Proof of concept for tidal re-connection as a blue carbon offset project

Sabine Dittmann, Luke Mosley, Michelle Clanahan, Jason Quinn, Steve Crooks, Igino Emmer, Sophie Min Thomson and Russell Seaman

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This project was undertaken in partnership with:

Enquires should be addressed to: Goyder Institute for Water Research
Level 4, 33 King William Street
Adelaide, SA 5000
tel: 08 8236 5200
e-mail: enquiries@goyderinstitute.org

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Executive summary

With salt field restoration opportunities arising in South Australia, this report provides an overview of the potential of this activity as a blue carbon project, following a Verified Carbon Standard method, based on a tidal re-connection trial. This proof of concept accompanies the technical report of the project *From Salt to C; carbon sequestration through ecological restoration at the Dry Creek Salt Field* by the Goyder Institute for Water Research. It is not intended to be a formal assessment of feasibility, nor has it sought to identify all the factors that would need to be considered in a detailed feasibility study.

The tidal re-connection trial had the combined objective of assessing a procedure that could avoid environmental impacts from acid sulfate soil mobilisation from the previous salt pond, and gain benefits from revegetation with saltmarsh and mangrove for carbon sequestration. Following the demonstration that tidal re-connection has no negative environmental effects and enables natural revegetation to occur, applying the re-connection to further salt ponds can have objectives of gaining carbon offset and additional co-benefits.

This proof of concept report is guided by criteria for project eligibility and carbon accounting based on the Verified Carbon Standard (VCS) Methodology VM0033 for tidal wetland and seagrass restoration, with the aim to support decision making and future design of a potentially larger scale blue carbon project. To achieve this aim, hypothetical scenarios of increasingly larger project areas were modelled, and permanence considered for each case under future sea level rise projections.

It was found that tidal re-connection of salt ponds meets the applicability conditions of the VM0033 methodology as the project activity is an eligible wetland restoration activity and does not cause leakage. The project activity of tidal re-connection also meets additionality requirements.

Project boundaries are described, with a temporal boundary set to a 30-year crediting time, and permanence to 2120. Geographic boundaries are not firmly set beyond the trial pond (scenario 1), as the further scenarios considered in this report are purely hypothetical. Strata were differentiated for the geographic boundaries based on elevation and predicted vegetation.

Carbon pools considered included soil carbon stocks for the 'business as usual' baseline and the 'with project' scenarios. Biomass carbon only applied to the 'with project' scenario. Where possible, data were used from field investigations in the 'Salt to C' project, but findings from further related research are also incorporated. Reasoning is provided for excluding further carbon pools. Typical complexity and cost burden of accounting for methane (CH_4) of wetland restoration projects elsewhere were found to be not an issue for tidal re-connection of salt ponds.

Greenhouse gas (GHG) emissions reductions and removals were quantified for the baseline and the trial pond, and estimated for purely hypothetical scenarios of re-connecting increasingly larger areas of salt ponds and their hinterland. For the 'business as usual' baseline, sea level rise (SLR) was not considered as levee banks have to be maintained above sea level. For SLR, the worst case of maximum SLR under the IPCC RCP8.5 scenario (rate of 6.8 mm/year by 2030 to 14 mm/year by 2090) was used given actual trends of sea level from Port Adelaide gauge and land subsidence.

It was found that the project will have a positive outcome in terms of GHG removals. Soil organic carbon stocks remained steady under baseline scenario, and increased significantly under the project scenarios. For 'with project' scenarios, the modelled carbon stock changes from soil and biomass show increases over the coming decades. Based on this assessment it was concluded that tidal re-connection is a viable blue carbon project under the VCS. The net GHG emissions reductions in 30 years crediting time could exceed 400 000 to 500 000 t CO_2e for hypothetical scenarios with further ponds re-connected, which could generate an estimated market value of $6–8 million with a carbon price of ca. $16. However, permanence can be affected

by sea level rise over a 100-year time period, unless the project area would be large enough to encompass landward retreat areas. The landscape slope and low development of the hinterland offers good potential for maintaining wetland resilience. Project planning, GHG accounting and permanence considerations will need to factor in whether to extend the site boundary landward beyond the current extent of marsh to account for future with sea level rise.

Tidal re-connection can be a recommended blue carbon project activity and meets criteria for tidal wetland restoration projects under the VCS, and also under Australia's integrity offset standard (Dittmann et al. 2019). It is thus recommended to move forward with a formal blue carbon project feasibility study and further project planning. The required further steps, such as financial and legal analyses, are described in the conclusions.

By demonstrating that introduction of tidal flow, as realised through tidal re-connection in the trial pond, can lead to carbon sequestration and reduction of GHG emissions, the trial has relevance for the potential wider application of this activity.

Acknowledgments

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The project location was part of the Dry Creek salt field, and we are very thankful to Buckland Dry Creek for providing access to the site for carrying out the investigations.

This proof of concept is building on the technical report for the Salt to C project, and also used data generated by the further project team, including Erick Bestland, Huade Guan, Gabriel Shepherd, Petra Marschner, Kieren Beaumont, James Stangoulis, Molly Whalen, Murray Townsend, Harpinder Sandhu, Beverly Clarke, Ryan Baring, Robert Costanza, Paul Sutton.

The final report benefitted from the reviews of Catherine Lovelock and Louisa Perrin.

1 Introduction

1.1 Background and objective

Re-introduction of tidal flow has been recommended as a prime activity to achieve blue carbon benefits and obtain carbon offset credits under Australia's Emissions Reduction Fund (ERF) (Kelleway et al. 2017). Tidal re-connection of ponded wetlands has been shown to increase carbon sequestration and lower greenhouse gas (GHG) emissions, as demonstrated for ponded freshwater wetlands (Krauss et al. 2017; Kroeger et al. 2017), or managed realignment of saltmarsh (Burden et al. 2013; Wollenberg et al. 2018). Carbon sequestration from re-connected salt ponds has been less studied, and mostly in regions without mangrove and where saltmarsh is dominated by marsh grass (*Spartina* spp.). With salt field restoration opportunities arising in South Australia, this report provides an overview on the feasibility of this activity as a carbon offset project, based on a tidal re-connection trial and further associated studies.

Despite the benefits to mitigate global warming from tidal re-connection or other blue carbon project activities, Australia has, at present, no method under the ERF for blue carbon projects. Carbon credits from the voluntary carbon market under international standards such as the Verified Carbon Standard (VCS) cannot be used in Australia due to potential issues with double counting of carbon abatement. In the absence of a methodology for the ERF, the project *From salt to C; carbon sequestration through ecological restoration at the Dry Creek salt field* (Salt to C) by the Goyder Institute for Water Research followed general principles of the VCS method VM0033 for tidal wetland and seagrass restoration (Emmer et al. 2015). This proof of concept accompanies the Salt to C project technical report (Dittmann et al. 2019) and can support the further design of a blue carbon project.

The overall aim of the tidal re-connection trial of an abandoned salt pond in the Dry Creek salt field, South Australia, was to assess whether carbon credits and co-benefits can be gained from tidal re-connection and coastal wetland restoration of salt fields. The project was not intended to be a formal assessment of feasibility but an assessment of whether this approach has merit for further consideration. The project has not sought to identify all the factors that would need to be considered in a feasibility study, such as land use and costs._In order to inform decisions for a possible grouped project (Emmer et al. 2015), further parts of the Dry Creek salt field, and the samphire coast area north of Adelaide were also considered in the project.

Outcomes from this assessment will inform decisions on future opportunities for salt field restoration, and the development of a blue carbon methodology in Australia. By demonstrating that introduction of tidal flow, as realised through tidal re-connection in the trial pond, can lead to carbon sequestration and reduction of GHG emissions, the trial has relevance for the potential wider application of this activity recommended by the technical review of blue carbon opportunities in Australia's ERF (Kelleway et al. 2017). The findings can further identify opportunities for how climate change adaptation goals could be met. Together with further co-benefits such as increases in biodiversity, provisioning ecosystem services and human well-being, coastal restoration through tidal re-connection can help meet the United Nations Sustainable Development Goals.

1.2 Background of VM0033 methodology

The VCS offers standards for a verifiable and accountable approach to issue carbon credits (Needelman et al. 2018, 2019), and is now managed by Verra. Standards developed by Verra are widely used in the voluntary carbon offset market (Needelman et al. 2018). Carbon projects can only be undertaken under voluntary schemes if there is a mechanism avoiding double-counting. The issue arises where carbon abatement is counted under a voluntary scheme (and carbon credits created and sold), and also reported under Australia's National Greenhouse Gas Inventory (NGGI). However, the applicability of the VM0033 methodology in the Australian context is being assessed by the Commonwealth Government for use as a potential ERF blue carbon method.

The VCS methodology VM0033 for tidal wetland and seagrass restoration is based on principles and concepts of the Clean Development Mechanisms (CDM), in particular the Afforestation/Restoration tool AR-Tool 14 for the estimation of carbon stocks and change in carbon stocks of trees and shrubs. The CDM was the main approach for emissions reduction under the Kyoto Protocol, which will be revised as part of the transition to the Paris Agreement under the United Nations Framework Convention on Climate Change (UNFCCC), to be implemented from 2020 (Herr et al. 2019).

