Optimising and managing coastal carbon

Comparative sequestration and mitigation opportunities across Australia's landscapes and land uses

Final report

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Anissa Lawrence, Elaine Baker and Catherine Lovelock

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For further information

Anissa Lawrence
TierraMar *Consulting Pty Ltd*

PO Box 1260
Sutherland, NSW 1499
AUSTRALIA
Mob: +61 419 903 800
F: +61 2 9528 0176
E: anissa@tierramar.com.au
ABN: 88 143 271 756
www.tierramar.com.au

Disclaimer

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List of Acronyms

AGB	Above ground biomass
BGB	Below ground biomass
C	Carbon
CFI	Carbon Farmers Initiative
CH_4	Methane
CO_2	Carbon dioxide
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DAFF	Department of Agriculture Fisheries and Forestry
DCC	Department of Climate Change
DECCW	Department of Environment, Climate Change and Water
ETS	Emissions Trading Scheme
GHG	Greenhouse gases
ha	hectare
IPCC	Intergovernmental Panel on Climate Change
LULUCF	Land-use, land-use change and forestry
MBI	Market-based instrument
Mg	Megagram
Mt CO_2 eq	Million tonnes of carbon dioxide equivalent
N	Nitrogen
NCAS	National Carbon Accounting System
NCOS	National Carbon Offset Standard
NGGI	National Greenhouse Gas Inventory
N_2O	Nitrous oxide
t	Tonne
Tg	Teragram
UNFCCC	United Nations Framework Convention on Climate Change

Contents

Executive summary

This report summarises the ability of Australia's coastal wetland ecosystems, particularly mangroves, saltmarsh and seagrass to capture and store carbon. Coastal carbon capture and storage was compared with carbon capture of Australia's terrestrial ecosystems, including native forests, grasslands, croplands, freshwater wetlands and agricultural land use.

It is internationally recognised that carbon sequestration, or removing carbon from the atmosphere and storing it in vegetation and soils is a key part of the strategy to mitigate against the world's changing climate. The focus of Kyoto and many other international forums has been on accounting for emissions and removals of greenhouse gases from the land, including the growth and life cycles of forests and agricultural crops, soils, land cover change and land management.

There is evidence and growing consensus that through avoided emissions, conservation, repair and sustainable use the world's coastal wetland ecosystems can play a major role in carbon management. Known as blue carbon sinks, mangroves, seagrass and saltmarsh can sequester and store carbon in their sediments and biomass at higher rates than those of terrestrial forests. Unlike most terrestrial ecosystems, the carbon stored in coastal wetland ecosystem sediments has extremely long residence times, potentially for millennia.

Australia's coastal wetland ecosystems sequester and bury carbon at rates of up to 66 times higher and store 5 times more carbon in their soils than those of our terrestrial ecosystems, including forests, on a per hectare basis.

Taking up less than 1% of landmass, the average national annual carbon burial of coastal ecosystems may account for 39% of that for all ecosystems (183.2 Tg (million tonnes) CO_2 eq yr^{-1} of a total of 466.2 Tg CO_2 eq yr^{-1}).

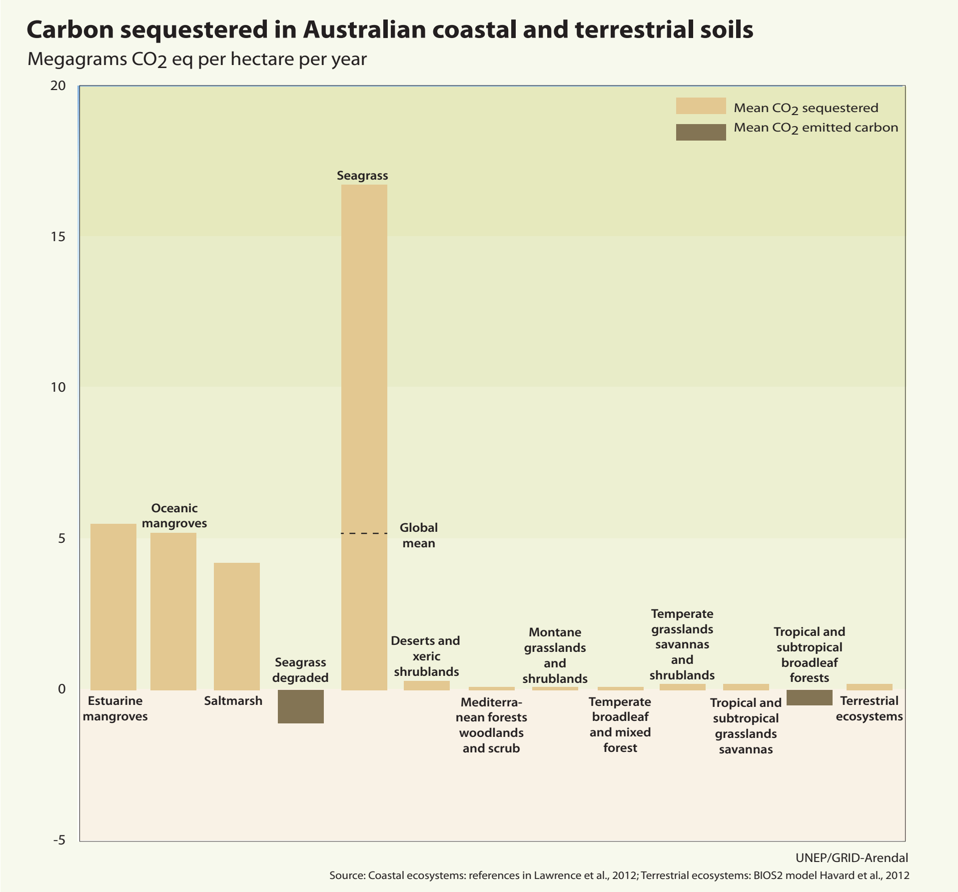
Australian coastal wetland ecosystems are estimated to store on average at least 5% of all carbon stored in Australian ecosystems (biomass and soils) (at least 22 Pg (billion tonnes) CO_2 eq of a total of 441.2 Pg CO_2 eq).

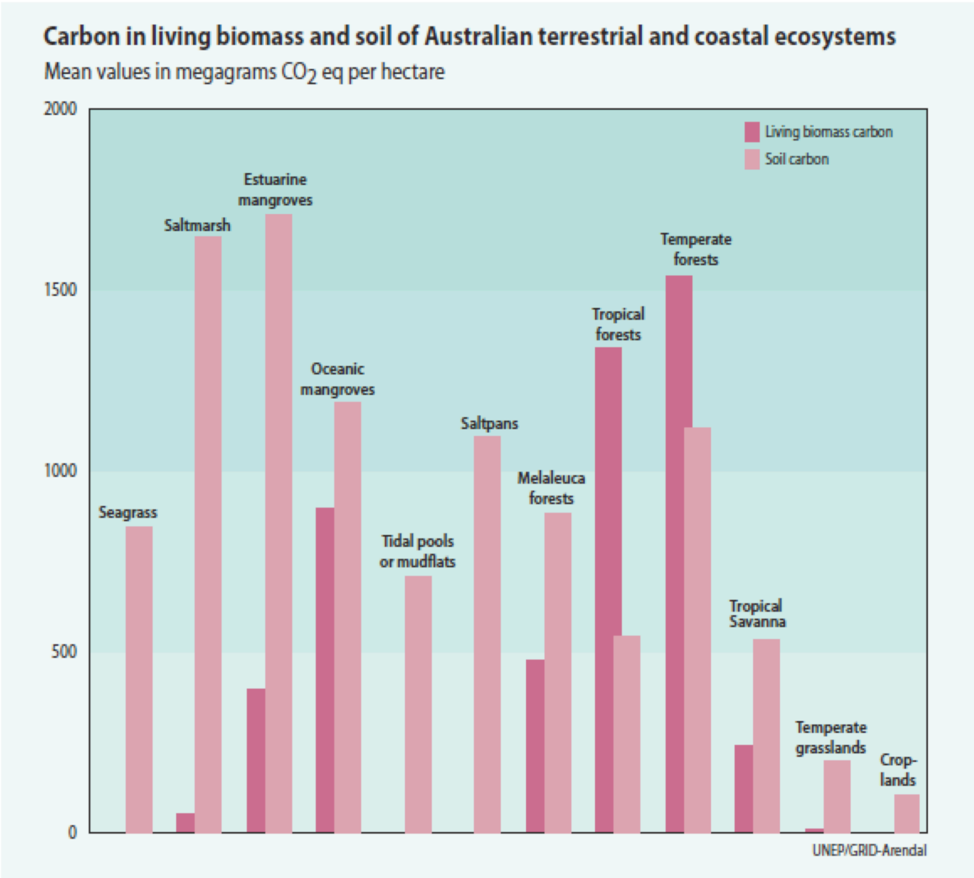
Australia is estimated to be losing its coastal wetland ecosystems at an annual rate of 0.01-1.99% for mangroves, 1.17% for saltmarsh and 0.05% for seagrass.

