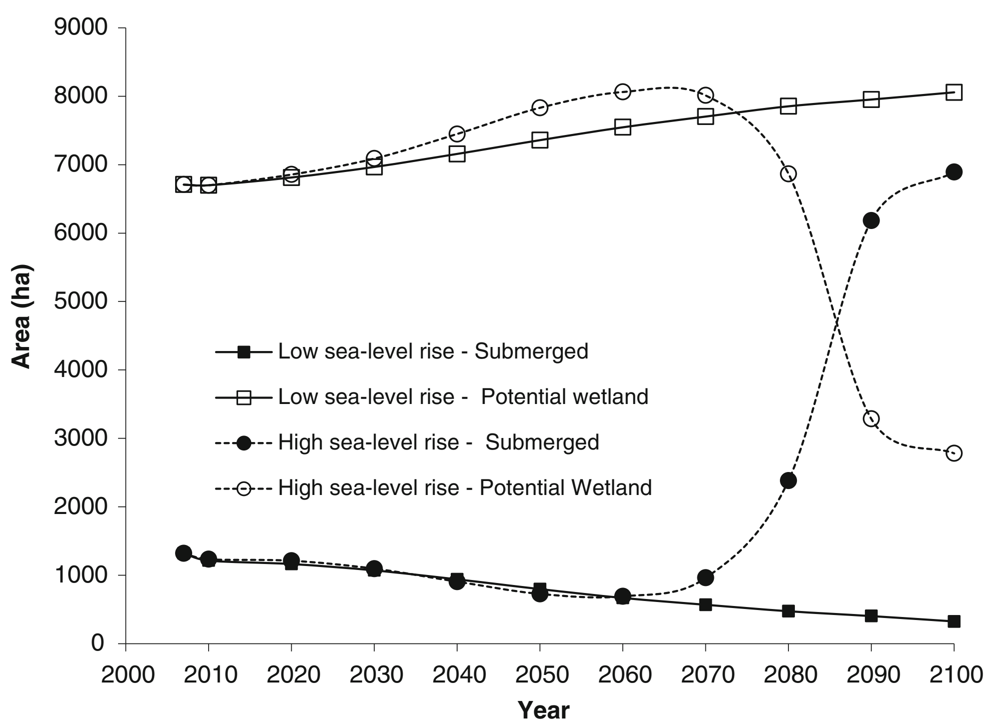

Fig. 4 Projected change in area of potential saline coastal wetlands and submerged area on the Hunter River between 2007 and 2100 based on a low and high sea-level rise scenario and all floodgates open. Submerged is defined as area below the elevation occupied by saline coastal wetland

Australian Government policy does not allow for notification to the Ramsar Convention under Article 3.2 of a change in ecological character where the principal cause is climate change (DEWHA 2009), even though Ramsar Resolution X.16 does not exclude climate change as a driver of altered ecological condition (Pittock et al. 2010). Opening of floodgates facilitates wetland retreat, but there is no mechanism in the Ramsar framework whereby the retreating wetland retains the status of international significance, a situation which progressively removes the wetland from Federal protection under the Federal *Environmental Protection and Biodiversity Conservation Act* (1999), which seeks to protect sites of international significance. The Ramsar Contracting Parties have not been able to provide clear guidance on how climate change should be dealt with under the Convention, though Resolution XI.14 from the recent COP 11 meeting (2012) recognizes the need to address the issue, referring the issue to the Scientific and Technical Review Panel for further consideration (Ramsar 2012).

In NSW (and elsewhere), State and local government planning instruments define the wetland as a mapped extent. The relevant maps were produced on the basis of aerial photography captured in the 1970s, and significant changes have already occurred since (Rogers and Saintilan 2002). There is currently no state level policy regarding buffers and accommodation space for coastal ecosystems impacted by sea-level rise. Hence, the primary level of planning control relevant to saline coastal wetland responses to sea-level rise in NSW is therefore via local government assessment of development applications.