The types of restoration activities considered in VM0033 include restoring hydrological conditions; improving water quality or re-introducing native vegetation, which lead to measurable reductions in GHG emissions or removals through increased biomass (Emmer et al. 2015; Needelman et al. 2018). Following principles of the CDM, the net benefit for carbon offset is obtained from the difference between the 'with project activity' carbon stock or GHG emission, subtracted by carbon stocks or estimated emissions under 'business as usual' (Needelman et al. 2019). Values used in calculations can be measured, be derived from published studies or modelling where justified, or follow default values, if applicable (Needelman et al. 2018). Assessments should be conservative and include uncertainty estimates.

1.3 Approach

This proof of concept report is guided by VM0033, but as not all aspects of it are applicable in the Australian context, the methodology is not strictly used. For example, we refer to tidal re-connection as an activity, while VM0033 requires a project method approach according to the CDM for all projects outside of the USA. Where of interest in the Australian context, we also refer to other methods or global guidelines. We did not follow the procedure for uncertainty estimates of the VM0033, but standard statistical procedures and the coastal blue carbon method manual (Howard et al. 2014).

For this project, we used field-collected data from the tidal trial and associated studies to measure and estimate GHG emissions for carbon accounting. We focused on the scientific outcomes and data needed for a pre-feasibility assessment: legal, finance and stakeholder aspects are not covered. This proof of concept report does not intend to be a full blue carbon project description and is limited by the scale and scope of the Salt to C project.

2 Applicability

2.1 Eligibility

Tidal re-connection of salt ponds falls under recognised project activities of the VCS standard, with the baseline being an 'open water' state of salt field operation (i.e. permanent inundation of an impounded hypersaline water body enclosed by levy banks), and the project activity the 'creation or restoration of conditions for afforestation, reforestation, or revegetation' (Emmer et al. 2015) (Figure 1).

The re-introduction of tidal flow is a tidal wetland restoration activity in alignment with the VM0033 methodology. In particular, the tidal re-connection project is meeting the following restoration project activities defined by the VM033 methodology (see Emmer et al. 2015):

- *Creating, restoring and/or managing hydrological conditions (e.g. removing tidal barriers, improving hydrological connectivity, restoring tidal flow to wetlands or lowering water levels on impounded wetlands)*
  The installation of a controllable tidal gate into a levee bank separating a salt pond from a mangrove creek restored natural hydrological conditions with regular tidal flooding and draining of the pond (Mosley et al. 2018). Restoration of these natural environmental processes enabled the tidal wetland restoration (Figure 2).
- *Changing salinity characteristics (e.g. restoring tidal flow to tidally-restricted areas)*
  Following tidal re-connection, the salinity in the previously hypersaline pond was lowered and became similar to ambient salinities in coastal wetlands of Gulf St Vincent (Mosley et al. 2018).
- *(Re-)Introducing native plant communities (e.g. reseeding or replanting)*.
  Tidal re-connection enabled seeds from saltmarsh and mangrove propagules entering the trial pond project area. The revegetation occurred through natural seed dispersal, and the pioneer saltmarsh establishing inside the pond became quickly mature, adding a local seed supply (chapter 4, Dittmann et al. 2019).

The tidal re-connection activity also includes aspects of the activity:

- *Improving water quality (e.g. reducing nutrient loads leading to improved water clarity to expand seagrass meadows, recovering tidal and other hydrologic flushing and exchange, or reducing nutrient residence time)*.
  Following the tidal re-connection, the natural tidal flushing improved environmental conditions in the pond, and did not lead to any adverse effects. It was thus also shown to be a suitable approach to mitigate acid sulfate soil accumulations in the ponded salt field (Mosley et al. 2018).

It is therefore concluded that the project activity of tidal reconnection of salt ponds would be an eligible activity under VM0033.

Under the 2006 IPCC Guidelines and the Wetlands Supplement (IPCC 2014), salt fields fall under 'managed' or 'constructed' wetlands. While the starting condition is not a drained wetland, it has an artificially raised water level and so the activity of tidal re-connection could be seen to align with 'rewetting, revegetation and creation'. Revegetation occurs through natural recolonisation after restoring natural tidal inundation (Dittmann et al., 2019).

Figure 1. The trial pond XB8A of the Dry Creek salt field north of Adelaide, showing adjacent ponds with hypersaline water still operated under a holding pattern, and the mangrove/saltmarsh coastline on the left. The tidal gate installed as project activity can be seen where the creek passes through the levy bank. Photo taken at low tide several months after re-connection of the pond. Photo credit: Department of Environment and Water.

Figure 2. Natural revegetation by saltmarsh inside the trial pond, ca. 1.5 years after re-connection.

2.2 Applicability conditions related to leakage

VM0033 can only be applied to projects which are not causing leakage, i.e. additional GHG emissions outside of the project area.

Prior to the project start date, the impounded salt pond had not been used for commercial purposes for several years and the entire Dry Creek salt field was free of commercial land use. There is thus no land-use displaced because of the project. As the project is not causing any activity shifting, it is concluded that no leakage arises.

3 Project boundary

The project boundaries defined here are the temporal and geographic boundary. The project boundary is currently kept broad at landscape scale and requires further consideration, subject to restoration and conservation options in the future, in particular for a potential grouped project.

3.1 Temporal boundaries

Temporal boundaries address crediting period, project longevity, permanence, and the peat or soil organic matter depletion time (PDT or SDT). The crediting period is set at a minimum of 20 years by the VCS, but as a longevity of at least 30 years is needed to be covered against non-permanence risk, the crediting period is usually set at 30 years (Emmer et al. 2015). These timeframes are comparable with the 25-year period under Australia's ERF.

The recommended crediting time for tidal re-connection of salt ponds is 30 years, in consideration of the mangrove colonisation time from a nearby chronosequence (Clanahan 2019). The crediting period can be renewed several times, but not exceed 100 years. The permanence timeframe under the VCS is 100 years. We used the RCP 8.5 from the IPCC for sea level rise to assess how key habitats may move during the project period and to assess permanence of a project.

The approach for eligible GHG emissions reductions from soils to be used for this project activity is the 'total stock approach', as the difference between the soil organic carbon stock in the 'with-project' and 'baseline' scenarios after 100 years. The assessment was made *ex ante* based on data obtained from the tidal trial and associated projects (Dittmann et al. 2019; Mosley et al. 2018, Clanahan 2019).

Soil organic matter depletion time (SDT) was not considered for this project, as the project is not seeking to claim 'avoided losses' from reduction in baseline GHG emissions. Also, no loss of soil organic carbon was measured due to oxidation under the business as usual (salt pond) scenario (Mosley et al. 2018), and thus no on-going emissions can be assumed for the baseline. The data from the tidal trial also show that after re-connection, no net carbon loss occurred from soil organic carbon oxidation, as a net sequestration of carbon was detected.

3.2 Geographic boundaries

3.2.1 GENERAL SPATIAL BOUNDARIES

For this proof of concept, the geographic project boundary is not firmly set at this stage. A tentative area for the geographic boundary is indicated in Figure 3, but no coordinates are given as the area to be included is subject to further stakeholder consultation and clarification on right of use. The regulatory regime under the *Mining Act 1971* for the current holding pattern of the salt field requires that levee banks are maintained to keep the sea out, implying they have to be increased in height to stay above the rising sea level.

For this report, a sufficiently large area was considered to allow possible landward retreat under sea level rise, and the landward progression of seagrass beds that are seaward of the mangroves. The flat terrain of the northern Adelaide plains is favourable for enabling a gradual transition of coastal blue carbon ecosystems with sea level rise, as long as the landward retreat space is maintained and no coastal squeeze arises. Sea level rise is further considered throughout chapter 5. Leakage and co-benefits were also considered for a wider landscape.