Degraded and lost coastal wetland ecosystems are estimated to have emitted at least 22.5 Tg (million tonnes) CO_2 eq into the atmosphere since European settlement and continue to emit up to 0.22 Tg CO_2 eq each year. This is the equivalent of an additional 4,397 cars on Australian roads each year.

There is potential for substantial gains in carbon sequestration associated with reinstatement of tidal flows to degraded coastal wetland ecosystems in a relative short time (<20 years).

Healthy coastal wetlands ecosystems produce negligible amounts of greenhouse gases such as methane and nitrous oxide and in some cases, can act as methane sinks which adds to their value to mitigate greenhouse gas emissions.


Carbon sequestered in Australian coastal and terrestrial soils
Megagrams CO2 eq per hectare per year
20
15
10
5
0
-5
Mean CO2 sequestered
Mean CO2 emitted carbon
Seagrass
Global mean
Oceanic mangroves
Estuarine mangroves
Saltmarsh
Seagrass degraded
Deserts and xeric shrublands
Mediterra-nean forests woodlands and scrub
Montane grasslands and shrublands
Temperate broadleaf and mixed forest
Temperate grasslands savannas and shrublands
Tropical and subtropical grasslands savannas
Tropical and subtropical broadleaf forests
Terrestrial ecosystems
UNEP/GRID-Arendal
Source: Coastal ecosystems: references in Lawrence et al., 2012; Terrestrial ecosystems: BIOS2 model Havard et al., 2012

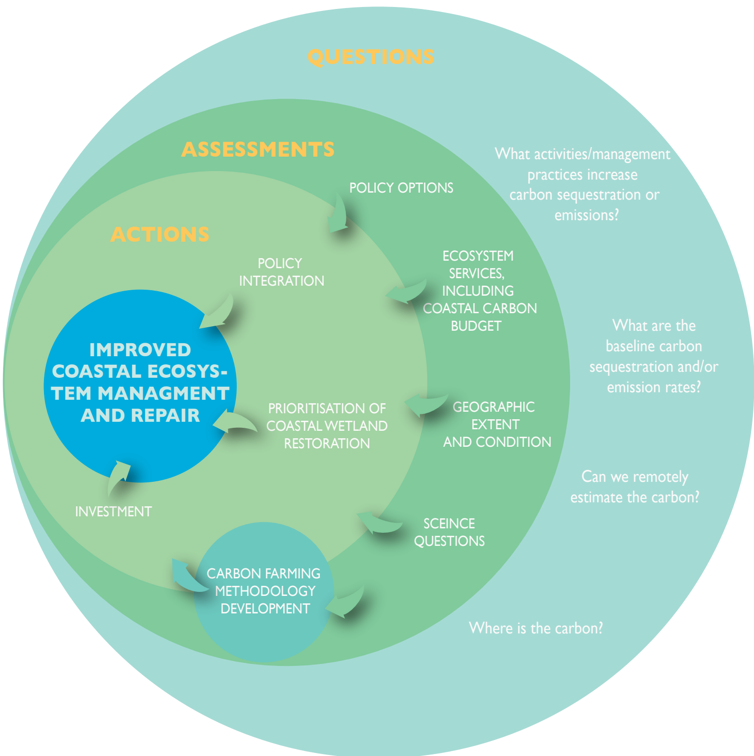

Carbon in living biomass and soil of Australian terrestrial and coastal ecosystems
Mean values in megagrams CO2 eq per hectare
2000
1500
1000
500
0
Living biomass carbon
Soil carbon
Seagrass
Saltmarsh
Estuarine mangroves
Oceanic mangroves
Tidal pools or mudflats
Saltpans
Melaleuca forests
Tropical forests
Temperate forests
Tropical Savanna
Temperate grasslands
Crop-lands
UNEP/GRID-Arendal

Australia has yet to fully recognise the important role that coastal ecosystems can play in carbon management. Coastal ecosystems are not part of our National Carbon Accounts. The Australian Government Clean Energy Futures Package through the Carbon Farming Initiative (CFI) is only supporting farmers and land managers to earn carbon credits by storing carbon or reducing greenhouse gas emissions on the land. Coastal ecosystems are the habitats or "productive farms" for our fishers, yet the Carbon Farming Initiative specifically excludes coastal ecosystems. Through these policy limitations Australia is not only limiting its carbon management options, but is ignoring many other community benefits of food production, biodiversity, flood control, coastal buffering water quality and recreational and aesthetic benefits that coastal ecosystems provide. As a direct consequence of coastal ecosystems being omitted from Australian policy, the peer-reviewed literature relating to carbon sequestration and storage within coastal wetland ecosystems for Australia is very limited compared to that available for terrestrial ecosystems and their land-uses. Contrastingly, scientific understanding of carbon sequestration and potential emissions from coastal wetland ecosystems globally is much higher. This body of international knowledge is sufficient for developing effective carbon management, policy, and conservation incentives for coastal carbon in Australia.

While the data we do have is generally consistent with global estimates, it is imperative that we strengthen the evidence base (the data) in order to improve the decision making process over the broad range of "blue carbon" habitats in Australia. The recognition and management of the carbon storage and sequestration potential of these coastal wetland ecosystems provides an opportunity to strengthen socio-economic resilience of Australia's coastal communities and estuarine and marine based industries, avoids significant emissions from ecosystem degradation, while also supporting existing wetland conservation efforts.

The newly adopted definition of wetland drainage and rewetting under the Kyoto Protocol provides an immediate incentive to account for anthropogenic greenhouse gas emissions and removals by Annex-I Parties, of which Australia is one. These represent further potential mechanisms for reducing emissions of coastal blue carbon to the atmosphere.