We also identified that nearly all existing wetland within National Park boundaries is projected to be lost under the high sea-level rise scenario. In NSW, reserves are managed according to boundaries that have been gazetted. However, there may be some capacity to alter National Park boundaries to facilitate wetland adaptation to sea-level rise; relevant authorities (e.g. NSW National Parks and Wildlife Service) may acquire land for reservation or adjust the boundaries of existing reserves. The NSW National Park Establishment Plan (DECCW 2008) indicates that assessments of the adequacy of reserve systems within a bioregion should consider 'the likely impact of climate change on the resilience of the reserve system'. In NSW, the current statement of intent for reserve systems in the coast and coastal ranges bioregion of NSW indicates that 'fine-tuning and building-up of existing reserves' may include 'high priority intertidal, estuarine and lakebed areas which adjoin existing coastal and estuarine reserves' and 'areas that help buffer DECC public reserves from adjoining land uses that may be threatening reserve viability'.

In the Hunter region the bulk of the land that could be used to mitigate the impact of sea-level rise on intertidal wetlands is currently in private ownership. Elsewhere in NSW rehabilitation of saline coastal wetlands on private land has been achieved through either land purchase (e.g. Yarrahapinni and Hexham Wetlands) or through improved floodgate opening following landholder agreement. This latter method is only successful if there is no impact on landholders or more generally where the management change provides some benefits to the landholders through improved management of acid sulphate soils, increased wetland pasture or improved in-drain weed maintenance. Floodgate opening at sites in the lower Hunter River would result in damage to existing land uses and as such land purchase or other financial agreements would be the only possible course of action to facilitate wetland migration.

We make the following recommendations which should be broadly applicable to facilitating saline coastal wetland responses to sea-level rise nationally and internationally:

1. Saline coastal wetlands, and in particular those within a reserve system such as Ramsar or national parks, should be assisted in climate change resilience by the designation where possible of appropriate buffers for landward retreat. In the case of Ramsar wetlands, Ecological Character Descriptions should describe the wetland as a four-dimensional entity by recognising their latitudinal, longitudinal and elevation boundaries and the projected trajectory of these boundaries over time. Similarly, a comprehensive, adequate and representative public reserve system, such as national parks, should consider the trajectory of reserve boundaries over time.
2. The true cost of levee and floodgate maintenance be factored in cost–benefit analyses of coastal land protection, and that this includes the opportunity cost of wetland loss to carbon sequestration and ecological benefits, and the ongoing maintenance cost of the levee, including the accumulating liability for coastal flooding as sea levels rise.
3. The full range of adaptation pathways of saline coastal wetlands should be considered when promoting wetland resilience. Both mineral and organic sediment sequestration is essential for in situ adaptation of saline coastal wetlands. Consideration should be given to actions that facilitate in situ elevation adjustment of wetlands, such as enhancing the productivity of vegetation; or replenishment or maintenance of sediment supply.
4. To achieve these outcomes scientists and natural resource managers at all levels of government should work with individual wetland managers to identify likely wetland retreat pathways and potential management actions to facilitate wetland adaptation

Our analysis has exposed the limitations of current conservation and planning mechanisms in responding to the pressure of climate change in saline coastal wetlands, both in adapting to the reorientation of wetland boundaries or in facilitating mitigation through natural sequestration. Shoo et al. (2013)

propose the application of dynamic wetland boundaries and increased investment in reserves over time to maintain wetlands within the "preservation" network. This is a problem facing ecosystems more broadly, as Craig (2010) notes:

> "*climate change means that regulatory objectives based on pre-climate change characteristics of particular places can and will become obsolete. Climate change adaptation law must be able to accommodate this transforming ecological realities of particular places and not attempt to freeze ecosystems and their components into some prior state of being,"* p. 31

The modelling approach we have adopted provides the spatial output necessary for a considered approach to saline coastal wetland management, both in the presentation of flood risk, lands required for critical coastal processes and the likely benefit of carbon offsets associated with managed retreat. The implementation of coastal planning informed by these types of analysis will go some way to prevent the loss of opportunities for adaptive response and the wise management of wetlands throughout Australia.

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