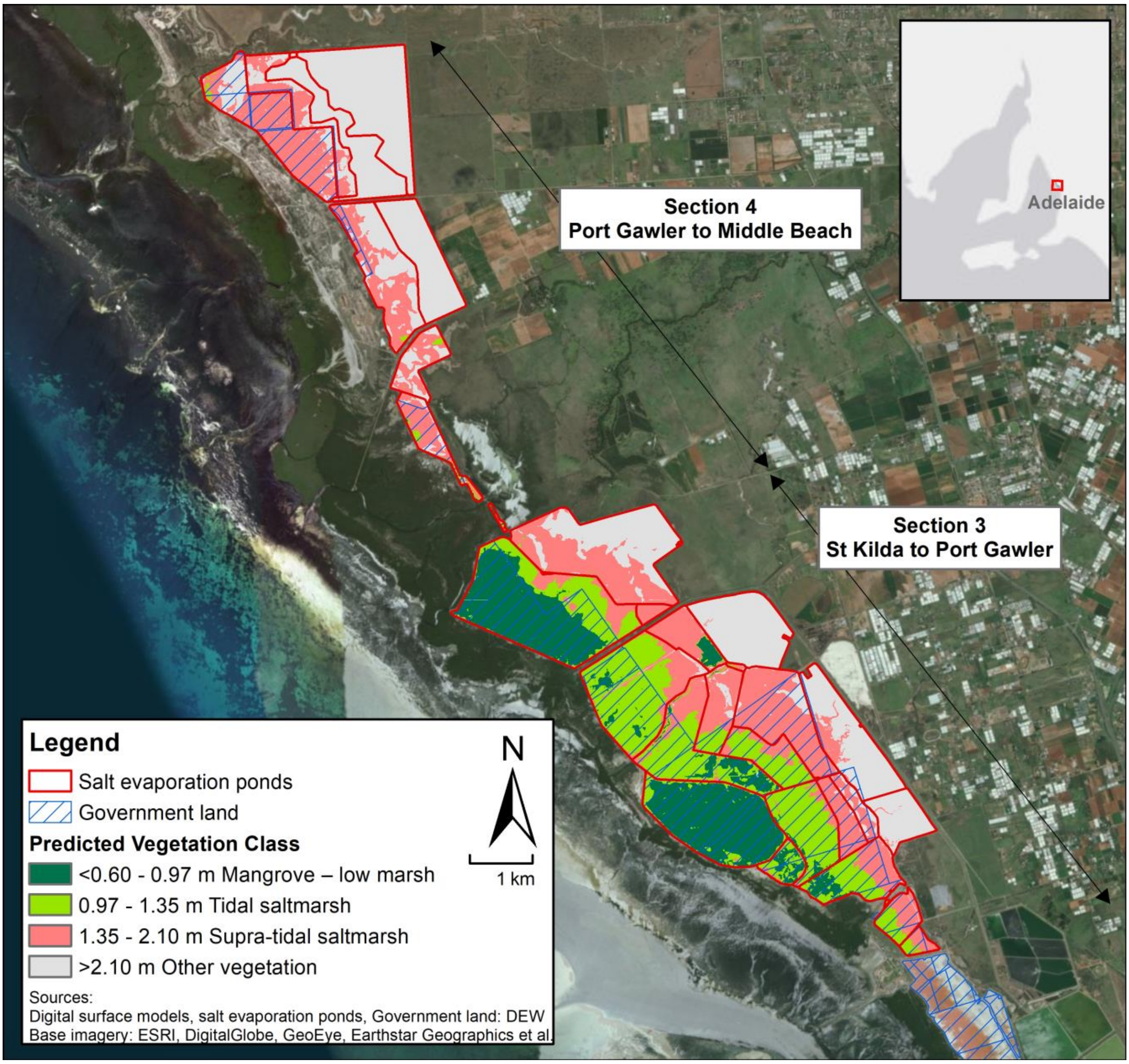

Figure 3. Map of the possible project area in the northern section of the Dry Creek salt field, considered for possible tidal re-connection, and under sea level rise scenarios for permanence considerations. The map also shows the three strata ('Mangrove-low marsh' elevation 0.60 -0.97 m. 'Tidal saltmarsh' elevation 0.97–13.5 m, 'Supra-tidal saltmarsh' 1.35–2.10 m) based on bathymetry estimates of submerged ponds. Crown land is outlined in blue lines.

Within this geographic boundary, the Salt to C project (Dittmann et al. 2019) investigated carbon stock changes following tidal re-connection in a ~31.5 ha pond of the Dry Creek salt field north of Adelaide, South Australia (Figure 4). While the trial is covering a fairly localised area only, it functions as an example and provided measurement data for this proof of concept. A grouped project could be envisaged to achieve a more economically attractive scale for a blue carbon project, and be possible if further salt ponds become available for tidal re-connection and wetland restoration. Grouped projects allow for flexibility with the integration of further areas if exact details of their demarcation or contractual arrangements are not yet known, as long as the intention to expand the project area is laid out in the initial validation (Emmer et al. 2015). Hypothetical scenarios carried out in the Salt to C technical report (Dittmann et al. 2019) revealed an increased potential of blue carbon benefits from re-connecting larger areas of salt ponds.

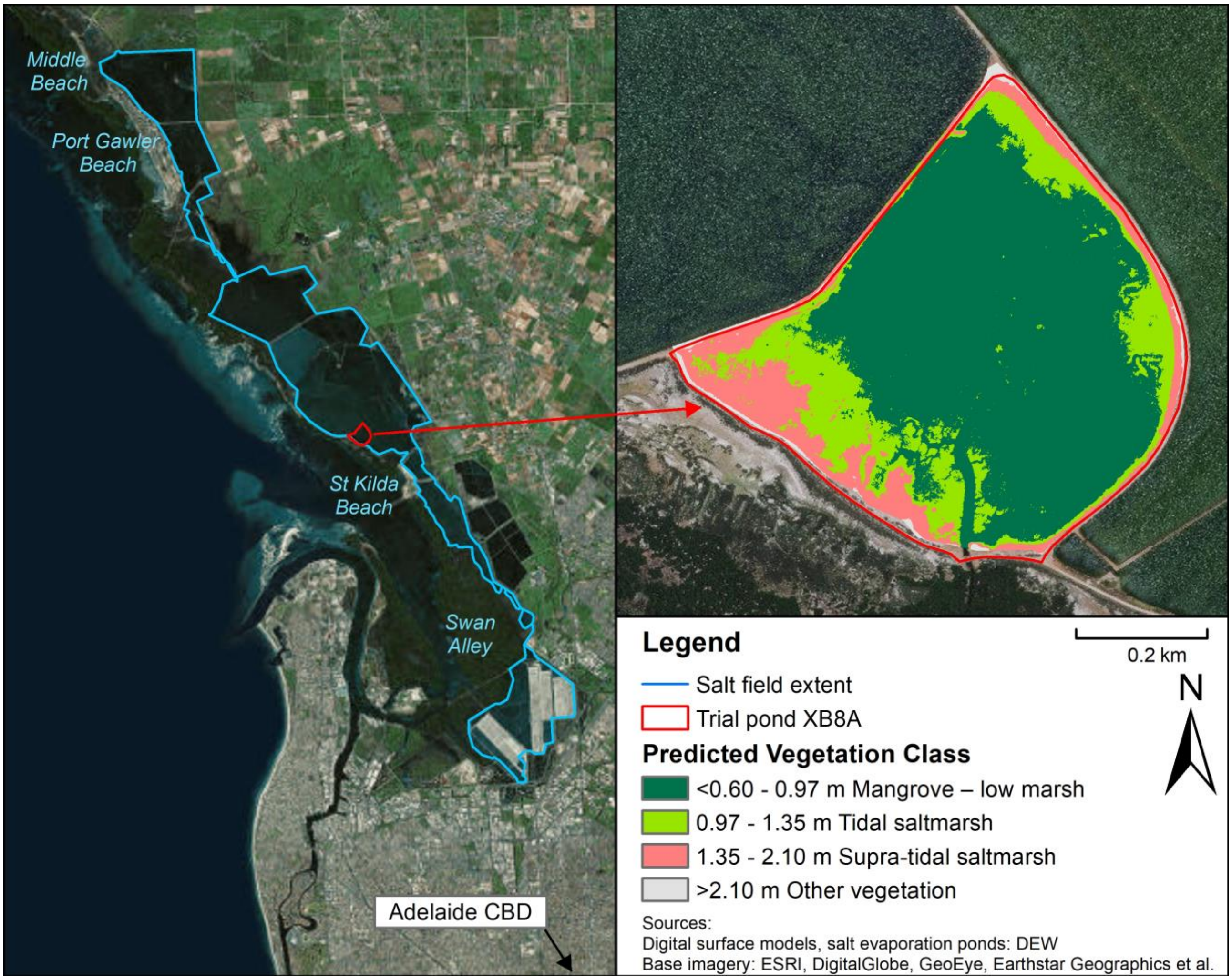

Figure 4. Location of the Dry Creek salt field north of Adelaide, South Australia, and the trial pond, XB8A.

3.2.2 STRATIFICATION AND SCENARIOS

The geographic boundary captures a wide intertidal terrain gradient, and several strata were differentiated. The project area was sub-divided into relatively homogenous zones, following approaches outlined in the VM0033, Emmer et al. (2015) and Fourqurean et al. (2014). We adopted a stratification by elevation levels and predicted vegetation cover classes, from mangrove and tidal salt marsh to supra-tidal salt marsh. Strata were based on elevation available from digital elevation maps (DEMs) and previous vegetation mapping in the samphire coast region with accurate elevation measurements (Herpich et al. 2017; Fotheringham 2016) (see Dittmann et al. 2019 for details).

Data on carbon pools were gathered for each of three elevation levels characterised by particular associated plant communities, which then allowed carbon sequestration estimation and modelling for scenarios of increased project areas for a possible grouped project (Table 1). The baseline scenario is where ponds continue to be operated as a salt field or are kept flooded with hypersaline water after operation as a salt field ceases. The purely hypothetical scenarios (see chapter 5) include:

> *Scenario 1* – Trial pond re-connected: under this scenario the trial pond XB8A stays re-connected and the development is modelled into the future.

Scenario 2 – Low-lying ponds re-connected: under this scenario adjacent salt ponds, especially those with a single levee bank on the seaward side that could be opened up for tidal re-connection, were included in the assessment. These low-lying ponds mostly overlap with area that is crown land.

Scenario 3 – Higher-lying ponds re-connected: this scenario comprised the ponds from scenario 1 and 2 and added additional salt ponds located in the upper intertidal, and ponds of the salt field area further north near Middle Beach.

Scenario 4 – Sea level rise retreat: an extended project area was considered for this scenario, to evaluate the retreat options as habitats shift inland. The elevation of the area included in this scenario is currently above supratidal.

A map with the area and delineation of strata in the Dry Creek salt field for scenarios 1 to 3 is shown in Figure 5. The finer resolution of strata indicated on the map allowed better estimates of habitat transition under sea level rise, in particular for seagrass beds moving inland.