To move forward Australia needs a comprehensive approach that leads to improved management and restoration for coastal wetland ecosystems in Australia, as shown below:



We make a number of recommendations:

Policy and management

Australia should set in place a timetable and processes to integrate *Blue Carbon* into national climate policy. There would be a range of follow on implications including:

- ensuring coastal ecosystems are a priority within implementation initiatives such as the Carbon Farming Initiative and Biodiversity Fund, commensurate with their high carbon values;
- identifying site and landscape scale restoration priorities to deliver improvements to coastal wetland ecosystems providing (or with the potential for) high carbon and other values such as fisheries habitat repair;
- updating national datasets for mangroves and seagrass ecosystems, and in the case of saltmarsh, developing national datasets, mapping the areal extent and assessing condition to provide a comprehensive understanding of status and land-use changes, comparable across regions and states and contributing to National Carbon Accounting;
- recognising the multiple values, including CO_2 mitigation values of coastal ecosystems to develop and implement a National Action Plan for the Conservation and Restoration of Australia's coastal ecosystems, that seeks to standardise conservation and management regulations and measures across regions and States and supports restoration and rehabilitation of priority coastal wetlands ecosystems; and
- exploring the feasibility of community monitoring approaches, management intervention and providing incentives for maintaining carbon rich ecosystems. Participation of key stakeholder groups such as commercial and recreational fishing groups, coastal farmers and indigenous communities in projects to generate new revenue streams related to coastal wetland repair would be important in this process;

Internationally Australia should lead policy development as part of the United Nations Framework Convention on Climate Change and its related processes and mechanisms, working within the UNFCCC and its related processes and mechanisms with partners (e.g. Indonesia) to incorporate blue carbon into the UNFCCC.

Scientific understanding

Recognising that coastal ecosystems, when compared to terrestrial ecosystems, are a very significant part of Australia's carbon stores and carbon management opportunities, we need to build on existing scientific data, analysis and available technologies to develop a coherent Australia-wide data gathering and assessment initiative focusing on:

- assembling sufficient data to support the development of policy and management activities;
- addressing gaps in knowledge in relation to carbon storage and sequestration for Australian coastal wetland ecosystems, utilising consistent internationally accepted measurement and assessment methodologies that are comparable across coastal and terrestrial ecosystems;
- undertaking detailed baseline carbon inventories of coastal wetland ecosystems and incorporate coastal carbon into the Australian Terrestrial Carbon Budget (being undertaken by CSIRO) to quantify national coastal carbon storage, sequestration and losses;
- undertaking a baseline assessment of related Australian coastal wetland ecosystem services (the need for a bundled/layered/stacked Blue Carbon plus other ecosystem services approach);
- conducting targeted research and monitoring to more accurately quantify the greenhouse gas emissions resulting from degradation, conversion and destruction of coastal ecosystems;

- establishing a network of field projects that demonstrate the capacity for carbon storage for coastal wetland ecosystems and the emissions resulting from degradation, conversion and destruction of coastal ecosystems; and
- conducting research quantifying the consequences of different coastal restoration and management approaches on carbon storage and emissions in coastal wetland ecosystems.

1 The role of coastal wetland ecosystems in carbon storage and sequestration

Coastal wetland ecosystems can sequester carbon at rates estimated to be up to 50 times higher than mature tropical forests and total carbon deposits per square kilometre may be up to 5 times the carbon stored in tropical forests (Murray et al. 2011).

Under Australia's National Carbon Accounting System (NCAS), land based emissions (sources) and removals (sinks) of greenhouse gases form a major part of Australia's emissions profile. The Australian Government reports that around 24% of Australia's human induced greenhouse gas emissions come from activities such as livestock and crop production, land clearing and forestry (DCCEE 2012).

The focus of the Australian Government to date has been on accounting for emissions and removals of greenhouse gases from the land - the growth and life cycles of forests and agricultural crops, climate, soils, land cover change and land management. More recently, the Australian Government Clean Energy Futures Package through the Carbon Farming Initiative (CFI) is supporting farmers and land managers to earn carbon credits by storing carbon or reducing greenhouse gas emissions on the land. These credits can then be sold to people and businesses wishing to offset their emissions. The CFI also helps the environment by encouraging sustainable farming and providing a source of funding for landscape restoration projects.

In contrast, coastal wetland ecosystems, in particular seagrass, mangroves and saltmarsh have received much less attention with respect to their role in greenhouse gas (GHG) emissions and removals. Ongoing research suggests that beyond providing important habitat for fisheries, their high biodiversity values and other ecosystem services, they also play a key role in mitigating global climate change through their ability to store carbon (McCleod et al. 2011). Known as **blue carbon** sinks, mangroves, seagrass and saltmarsh can sequester and store carbon in their sediments and biomass at higher rates than those of tropical forests (Murray et al. 2011). Unlike most terrestrial ecosystems, the carbon stored in coastal wetland ecosystem sediments has extremely long residence times[1], potentially for millennia.

[1] The average time spent in a reservoir by an individual atom or molecule. With respect to greenhouse gases, residence time refers to how long on average a particular molecule remains in the atmosphere or locked up within the environment such as within a tree or in the soil.

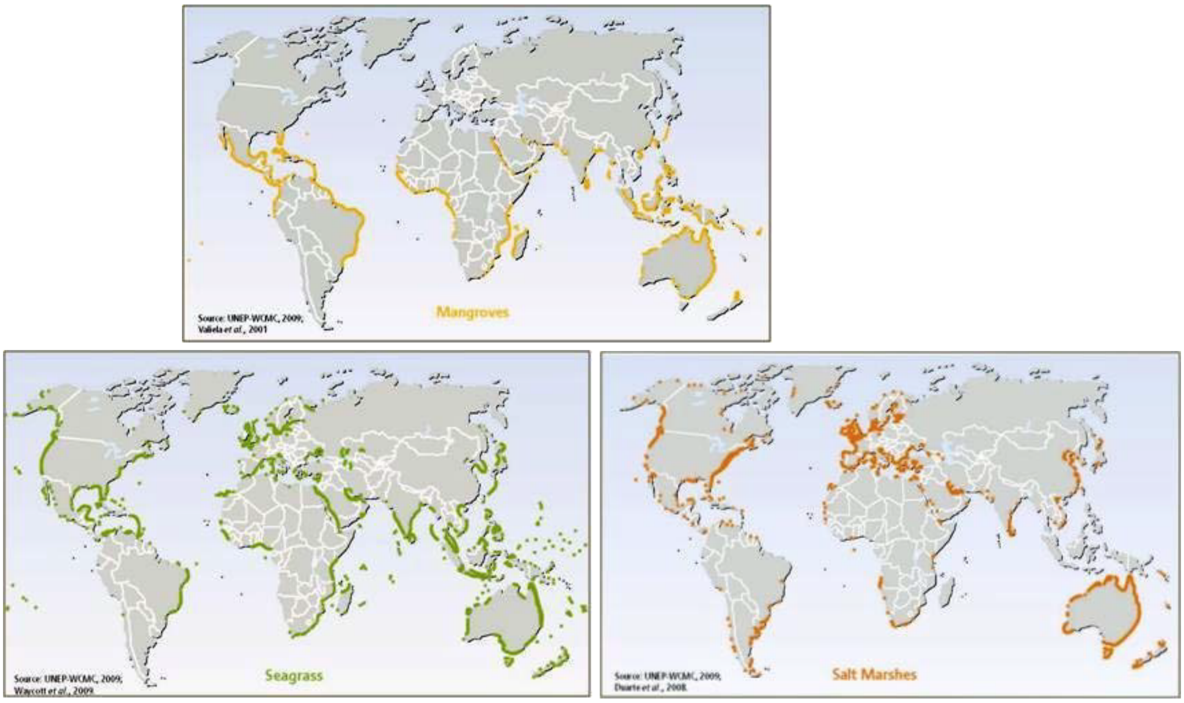


Figure 1. Global distribution of mangroves, seagrass and saltmarsh (Source: Murray et al. 2011)

Australia has a significant proportion of global coastal wetland ecosystems. It holds 7.1% of world mangroves and has the world's second largest mangrove area, after Indonesia (Giri et al. 2011). Australia has the highest diversity of seagrass species and the most extensive seagrass beds worldwide (Green and Short 2003, Butler 1999). It also has large areas of saltmarsh, particularly in the tropical north. While together these ecosystems cover a relatively small area – around 1% of the Australian coverage of forests (BRS 2009) – they are some of our most threatened ecosystems. Globally, about 30% of mangroves have been lost (Alongi 2002). We have lost around 17% of our mangroves at some sites since European settlement (Duke 2006), mostly due to urban development. In some places we have lost almost 100% of our saltmarsh due to land reclamation and mangrove encroachment (Wilton 2002). Over at least the last 50 years substantial invasion of saltmarsh by mangrove has occurred along about 2,000 km of coastline, from southern Queensland to South Australia (Saintilan and Williams 1999). Upward movement of mangrove at the expense of saltmarsh has been predicted as an early response to rising sea level (Vanderzee 1988). In 1999 it was reported that in the prior ten years Australia had lost over 45,000 ha of seagrass from human induced excess nutrients and increases in sediments and 100,000 ha from natural events (Kirkman 1997) while Waycott et al. (2009) reported around 20,029 ha loss. With the majority of Australians living on the coast and coastal development continuing to grow, even with legislative protection, these ecosystems continue to be degraded and lost.

Similar to terrestrial ecosystems, in the coastal zone, landuse change has lead to high levels of CO_2 emissions. The draining, conversion or destruction of coastal wetland ecosystems for other uses can disrupt the carbon sequestration by coastal wetland ecosystems and may switch these ecosystems from being net sinks to net sources of carbon (McCleod et al. 2011, Lovelock et al. 2011). For example, at a site in the Hunter region of New South Wales that had been drained for pasture production since the 1950s, losses of 135 Mg and 180 Mg $C0_2$ (eq) per hectare from the top 0.2 m of mangrove and saltmarsh profiles, respectively, were estimated. This represented a loss of nearly 40% of organic carbon over a period of approximately 50 years (Howe et al. 2009, Page and Dalal 2011).

Even though the total land area of mangroves, coastal marshes, and seagrass is small compared with land in agriculture or forests, the carbon beneath these habitats is substantial. If released to the atmosphere, the carbon stored in the living biomass and soils of a typical hectare of mangroves could contribute as much to greenhouse gas emissions as three to five hectares of tropical forest. A hectare of intact saltmarsh may contain carbon in its soils with a climate impact equivalent to 488 cars on roads each year. Even a hectare of seagrass meadow, with its small living biomass, may hold as much carbon in its soils as one to two hectares of typical temperate forest (Murray et al..l 2011).

The recognition and management of the carbon storage and sequestration potential of these coastal wetland ecosystems provides an opportunity to strengthen socio-economic resilience of Australia's coastal communities, avoid significant emissions from ecosystem degradation, while also supporting existing wetland conservation efforts. There is now global and regional momentum to look at the role of coastal wetland ecosystems and habitats in climate mitigation, particularly the role that they play in carbon storage and sequestration. For Australia, with its large areas of coastal wetland ecosystems and increasing coastal development pressure, Blue Carbon may be a novel mechanism for connecting the purchasers of carbon with the suppliers of these ecosystem services to create public-private partnerships through the voluntary carbon market. Potential Blue Carbon credits generated for restoration or offset projects may provide some level of sustainable financing for the maintenance or ongoing restoration of the "natural capital" that is generated from the ecosystem services provided by coastal wetland ecosystems.

Australia is yet to recognise the role of coastal wetland ecosystems in combating or contributing to climate change. We do not currently account for the emissions and removal of greenhouse gases from these ecosystems or changes in their CO_2 emissions/removal through changes in landuse of wetlands. This document seeks to provide information to inform decision makers about how management and recognition of coastal wetland ecosystems can be useful in meeting Australia's commitment to reducing its emissions by between 5 and 15 or 25 per cent below 2000 levels by 2020[2].

1.1 Scope and objectives

The core task of the Consultants was to provide a summary of baseline information about coastal ecosystem carbon in Australia in an Information Paper. The paper is background to guide follow-on forums that will develop a National Action Plan for the Restoration and Management of Australia's estuarine and coastal wetland ecosystems. The objectives of this project were to:

undertake a comparative assessment of the carbon sequestration potential and climate change mitigation opportunities from coastal wetland ecosystems (in particular mangroves, saltmarsh and seagrass) in comparison with other key Australian ecosystems and their land-uses.

estimate the relative contribution of poorly managed and drained coastal wetlands and the emission reduction benefits associated with remedial activities – for example, through increasing tidal flow, removing barriers, repairing mangrove, saltmarsh, seagrass habitats and compare these to other proposals for ecosystem management such as changed fire regimes for tropical savannas or livestock and manure management.

derive a series of look up tables that compare and contrast various sequestration/reduced emissions opportunities.

[2] Sourced from Australian Government Fact sheet: Australia's emissions reduction targets available accessed from http://www.climatechange.gov.au/en/government/reduce/national-targets/factsheet.aspx 21 September 2012

1.2 Project methodology

This project was undertaken using the following methods:

- We used a literature review of existing relevant scientific (peer reviewed) and "grey" literature across key natural and moderately altered ecosystems across Australia, including all major terrestrial, freshwater and coastal/estuarine ecosystems. This data was then combined with existing spatial data sets such as the Natural Vegetation Information System, various wetland mapping data sets and the Australian Land Use Dataset to estimate regional emissions/sequestration. Systems already significantly altered such as urban and cultivated lands were excluded from the analysis. Where no Australian examples were available, the results from overseas studies in comparable landscapes were used.
- Discussions with key experts and other stakeholders were conducted as required, clarifying or seeking additional information to that provided in the literature.
- Development of a comparison methodology occurred to ensure the information provided was consistent and could be compared effectively.

1.3 Project limitations

Given the limited published literature available on the carbon sequestration and emissions associated with Australian coastal wetland ecosystems, it was not possible to accurately estimate the total amount of carbon stored, sequestered or emitted by coastal wetland ecosystems for all of Australia or even at a sub-regional level. Despite these data gaps we have attempted to estimate values for particular ecosystems using the available data. For example, through translating sequestration or emissions per unit area into an overall opportunity based on the area of that ecosystem. Given the lack of data for Australia, such as for seagrass in particular, suitable international data has been utilised to provide an indication of the likely range for Australian coastal wetland ecosystems. The necessary narrative on any errors, assumptions or divergences between published findings and the final estimates has been provided in the look-up tables.

As more research is undertaken to quantify carbon stocks, sequestration and emissions, greater accuracy will be available to update the estimates provided, provide greater representation across regions and reduce uncertainties. The estimates for carbon stored, sequestered and emitted however for coastal wetland ecosystems provided in this report, provide a baseline from which to build.

No Geographic Information Systems (GIS) analyses have been undertaken within this project but rather existing maps and datasets have been used where easily available and suitable, with notations added.

1.4 Format of this report

Section 1 provides background information on the role of coastal wetland ecosystems in carbon storage and sequestration in an Australian context. It also provides an overview of the scope of the project and outlines the methods applied and limitations in undertaking the work.

Section 2 provides a comparative analysis of the potential sequestration opportunities for all key Australian ecosystems and included in the look-up tables.

Section 3 provides a discussion and makes recommendations for optimising coastal ecosystem carbon, including the benefits of managing and repairing degraded ecosystems.

Section 4 provides a summary of the key findings and suggestions for the next steps.