Table 1: Elevation range of the main strata differentiated in the project area, and the spatial extent of each stratum at the scale of the trial pond, and possible additional ponds as considered in scenario 2 and 3 (based on Dittmann et al. (2019)). In each case, project areas for stratification were based on elevation and predicted vegetation classes (Herpich et al. 2017).

STRATA	ELEVATION RANGE (m AHD)		AREA (ha)		
	LOWER BOUNDARY	UPPER BOUNDARY	SCENARIO 1	SCENARIO 2	SCENARIO 3
Mangrove – low marsh (MLM)	<0.6	0.97	20	473	571
Tidal saltmarsh (TSM)	0.97	1.35	6	532	689
Supra-tidal saltmarsh (SSM)	1.35	2.1	4	205	703

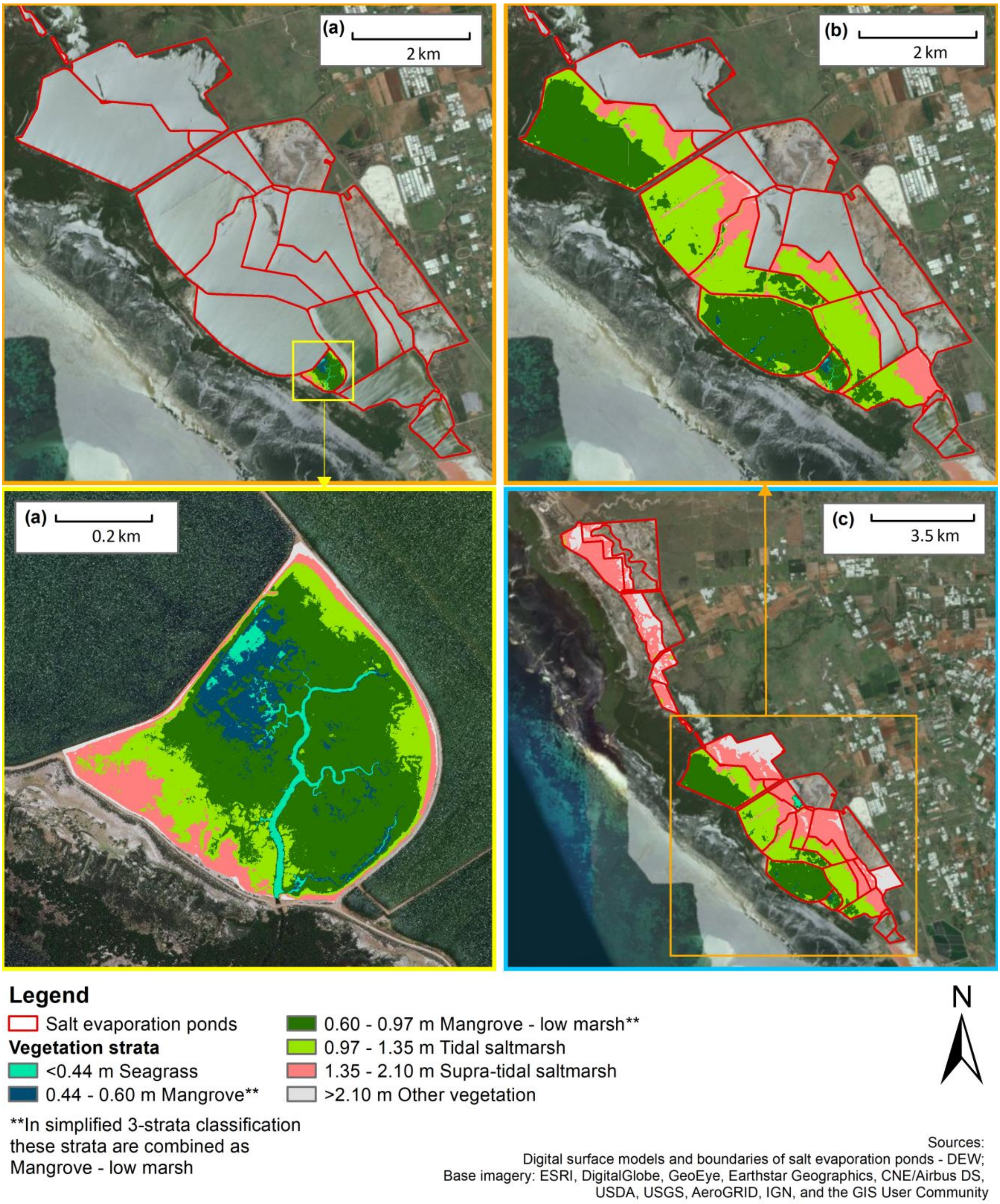

Figure 5. Maps of the samphire coast north of Adelaide outlining the boundaries and extent of vegetation strata considered under the scenarios: (a) scenario 1 – the trial pond stays re-connected; (b) scenario 2 – additional low-lying ponds are re-connected to tidal flow; and (c) scenario 3 – higher-lying ponds are re-connected in addition to those opened up under scenario 1 and 2.

3.3 Carbon pools

Under the baseline scenario, the only carbon stock included is soil carbon, as no vegetation is present within ponds operated as salt fields. For project scenarios, two carbon pools were included: changes in carbon stocks of soil and biomass.

For the soil organic carbon pool, carbon stock changes were measured repeatedly in the trial pond (project activity) over a period of 1.5 years after re-connection occurred. For the business as usual baseline, soil organic carbon was measured over time from sediments in an existing salt pond still flooded with hypersaline water (Mosley et al. 2018). Soil organic carbon data used in calculations were to 30 cm depth, which was the shallowest depth across the various cores taken from trial and reference sites, although some cores were to at least 40 cm depth. In many cores from saltmarsh and also mangrove strata, peat and shell grit layers occurred at ~40 cm depths. Similar peat layers at nearby sites had previously been dated to be about 1500 years old (Dittmann et al. 2016), and reflect the Holocene evolution of the coastline (Belperio et al. 1995). Sediment layers beneath the peat had been aged to 6440 ±90 years by Belperio et al. (1995). Soils in the project area are mineral soils, as peat occurs only in a narrow defined band at greater sediment depth, and centuries of relatively low and slow mineral sediment accumulation (1.5–2.2 mm yr^{-1}) at reference sites (chapter 3.3, Dittmann et al. 2019) have occurred after the time of the peat deposition.

For the biomass carbon pool, above and below ground carbon from saltmarsh and mangrove was included. Saltmarsh vegetation was the first coloniser and established quickly within the trial pond (Dittmann et al. 2019). Mangrove did not successfully establish within the trial pond during the Salt to C project timeframe, but was expected to colonise and encroach into saltmarsh in the future, based on a nearby chronosequence (Clanahan et al. 2019). The chronosequence spanned nearly nine decades following breaches of levy banks at Swan Alley in 1935, in an area which had been impounded for about 40 years (Burton 1982; Harbison 2008). In the absence of other data on decadal scale of tidal wetland vegetation changes, the colonisation rates and mangrove forest growth pattern from the Swan Alley chronosequence were used for modelling permanence and sea level rise effects, although Swan Alley had not been operated as a salt pond like the project area.

Seagrass wrack deposition in saltmarsh and mangrove is a natural and common process in coastal ecosystems of southern Australia. Litter accumulations occurred in reference areas and inside the trial pond, and were mainly comprised of seagrass detritus. In particulate organic matter form, seagrass litter was not included in the carbon pool, as it came mostly from outside the project area (allochthonous). Yet, carbon from the decomposition of seagrass wrack within the trial pond was considered autochthonous, as it decomposed on site and became part of soil organic carbon by natural processes inside the project area.

The carbon stock of seagrass beds outside the pond was not determined in this project, but when future sea level rise and landward migration of coastal habitats were considered, seagrass colonisation of lower lying areas of re-connected salt ponds was included, using regional values (Lavery et al. 2019).

Greenhouse gas flux was measured directly in the project, from the trial pond, a control pond (business as usual), as well as a nearby reference area (Dittmann et al. 2019). The measurements showed that emissions of CH_4 and N_2O were negligible. In particular, the very low methane emissions were consistent with studies elsewhere, which illustrate that at salinities >18–20 ppt, methane emissions are typically very low (due to sulfate availability) (Poffenbarger et al. 2011; Negandhi et al. 2019). Conditions were still hypersaline inside the trial pond, especially in sediments (Mosley et al. 2018), and salinities are in general high in Gulf St Vincent (>36 ppt, Bye & Kaempf 2008). Emissions of CH_4 and N_2O were thus excluded as *de minimis* from baseline and project scenarios. CO_2 does not need to be measured with the total stock change approach, as CO_2 emissions will be accounted for within any observed changes in the soil and biomass carbon pools.

During construction of the tidal gate for the re-connection, vehicle and machinery use would have created emissions from fossil fuel use. This construction period was completed within several weeks, and number of vehicles, distances and fuel type not recorded. This fossil fuel use due to the project activity can thus not be determined, but is considered to be low in relation to other carbon pools over the project duration. Energy need for any ongoing operation of the tidal gate is provided by solar power.