2 Comparative carbon burial potential of Australian ecosystems

This section summarises what is known about the carbon benefits (stocks and sequestration) and emissions associated with Australian coastal wetland ecosystems, in comparison to terrestrial ecosystems and their key land uses. Key to undertaking such a comparison is an understanding of changes to carbon stored in each ecosystem (carbon sequestration or burial expressed as CO_2 flux) and fluxes of other greenhouse gases (GHG) over time with different land uses. While research on carbon stocks and sequestration associated with Australian terrestrial ecosystems and their land uses is more comprehensive than for coastal wetland ecosystems, comparisons of estimates for carbon stored and sequestered as well as GHG emissions by all key Australian ecosystems and land uses have been provided, or global averages used where data is lacking. Where possible, comparisons have been made at a bioregional level, in order to take account of differences among landscapes, however in most cases, this was not possible due to the limited data available.

Detailed lookup tables for coastal wetland ecosystems are provided in Appendix A which contain the data used to calculate Australian estimates.

2.1 The Australian landscape: Vegetation communities and land uses

Australia's vegetation, soils and land uses all play a role, to varying degrees, in the capture and storage of carbon from the atmosphere and ocean. Australia's coastal wetland ecosystems have low areal cover compared to most terrestrial ecosystems, representing less than 1% of area (Table 1). While globally the importance of wetlands as carbon sinks is widely recognised (Adams et al. 1990, Watson et al. 2000), the contribution of Australian wetlands, in particular coastal wetland ecosystems is still unclear.

Table 1. Continental extent of Australian vegetation adapted from ABARES (2010)

Vegetation category	**Area (million hectares)**	**Area (%)**
Native shrub lands and heathlands	283	36
Native grassland and minimally modified pastures	257	33
Native forests and woodlands	148	19
Annual crops and highly modified pastures	66	9
Ephemeral and permanent water features. Note – areal extent of mangroves, saltmarsh and seagrass (to 6m depth) are included within this category. Deep water seagrass is excluded.	7	1
Intensive uses (includes urban, peri-urban, mining)	3	0.4
Plantation forests	2	0.3
Perennial crops	1	0.1
Bare	1	0.1

Vegetation category	Area (million hectares)	Area (%)
Horticultural trees and shrubs	0.7	0.1
Total	**768.7**	**100**

Terrestrial ecosystems

Australia's terrestrial vegetation includes both native and introduced plant species. The most extensive types of Australian native vegetation are grasslands, woodlands dominated by eucalypts, and shrublands dominated by acacias. Australia has 148 million hectares of native forest and 2 million hectares of forestry plantations (Table 1). Together these cover approximately 19% of the continent (and make up about 4% of global forest cover). The distribution of forest types is mainly determined by climate and soil properties. Other factors, especially fire frequency and intensity, are also important. Nearly half of Australian forest is classified as open woodland (20-50% crown cover) and only 1% is tall, closed forest. The area of tall eucalypt forests where timber harvesting occurs is now estimated to be 86.6% of the original extent (DAFF 2005). Land clearing still accounts for a significant proportion of Australia's total emissions (as much as 14%; Wentworth Group, 2009).

Non-native vegetation includes a diverse array of annual, perennial and horticultural crops and plantation forests. Australia's native vegetation has been modified to varying degrees by different land uses and management practices (Table 2). Since European settlement, around 13% has been completely converted to other land uses, and a further 62% is subject to some level of disturbance (SEWPaC 2011). Australia's area of plantation forests has been expanding steadily for several years. Since 1990 the majority of new plantations have been eucalypts established on farmland and managed to produce woodchips for paper manufacture.

Table 2. **Estimated change in native vegetation extent, pre-European to 2011** (DAFF, 2011)

Major native vegetation group	Proportion remaining %
Acacia forests and woodlands	82.5
Callitris forests and woodlands	80.2
Casuarina forests and woodlands	89.7
Eucalypt low forests and woodlands	73.2
Eucalypt tall forests	86.6
Mallee woodlands and shrublands	70.2
Other shrublands	78.3
Rainforests and vine thickets	65.3
All groups	**87.5**

Grazing land/Savanna

The most extensive land use in Australia is livestock grazing –grazing accounts for over half of all land use. Native pastures dominate grazing land over a large part of northern Australian and some temperate southern areas. These native pasture rangelands contain a mix of ecosystem types including native grasslands, shrublands, woodlands and tropical savanna woodlands. Rangelands contain significant biodiversity, which is under pressure from pastoral businesses that operate across the country. The 2001 National Land and Water Resources Audit (Cofinas and Creighton 2001), estimated that there were 8–14 million cattle and 18–40 million sheep grazing Australia's rangelands.

Fire is an important element in the management of rangelands. It controls woody vegetation and stimulates the growth of grass. Natural fires are caused by lightning strikes and are more common during periods of drought and when fire management is poor. In central and northern Australia intentionally lit fires remain the most common, but a decrease in traditional fire management has generally resulted in larger and more intense fires (Cook et al., 2010).

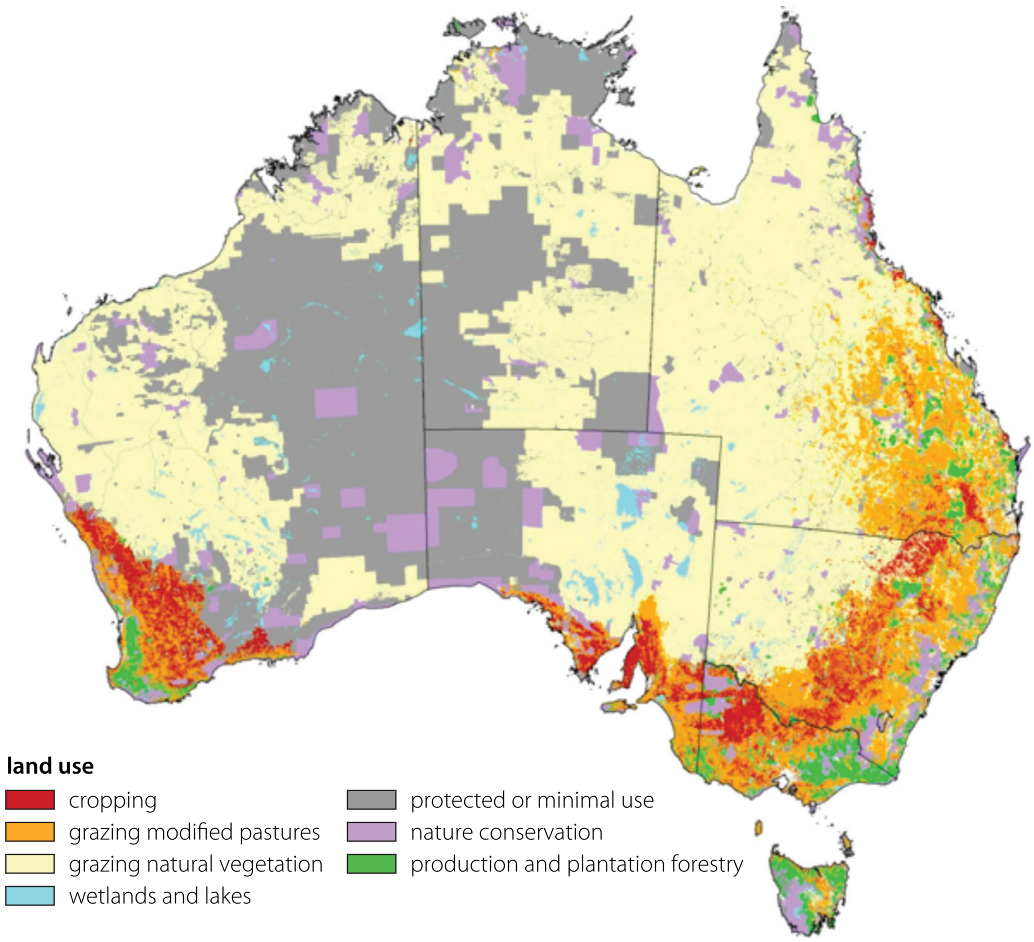


Figure 2. Map illustrating cropping and grazing landuse (ABARES 2010)

Cropland

Australian cropland covers an area of over 20 million hectares (66 million hectares if perennial pastures are included, Table 1, Figure 2). Croplands usually have high land value with a moderate to high soil nutrient status and are therefore not generally converted to forest land or grassland. Australian croplands range over a large area, so there are major variations in climatic conditions and the types of crops grown - rainfall variation is the main contributor to changes in crop yields from one year to another. Australia's croplands can be divided into northern and southern regions. Production in the northern region is mostly sugar cane, grain sorghum and cotton. In the south winter crops dominate, especially wheat, barley, oats, lupins and canola (ABARES 2010).