4 Additionality

A project needs to document that it is not business as usual and would not occur without finance from carbon offset generation (Emmer et al. 2015.). Without the tidal re-connection project, the scenario for the pond would be a continued submergence by hypersaline water and possibly resumption of salt production in the salt field. The area is still under mining lease, but currently in a holding pattern, until final closure in compliance with regulatory requirements is met. Potential carbon offset credits demonstrated from the tidal trial will inform decision-making by stakeholders (owner of salt field, state government), as these credits would provide a financial incentive to re-connect additional ponds in the salt field to tidal flow. The envisaged project activity to connect a salt field to tidal flow is not common practice, and has not been undertaken anywhere in Australia. The activity of restoring coastal wetland through tidal re-connection of salt ponds is thus considered to meet additionality requirements.

Regarding a regulatory surplus test, i.e. whether the project activity is not mandatory under any regulatory requirements, further legal assessment is needed to clarify whether tidal re-connection of the salt field could be a project activity required by law, in which case it would not meet additionality requirements. For the Dry Creek salt field, salt mining operations fell under the *South Australia Mining Act 1971*, which requires restoration of land disturbed by mining operations to the satisfaction of the Minister (section 92 (i) Regulations). The regulatory framework does not specify whether this restoration has to be to the previous habitat condition, nor how it is achieved. It can thus be argued that the particular project activity of tidal re-connection of salt ponds is beyond the required restoration under the *South Australia Mining Act 1971*, and thus additional.

The accompanying technical report (Dittmann et al. 2019) includes an assessment of tidal re-connection of salt ponds under Australia's ERF, and demonstrates that this activity can meet the criteria and formal requirements in the Australian context.

The tidal re-connection trial had the combined objective of assessing a procedure that could avoid environmental impacts from acid sulfate soil mobilisation from the previous salt pond, and gain benefits from revegetation with saltmarsh and mangrove for carbon sequestration. Following the demonstration that tidal re-connection has no negative environmental effects (Mosley et al. 2018) and enables natural revegetation to occur (Dittmann et al. 2019), applying the re-connection to further salt ponds can have the objectives of gaining carbon offset and additional co-benefits.

5 Quantification of greenhouse gas emission reductions and removals

Greenhouse gas emissions of the project were quantified for the defined carbon pools (section 3.3) in the three differentiated strata (section 3.2.2). No strata could be differentiated for sampling in the control salt pond (business as usual baseline), which was still filled with hypersaline water.

Carbon stock changes are used to estimate GHG emissions reductions and removals for the baseline and project activity. Changes over time are based on direct measurements of fluxes and stocks in the various strata of the project area. Some of the direct measurements showed a high variability across sites and over repeated surveys, both within the trial pond and reference area. The study time frame was 1.5 years after re-connection. To project to longer timeframes, additional data were used from a chronosequence (see section 3.1) and long-term sediment accumulation rates (Dittmann et al. 2019). These data provided knowledge of natural plant succession and historic trends in soil carbon developments, which informed the models used (sections 5.2, 5.3).

Quantifications for GHG emissions reductions and removals under baseline and project scenarios were made for four scenarios (see section 3.2.2, Table 1 and Dittmann et al. 2019). The salt pond site is currently under a mining lease with a mix of government and privately-owned land, hence the scenarios described are considered purely hypothetical for potential consideration, should these options be pursued in the future.

The location of levy banks, which are forming the outside and the inner boundaries of ponds in the salt field, were mapped for scenario 1 to 3 (Figure 5). The available bathymetry of flooded ponds revealed the location of submerged creeks, which are the most suitable locations for further re-connection.

We calculated GHG emissions reductions and removals for a crediting period of 30 years and permanence of 100 years, with consideration of sea level rise as outlined in section 5.1. We considered location specific variables of the strata as they shift with rising sea level, being informed by elevation and flooding ranges for major vegetation types (Quinn 2017).

5.1 Sea level rise

The project area and its geographic boundaries will be affected by sea level rise (SLR), which was 1.5–4 mm per year between 1965 and 2016 along the South Australian coast (Department for Environment and Water 2018). Regional models for Australia show that the median SLR for Port Adelaide is projected to be about 21–25 cm by 2050 compared to the average between 1986–2005, and 39–61 cm by 2090 (for the IPCC scenarios RCP2.6–RCP8.5 respectively[1]), but could be higher if the Antarctic ice sheet should collapse (CSIRO and Bureau of Meteorology 2015; McInnes et al. 2015) (Figure 6). Localised land subsidence in the Port Adelaide region, due to human activities such as land reclamation and groundwater extraction, could increase the relative SLR (Belperio 1993, McInnes et al. 2015). We thus selected the worst case SLR scenario based on the IPCC.

[1] RCP=Representative Concentration Pathways

We modelled effects of the maximum sea-level rise scenario from RCP8.5 by the IPCC from 2020 on the project area for ten decades up to 2120. Beyond 2090, uncertainty of forecast models increases, given the accelerated rate of change in current climate trajectories.

How relative SLR will affect coastal wetlands and their carbon sequestration requires knowledge of the accommodation space and sediment supply (Rogers et al. 2019; Woodroffe et al. 2016). For the Port Adelaide region and samphire coast, little information exists about sediment dynamics and supply is limited as riverine influx reduced following urban and industrial developments. Sediment accumulation rates were highly variable at a local scale (Dittmann et al. 2016; Dittmann et al. 2019; Lavery et al. 2019).

The flat terrain of the northern Adelaide plains is favourable for enabling a gradual transition of coastal blue carbon ecosystems with sea level rise, as long as the landward retreat space is maintained and no coastal squeeze arises. We allowed for landward retreat beyond the Dry Creek salt field in scenario 4. With sea level rise, the strata can geographically shift inland over the project duration, whereby carbon stocks in soil and biomass are expected to remain intact.

Spatial models were created for the trial salt pond and adjacent area, as well as for the wider landscape north of Adelaide, to assess coastal rollover (i.e. the landward migration of coastal wetlands with SLR and erosion of the seaward margin).

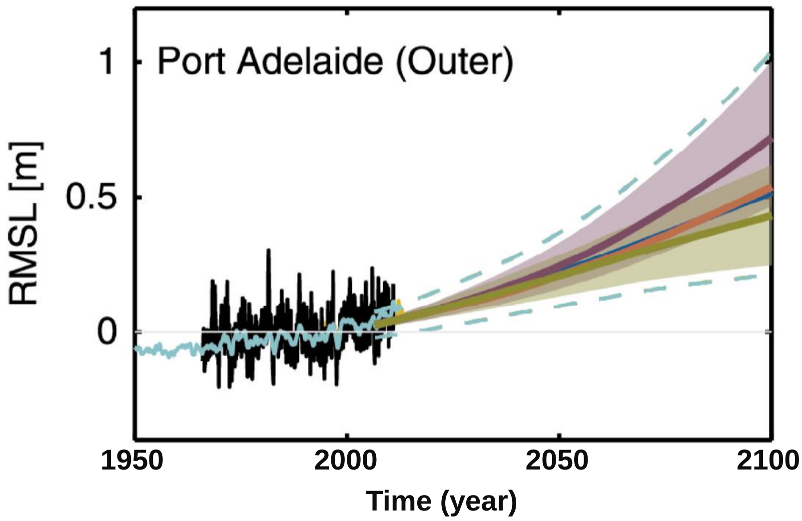

Figure 6. Relative sea level changes for Port Adelaide with observed records (black lines) and projected rises based on modelled scenarios for RCP8.5 (purple), RCP6.0 (orange), RCP4.5 (blue) and RCP2.6 (olive), with uncertainty ranges for RCP8.5 and RCP2.6 shown in purple and olive shading. Figure based on McInnes et al. 2015.

5.2 Baseline emissions

The baseline scenario refers to the conditions in ponds operated as a salt field or kept flooded with hypersaline water after operation as a salt field ceased. Emissions in the baseline scenario are only soil carbon changes, as no saltmarsh, mangrove or seagrass vegetation occurs. Sea level rise was not considered for the baseline due to the current regulatory requirement to maintain levees in order to keep the sea out. Under the business as usual baseline for the salt field, levee banks have to be maintained at a height above sea level, and soil carbon stocks are not disturbed by salt field operation to create GHG emissions.

The VM0033 methodology estimates emissions in the baseline (BSL) scenario as:

$$GHG_{BSL} = GHG_{BSL\text{-}biomass} + GHG_{BSL\text{-}soil} + GHG_{BSL\text{-}fuel} \quad \textbf{(1)}$$

In the case of the Dry Creek salt field, only soil carbon needed to be considered for the baseline without strata differentiation (see section 3.3).

Carbon stock change can be used as a proxy for CO_2 emissions from soil organic carbon pools according to equation (2) from VM0033 as t CO_2 e yr^{-1}:

$$GHG_{BSL\text{-}soil\text{-}CO2,i,t} = 44/12 \times -(C_{BSL\text{-}soil,i,t} - C_{BSL\text{-}soil,i,,(t\text{-}T)}) / T \qquad \textbf{(2)}$$

whereby $C_{BSL\text{-}soil,i,t}$ is the soil organic carbon stock in the baseline scenario in stratum *i* in year *t* since the start of the project activity; t C ha^{-1}, and *T* the time elapsed between two successive estimations ($T=t_2-t_1$).

Field sampling occurred over ca. 1.5 months. Based on six sampling events over this time, the rate of change in soil organic carbon in the business as usual scenario (control pond with hypersaline water) was not significantly different over time (Table 2). Without a net change in CO_2e emissions from the soil organic carbon pool under business as usual, and the further aspects of the baseline scenario described above, carbon stocks were set to remain steady over time without a project activity.

Due to the short timeframe for field investigations, and the lack of published data or default values which could be used for ponds under salt field operation, further investigations in additional ponds should be carried out over time to establish a better understanding of carbon stocks under the baseline scenario.

Table 2: Regression of soil organic carbon (t CO_2e ha^{-1}) over time, starting two months before re-connection and followed over 15 months post re-connection (six sampling events), for the baseline (control pond) and project activity (trial pond). Based on data from Mosley et al. (2018) and the tidal trial. The equations are for a linear correlation, with coefficient of determination R^2, and Pearson's correlation coefficient.

SITE	CORRELATION	R^2	PEARSON CORRELATION
Control pond	y=116.35-4.34x	0.005	*P*=0.45
Trial pond	y=221+44.52x	0.638	*P*=0.03

5.3 Project emissions

The project activity is the tidal re-connection of a salt pond via a tidal gate, allowing natural flood and ebb flow from an adjacent mangrove creek to enter or drain the pond naturally. The gate can be remotely closed should this be necessary because of water quality issues (which has not arisen in the 1.5 years of operation). The activity will also require construction to close gaps in parts of the levee bank to keep the trial pond isolated from adjacent salt field ponds.

Field measurements were carried out in the trial pond (Mosley et al. 2018; Dittmann et al. 2019) over a 1.5 year time span. The longer timeline projection for 'with project' conditions utilised data for soil and biomass carbon stock changes from a chronosequence, which allowed a space-time substitution (Mosley and Tan Dang, unpubl., Clanahan 2019).

The VM0033 methodology estimates emissions in the project scenario (with project WPS) as:

$$GHG_{WPS} = GHGW_{WPS\text{-}biomass} + GHG_{WPS\text{-}soil} + GHG_{WPS\text{-}burn} + GHG_{WPS\text{-}fuel} \qquad \textbf{(3)}$$

The carbon pools considered for the tidal re-connection project activity are soil and biomass carbon (see section 3.3). The dead mangrove trees which occurred due to the salt field creation and operation were considered necromass. In the trial pond, these trees would have been dead since the salt pond was created in 1937.

For CO_2 emissions from soils, the same stock change approach can be followed as for the baseline (equation 2). Field measurement took place for only a short time period, and while the soil organic carbon increased significantly (Table 2), we followed a different modelling approach to estimate soil carbon in 30 and 100 years.

The model was guided by historic carbon sequestration rates measured from soil carbon and ^{210}Pb accumulation rate data (Dittmann et al. 2019) at a reference site near the trial pond. Soil carbon sequestration rates were determined for the mangrove low marsh stratum, but for other strata, the sequestration rate was adjusted by a relative ratio from minimum organic carbon accumulation rate values of coastal wetland and mangrove, based on global data in Wilkinson et al. (2018). The starting point for our model was the soil organic carbon stock measured by Mosley et al. (2018) prior to the re-connection of the pond. The model also included the area for each of the strata, and this approach was followed for the strata specific areas in each of the four scenarios, and with sea level rise.

A stock change approach was followed for CO_2 emissions from the biomass carbon pools. This approach is based on the sum of the net carbon stock changes in tree and shrub carbon pools during the project scenario per year for each of the strata. We used data obtained from the Salt to C project on saltmarsh and mangrove biomass carbon. Due to the short timeframe of the research project (1.5 years post-connection), and as no mangrove established during that time within the trial pond, we assumed a three-year period prior to initial mangrove establishment, with linear expansion and retreat rates. These linear expansion rates were compared to rates of increase in mangrove vegetation at a nearby chronosequence (Clanahan et al. 2019). They were found comparable for approximately the first 30 years of the SLR scenarios, increasing to exceed the rates found in the chronosequence in later years. This is expected as SLR rates increase, resulting in more rapid vegetation expansion. For each of the strata, we further considered the mix of vegetation occurring at different elevations (Dittmann et al. 2019). These data were modelled to obtain biomass for each of the scenarios and under the SLR scenario selected (see section 5.1).

A number of gaps and assumptions made in this assessment must be noted, as these highlight areas where further research is required to better understand the carbon dynamics in tidal re-connection systems. Once a stratum has retreated, it has been assumed that the carbon stock in the retreat area is zero. In the absence of information about vegetation and carbon decay rates, this is a very conservative approach, as it can be expected that carbon in inundated vegetation, particularly in mangroves, will be stored for some time after inundation, and could then contribute to soil carbon as vegetation breaks down. For scenario 4, in total carbon stock calculations, there has been no inclusion of carbon in existing vegetation in areas outside the salt ponds, as this is unknown. Similarly, in the absence of data about soil stocks outside the ponds, it has been assumed that the existing soil carbon in these areas is similar to that in the ponds. This could, however, be significantly different.

Sea level rise will likely lead to progressive inundation of re-connected salt ponds (Figure 7), particularly in section 3 of the Dry Creek salt field (Figure 3). The ponds, which can be expected to be fully colonised by saltmarsh and mangrove within 30 years of re-connection by ca. 2050, are at risk of being submerged under the worst case SLR scenario selected, and could be replaced by seagrass beds from 2070 or 2090. However, coastal salt marsh and mangrove environments may build up in elevation over time due to sediment and organic matter deposition (Dittmann et al. 2019; Oosterlee et al. 2018, Palinkas & Engelhardt 2019; Woodroffe et al. 2016). This may potentially enable them to keep pace with SLR, or reduce predicted strata changes shown in Figure 7, but the rate of this process is highly uncertain for the project area and South Australia generally. Without marsh build up, area at lower elevation strata could be lost due to SLR and may not equal the area increase at higher elevation strata. Landward retreat options, as shown in scenario 4, are required to allow continued presence of saltmarsh and mangrove along the samphire coast with SLR, and assure permanence of carbon sequestration through these tidal wetland ecosystems (Figure 7).

The total area for scenarios 1 to 3 remains relatively unchanged over the century, while area of blue carbon ecosystems will increase over time for scenario 4 as a consequence of inland retreat with SLR (Figure 8). The strata composition within each scenario changes over time, as seagrass beds replace mangrove with SLR, especially after about 2070–2090. Under scenario 4, the area of mangrove and saltmarsh strata remains in extent but shifts inland.

The carbon stock changes under the project scenarios indicate a gradual increase in soil carbon over the decades as SLR continues (Figure 9). Carbon stored in above ground biomass of mangrove and saltmarsh may be lost when the particular elevation depth strata become submerged, particularly in scenarios 1 and 2 (Figure 9). This can also lead to a decrease in total carbon stocks in these scenarios. The outlined effects of SLR could pose challenges for a tidal re-connection carbon project, unless landward retreat is included in the project boundary considerations. Total carbon stock will gradually increase for project scenarios 3 and 4.