Soils

Australian soils are geologically old, and as a result, are highly weathered, fragile and high in salt. Australian soils have been classified into 15 main types (Figure 3). Parent rock, topography, organic matter and age influence the distribution of soil types. In general Australian soils are deficient in nutrients, especially phosphorus and other micronutrients such as copper, zinc and molybdenum (Chen et al. 2005). Consequently they require additional nutrients to maximise crop yields under current farming practices.

The conversion of native vegetation to agricultural land increases soil erosion. Soil degradation is a major problem in Australia and it is estimated that about two thirds of agricultural land is degraded. In addition to soil erosion, this degradation includes increased soil salinity, soil acidity, soil contamination, nutrient loss and declines in soil structure (Sanderman et al. 2010).

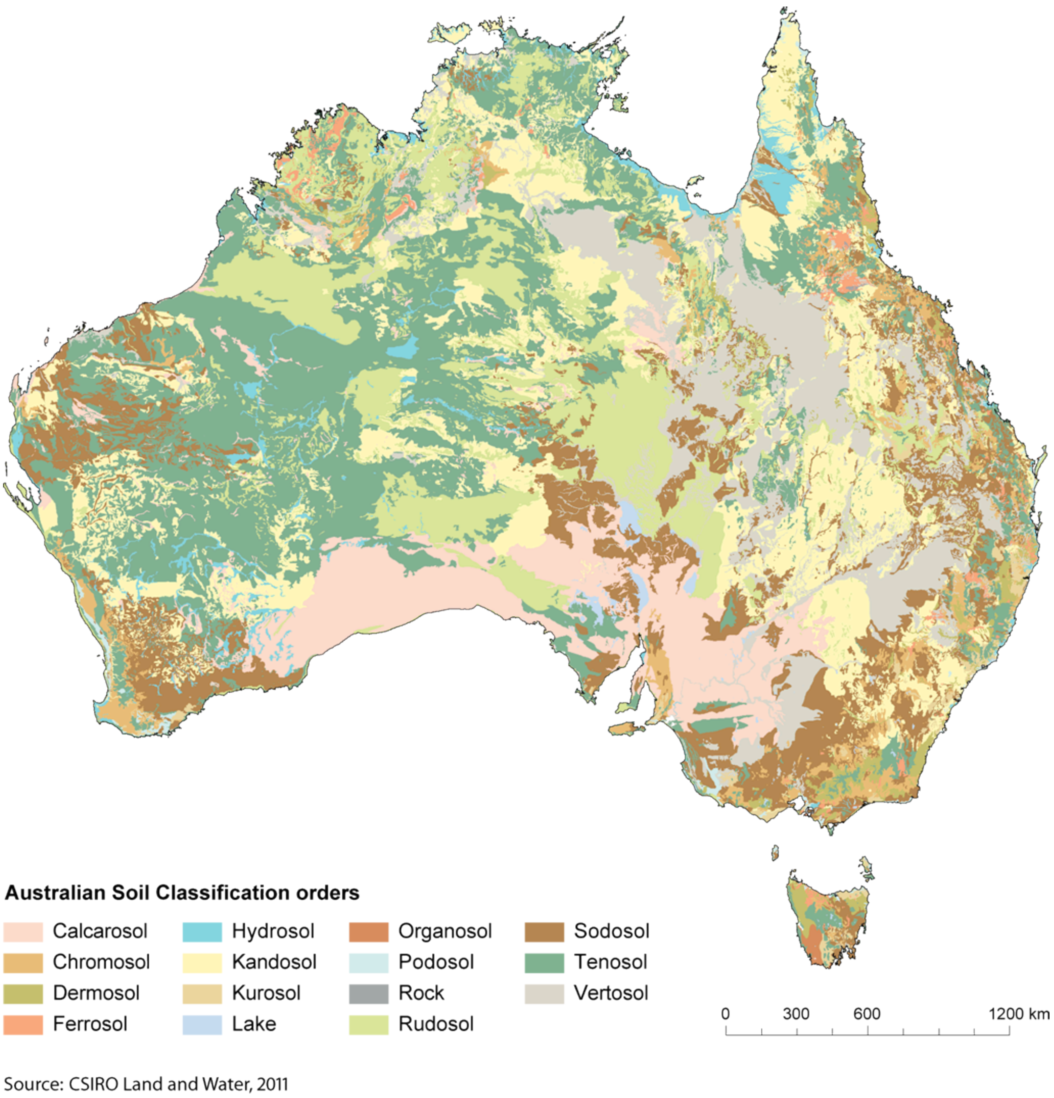


Figure 3. Generalised map of soil orders for Australia (Source: CSIRO Land and Water 2011)

Coastal wetland ecosystems

Figure 4 shows mangrove, saltmarsh and seagrass dominated areas across Australia, as provided by the Geosciences Australia OZCoasts online coastal information system[3]. As indicated in Table 7 in section 3.a of this report, Australia has between 977,975 - 990800 ha of mangroves (Sinclair and Boon 2012, Giri et al. 2011, MIG 2008, Creese et al. 2008,), 1,376,500 ha of saltmarsh (including saltpans) (Sinclair and Boon 2012, Creese et al. 2008, Bucher and Saenger 1991) and 9,256,900 ha of seagrass (McKenzie et al. 2010, Creese et al. 2008, Green and Short 2003), collectively making up less than 1% of total land area.

At the scale of the continent the productivity of mangroves is driven by temperature, but at a local or regional scale rainfall, tides inundation, waves and river flow also play a role (Alongi 2012). For saltmarsh, tidal inundation and soil moisture that are important determinants of productivity, combined with temperature, rainfall and groundwater discharge (Saintilan 2009). Water depth and light availability, as well as salinity, oxygen availability, temperature and epiphytic algal cover influence seagrass productivity (Larkum et al. 2006).

Australia has the fourth highest species diversity for mangroves globally, with the areas of greatest abundance occurring along the wet tropical coast of northern and eastern Australia. The largest forested areas of mangroves in Australia (around 75%) occur in the humid tropics in northern Australia where human population densities are low. Notable areas of mangroves however do exist in temperate regions as far south as Corner Inlet in Victoria and are generally closely linked to adjacent saltmarsh and salt pannes (Wightman 2006, Duke 2006).

Australia has diverse saltmarsh communities with a broad geographic distribution. Species richness for Australian saltmarsh is highest in temperate latitudes, with the highest number of species being recorded in Tasmanian marshes (Saintilan 2009). Many Australian saltmarsh communities, especially in New South Wales and Victoria are on private land and subject to stock grazing (Laegdsgaard 2006, Sinclair and Boon 2012).