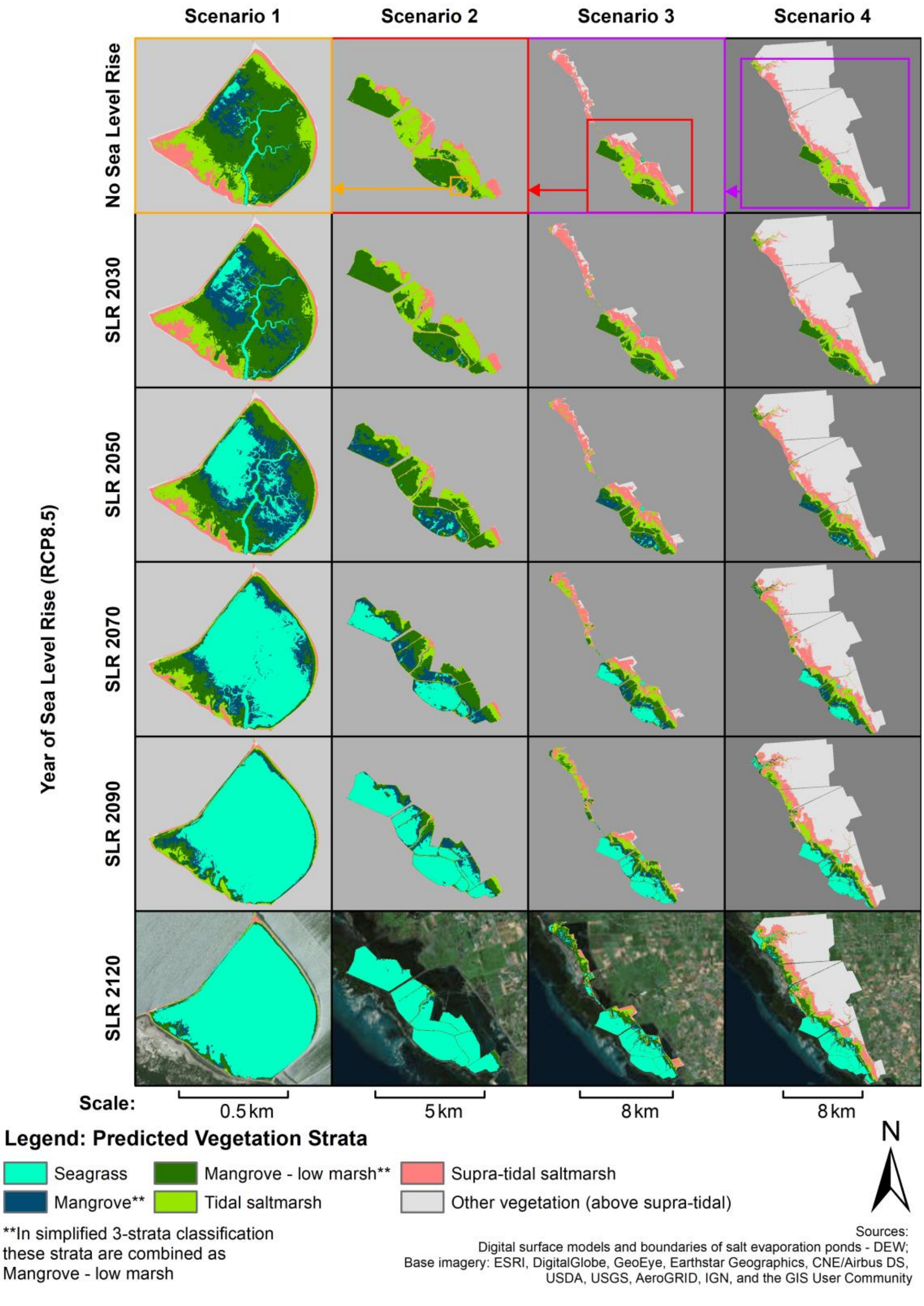

Figure 7. Maps of the predicted extent of vegetation strata for four scenarios under sea level rise (SLR), using the maximum SLR from RCP8.5 of the IPCC, estimated for several intervals over the coming century (2020–2120). The scenarios included (1) the trial pond stays re-connected, (2) additional low-lying ponds are re-connected to tidal flow, (3) higher-lying ponds are re-connected in addition to those opened up under scenario 1 and 2, and (4), including an extended inland area for landward retreat. Note the elevation intervals, where it is predicted that the vegetation strata will change with SLR and therefore differ for each modelled year. Base imagery is only shown in SLR 2120. The scale for each scenario differs.

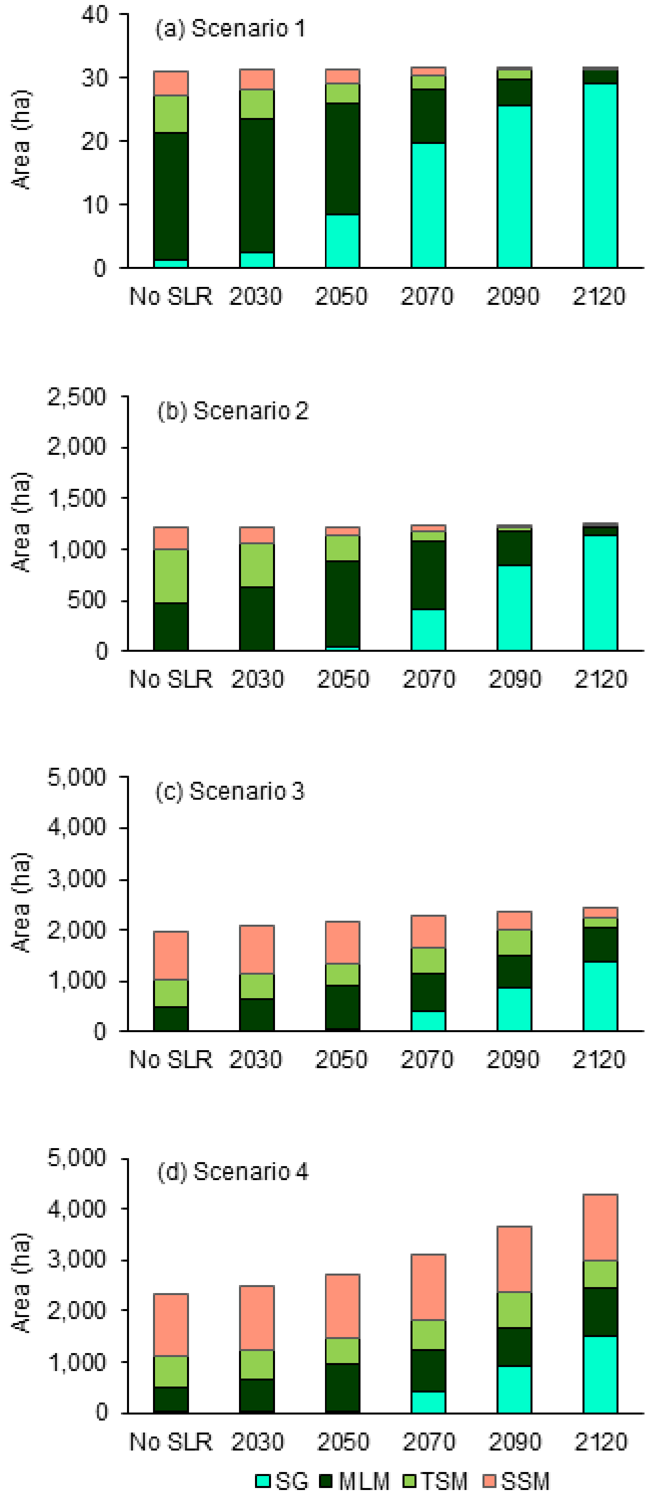

Figure 8. Progressive change in area of each stratum for the four scenarios under sea level rise (SLR), using the maximum SLR from RCP8.5 of the IPCC, estimated for several intervals over the coming century (2020–2120). The 'No SLR' area shown is the ultimate area of the strata that would be achieved for the four scenarios under current (2019) sea levels. SG = seagrass, MLM = mangrove –low marsh, TSM = tidal saltmarsh, SSM = supra-tidal saltmarsh.

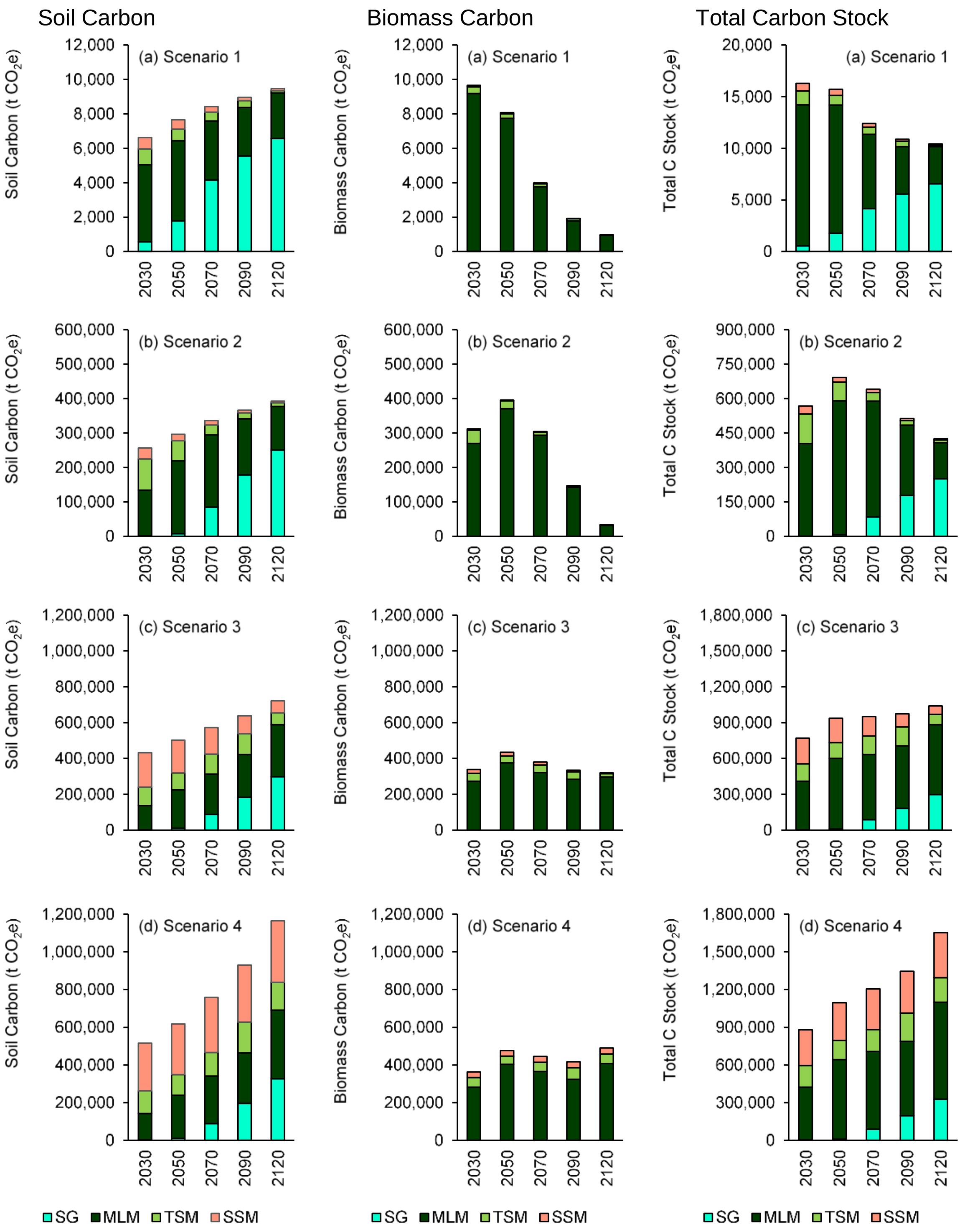

Figure 9. Total soil carbon, biomass carbon (including carbon in above ground biomass and below ground biomass) and total carbon stocks for each stratum for the four scenarios under sea level rise (SLR), using the maximum SLR from RCP8.5 of the IPCC, estimated for several intervals over the coming century (2020–2120). Note no biomass component was calculated for seagrass, as above ground biomass is negligible, and below ground biomass is accounted for in the soil carbon for seagrass. Note for scenario 4, biomass carbon and total carbon for areas outside the ponds exclude any carbon stocks in existing vegetation. Existing soil carbon outside the ponds was assumed to be the same as determined in the reference pond, but may differ from this. SG = seagrass, MLM = mangrove –low marsh, TSM = tidal saltmarsh, SSM = supra-tidal saltmarsh.