Australia hosts the highest number of seagrass species of any landmass in the world (about 30 species: Walker 2003; Coles et al. 2003a), with large multispecies meadows across vast shallow areas of the coastal fringe. Australia's seagrasses are divided into temperate and tropical distributions with Shark Bay in WA and Moreton Bay in QLD being located at the centre of the overlapping tropical-temperate zones. Temperate species are distributed across the southern half of the country, with the highest biomass and regional species diversity occurring in south Western Australia. Western Australia has a higher diversity of seagrass species than anywhere else in the world (Wasik and Prince 1987), mainly due to the presence of tropical species within the temperate region due to warm currents and due to the large extent of suitable shallow water habitats. Tropical seagrasses are highly diverse but generally have lower biomass than those found in temperate regions. Most tropical seagrasses are found in the Gulf of Carpentaria, the Torres Straits and the central and southern Great Barrier Reef World Heritage Area (Green and Shore 2003). McKenzie et al. (2012) reported that the Great Barrier Reef World Heritage Area contains more than 50% of the total recorded area of seagrass in Australia (Green and Short 2003) and between 6% and 12% globally (Duarte et al. 2005) making the Great Barrier Reef's seagrass resources globally significant.

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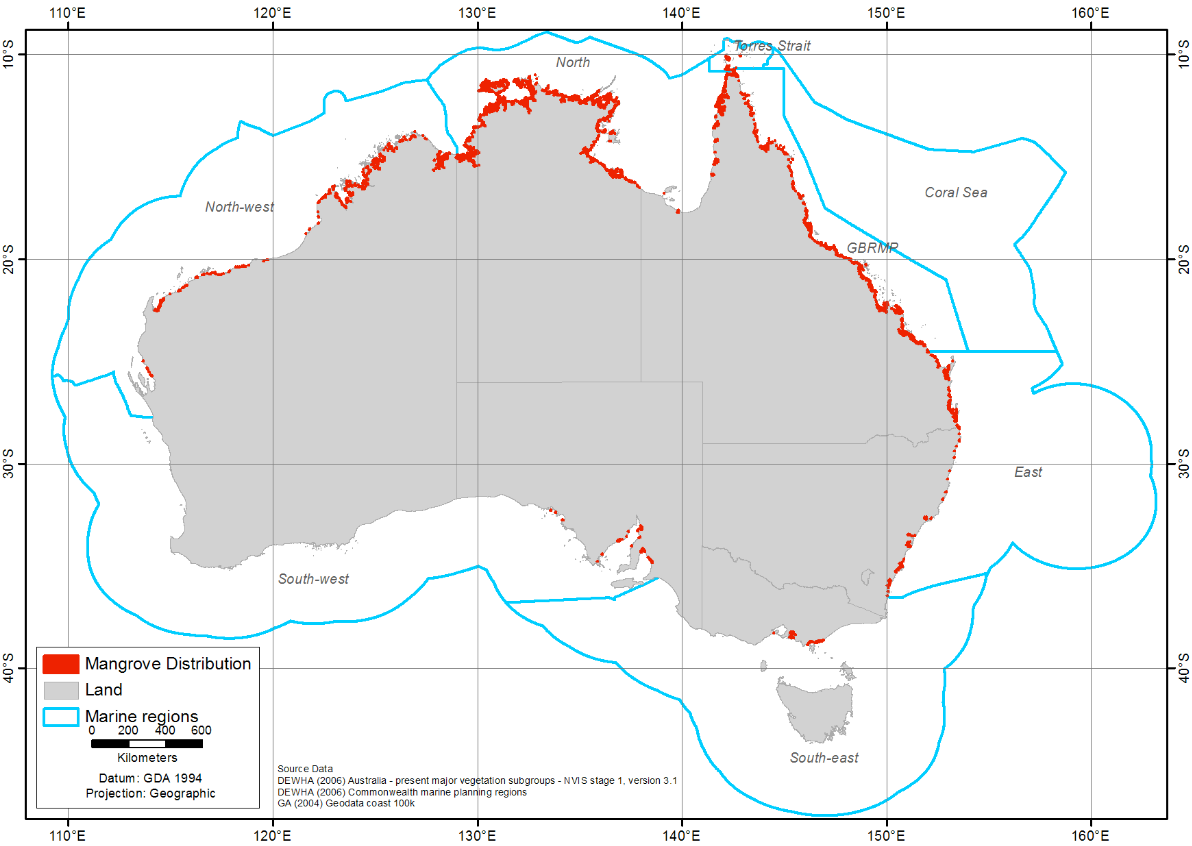
[3] Available from www.ozcoasts.gov.au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South-east
Mangrove Distribution
Land
Marine regions
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Kilometers
Datum: GDA 1994
Projection: Geographic
Source Data
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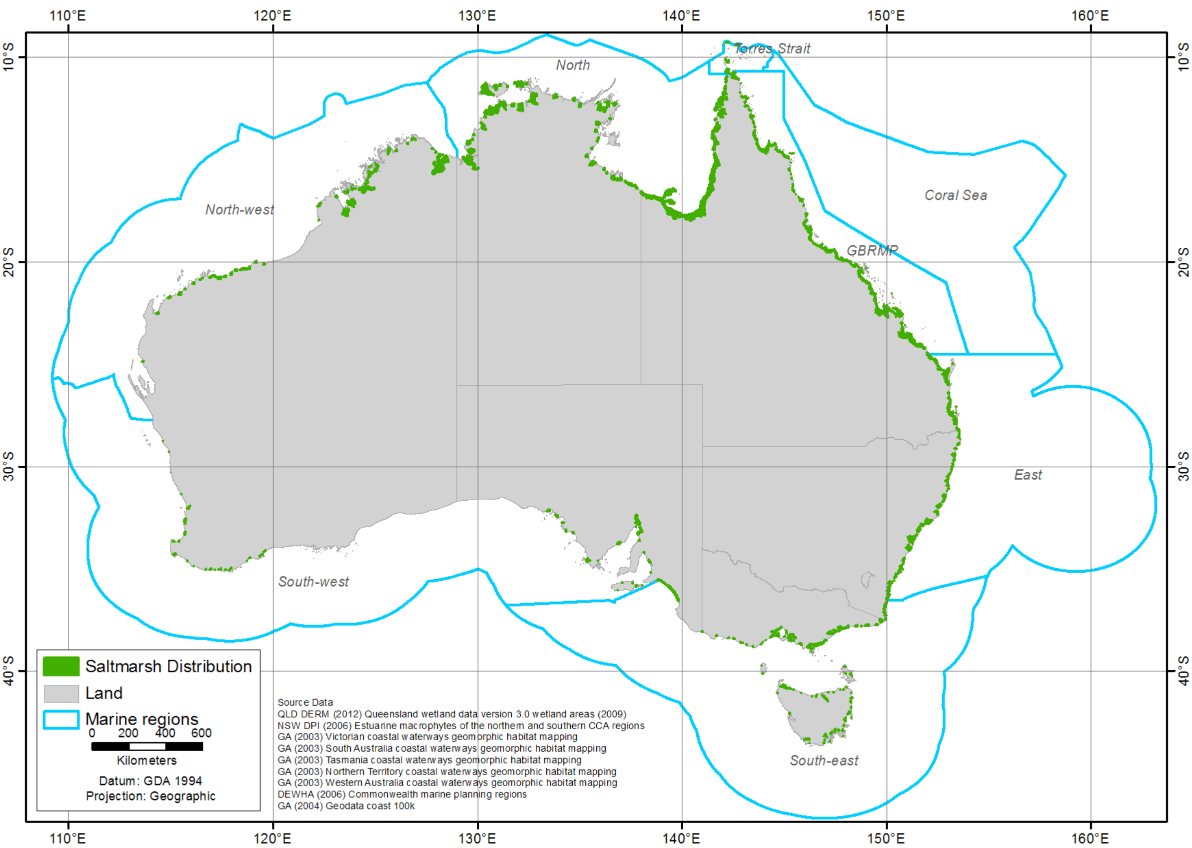
DEWHA (2006) Commonwealth marine planning regions
GA (2004) Geodata coast 100k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South-west
South-east
Saltmarsh Distribution
Land
Marine regions
0 200 400 600
Kilometers
Datum: GDA 1994
Projection: Geographic
Source Data
QLD DERM (2012) Queensland wetland data version 3.0 wetland areas (2009)
NSW DPI (2006) Estuarine macrophytes of the northern and southern CCA regions
GA (2003) Victorian coastal waterways geomorphic habitat mapping
GA (2003) South Australia coastal waterways geomorphic habitat mapping
GA (2003) Tasmania coastal waterways geomorphic habitat mapping
GA (2003) Northern Territory coastal waterways geomorphic habitat mapping
GA (2003) Western Australia coastal waterways geomorphic habitat mapping
DEWHA (2006) Commonwealth marine planning regions
GA (2004) Geodata coast 100k