5.4 Leakage

Leakage through activity shifting from re-connection of salt fields will not be relevant, as the salt field is already in a holding pattern since commercial salt mining operation ceased. The salt from Dry Creek had been used for industrial soda ash production, which ceased operation (Bell 2014). With the loss of demand for salt, it is highly unlikely that a new salt field would be created elsewhere because of a blue carbon project activity.

Ecological leakage can also be considered as not applicable, as re-connection of salt ponds will not cause changes in the water table outside of the project area, nor vegetation dieback elsewhere, and no increased GHG emissions because of an effect on adjacent habitats.

5.5 Net greenhouse gas emission reduction and removals

The net GHG emissions reductions (NER, in t CO_2e) from tidal re-connection can be calculated as:

$$NER = GHG_{BSL} - GHG_{WPS} \qquad \textbf{(4)}$$

based on the VCS methodology VM0033, but without a fire reduction premium or leakage which are not applicable in this case. GHG_{BSL} and GHG_{WPS} are CO_2e emissions from the baseline (BSL) and with project scenario (WPS) respectively.

The net project benefit from the four scenarios follows the pattern seen for the total carbon stock changes (Figures 9, 10), and the baseline considerations made (section 5.2). For a crediting period of 30 years, the net increase by 2050 was estimated to be around 400 000–500 000 t CO_2e for scenarios 2 to 4 (Figure 10). Permanence would be assured under scenarios 3 and 4, i.e. with hypothetical conversion of larger areas of salt ponds to tidal flows and providing landward retreat.

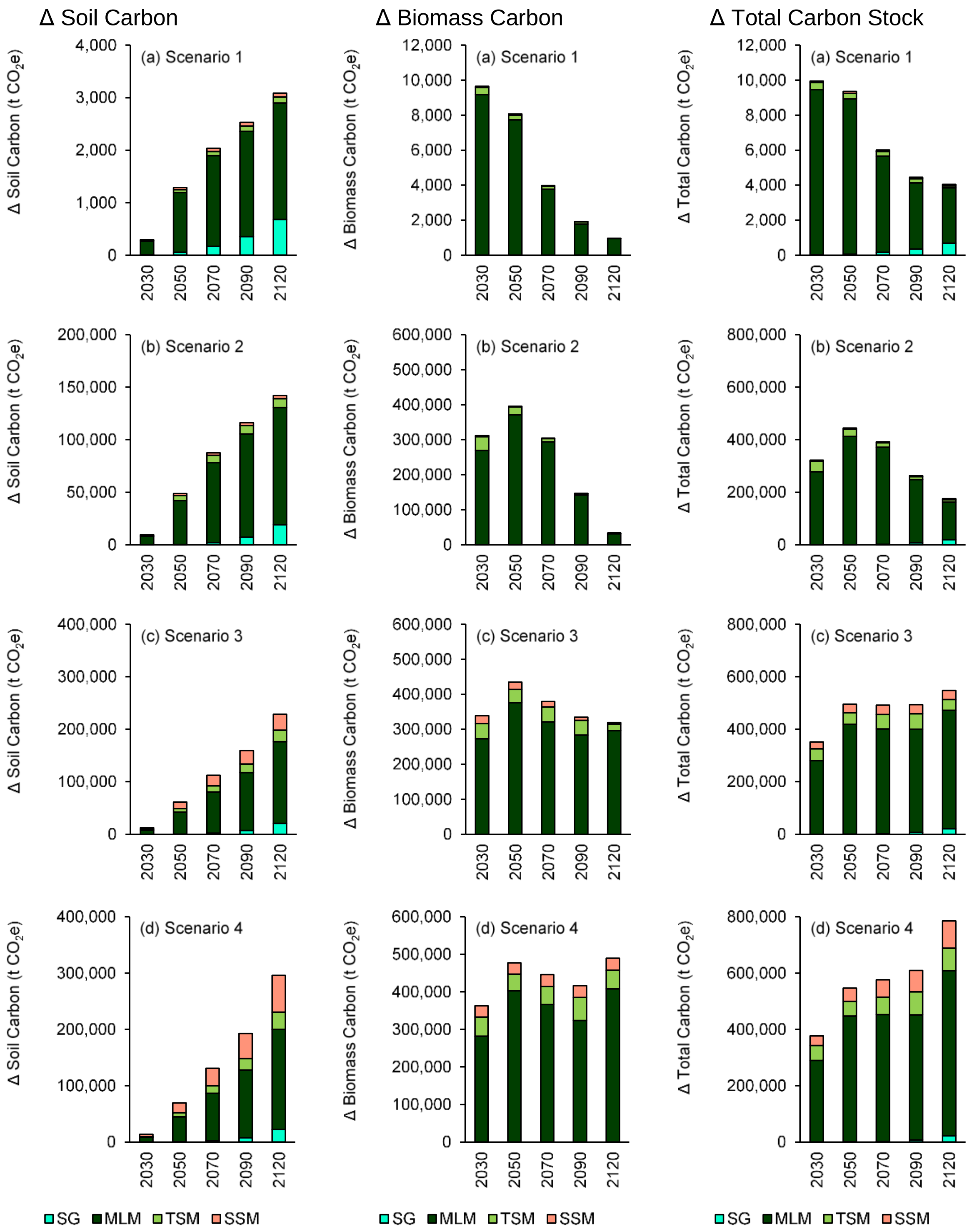

Figure 10. Change in soil carbon, change in biomass carbon (including carbon in above-ground biomass and below-ground biomass) and change in total carbon stocks for each stratum for the four scenarios under sea level rise (SLR), using the maximum SLR from RCP8.5 of the IPCC, estimated for several intervals over the coming century (2020–2120). Note no biomass component was calculated for seagrass, as above ground biomass is negligible, and below ground biomass is accounted for in the soil carbon for seagrass. SG = seagrass, MLM = mangrove –low marsh, TSM = tidal saltmarsh, SSM = supra-tidal saltmarsh.

6 Conclusions and recommendations for blue carbon opportunities through salt field restoration

The investigations by the Salt to C project have illustrated that re-introduction of tidal flow to a salt field will have positive outcomes for GHG removals. Tidal re-connection is a project activity suitable for a blue carbon project which can be applied in South Australia. Carbon stocks in the soil and biomass carbon pools can be projected *ex ante* to be higher under the project scenario than the business as usual baseline state of continued salt pond operation. The net GHG emissions reductions in 30 years crediting time could exceed 400 000 to 500 000 t CO_2e for hypothetical scenarios with further ponds re-connected, with an estimated market value of $6–8 million for a 30 year crediting period using the latest carbon price from the CER ($16.50).

For tidal re-connection of salt ponds, the typical complexity and cost burden of accounting for methane emissions is not an issue. With the prevailing hypersaline conditions, no methane emissions were detected.

The landscape slope, low development and size of the hinterland of the salt field offers good potential for maintaining wetland resilience. The shoreline profile of the samphire coast offers inland retreat options, which can counter a lack of sediment supply from little riverine inflow. Project planning, GHG accounting and permanence considerations will need to factor in whether to extend the site boundary landward beyond the current extent of saltmarsh to account for future sea level rise.

Carbon offset gains would be higher the larger the area of salt ponds re-connected, as modelled for several scenarios of potential further re-connections. If SLR continues along the worst case projection by the IPCC, mangrove and saltmarsh would be replaced by seagrass beds in lower lying strata, yet permanence can be achieved if the initial project boundary is set to be large enough to include retreat options inland from the salt field.

This report used data obtained in the Salt to C and associated local projects, instead of default values which are not very applicable for the southern Australian context. In the future, regionally-specific default values should be generated through further studies.

The Salt to C project filled knowledge gaps on tidal re-connection and blue carbon in South Australia. Outcomes can inform a project specific methodology under the ERF, and function as a pilot study for the state and Commonwealth governments.

Tidal re-connection can be a recommended blue carbon project activity and meets criteria for tidal wetland restoration projects under the VCS, and also under the offset integrity standards used by the Commonwealth Government to assess new ERF projects (Dittmann et al. 2019). It is thus recommended to move forward with a formal blue carbon project feasibility assessment and project planning, including:

(a) analysing scenarios (area size, whether seagrass can be included) and selecting preferred alternatives

b) financial and legal analysis

c) determine project proponent and partners

d) evaluate benefits of a grouped project approach

d) confirming South Australian Government and Commonwealth government support

e) start drafting project description document.

These items can be done in parallel, but stakeholder negotiations may take some time. The most promising approach for a blue carbon project would be a grouped project with a large boundary. The trial pond should be progressed as a pilot project for tidal re-connection as a carbon offset accounting project activity in Australia.

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