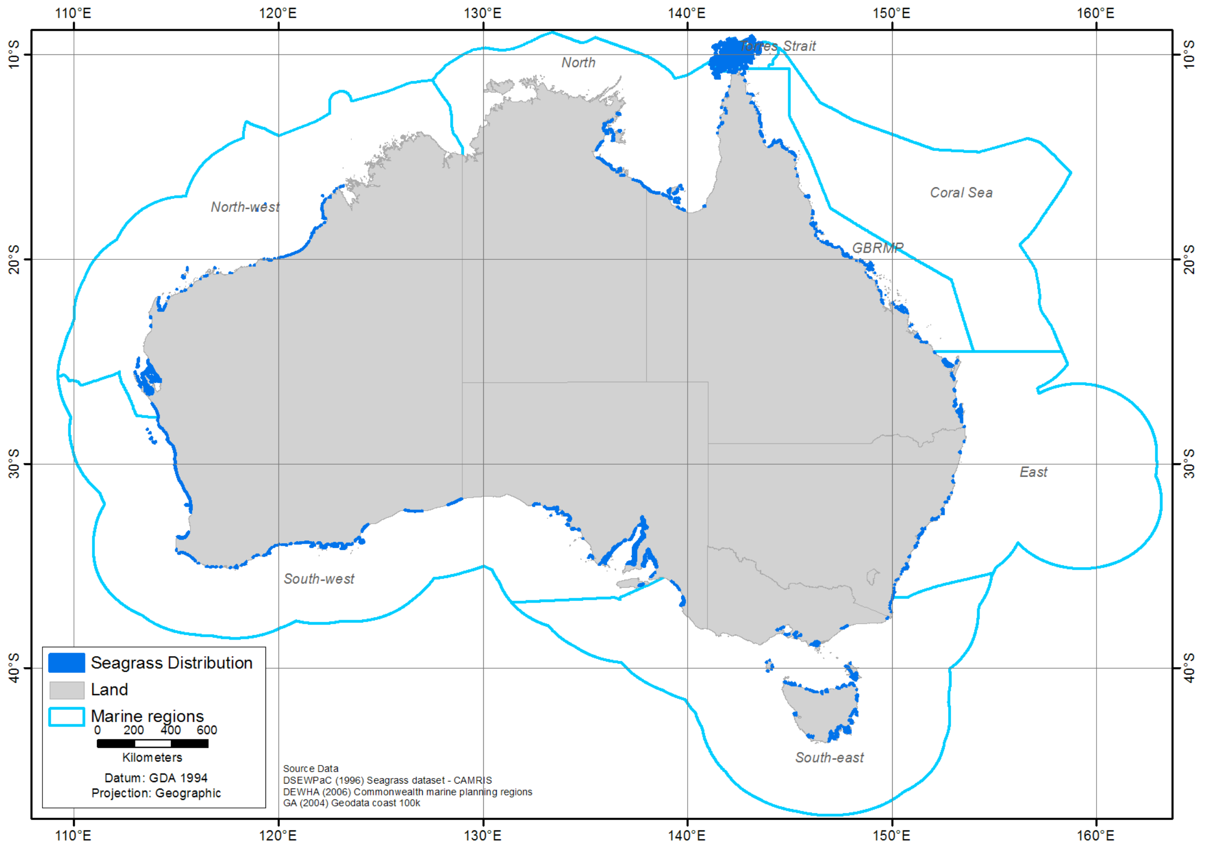

Figure 4. Mangrove, saltmarsh and seagrass dominated areas across Australia produced by query to Geosciences Australia via OzCoasts online coastal information system. Note although coastal wetland ecosystems have been mapped across Australia, the scale and age of maps varies across states, therefore national level maps have been used

2.2 Analytical framework

Stock change and carbon sequestration

Carbon can be stored in ecosystems as a part of the living and dead above and below ground biomass (AGB, BGB) as well as in soils. Carbon sequestration and emissions are measured as changes in carbon stocks over time (usually 5 years or more, IPCC 2007), although direct fluxes of GHG over shorter periods of time, which can then scaled up to annual rates are also used. While assessment of all components at multiple times is optimal for estimating total ecosystem carbon stocks and rates of sequestration/emissions, the complete data sets required to make these calculations are often not available.

Assessment of AGB (e.g. tree biomass) and changes in AGB (i.e. sequestration) is often reported because of the relative ease of measurement, BGB is less often measured because of the difficulty in extracting the biomass of plants from soils. Often allometric relationships, which are established relationships between for example AGB and BGB, are used to estimate BGB (Komiyama et al. 2008). Changes in BGB have not often measured or reported in mangroves, salt marsh or seagrass ecosystems. Assessment of soil carbon is important because soils contain a large proportion of ecosystem carbon stock and may have high rates of carbon sequestration. In both terrestrial and wetland habitats changes in soil carbon stocks over time can be assessed directly or indirectly, for example, by measuring changes in carbon over a chronosequence of years since restoration (Osland et al. 2012). In wetlands increases in carbon sequestered in soils over time have been estimated by measuring changes in soil volume (accretion) over time and extrapolating usinbased on known carbon densities in soils (Howe et al. 2009).

While the carbon stored in wetland soils can be many metres deep, generally only the first metre of soil depth has been considered in carbon estimates (e.g. Fourqurean et al. 2012). This recognises the fact that the top metre of carbon is most at risk after conversion of the ecosystem to other uses and allows for consistent comparisons among habitat types. Following IPCC protocol for tracking changes in carbon stocks (IPCC 2007) and to facilitate comparison among most other carbon assessments, we express ecosystem carbon sequestered or emitted in terms of potential CO_2 emissions (obtained by multiplying changes in carbon stocks by 3.67, the molecular weight ratio of CO_2 to carbon).

Flux and greenhouse gas emissions

Greenhouse gas production from wetland systems differs from that of many terrestrial areas due to the waterlogged nature of wetland soils. Waterlogging limits oxygen (O_2) diffusion into soils and favours the development of anaerobic conditions, which under low salinity conditions are conducive to the production of GHG such as methane (CH_4) and nitrous oxide (N_2O) (Page and Dalal 2011). GHG flux[4] from coastal wetland systems, both in their natural state and following drainage, has not been well accounted for in the carbon accounting process. There is limited data for Australia of baseline emission rates from undisturbed coastal wetland ecosystems in order to assess the effect of any anthropogenic processes on these environments (Page and Dalal 2011). In many freshwater wetland environments, soil carbon sequestration is offset by methane emissions from plant decomposition (Whiting and Chanton 2011). Many coastal wetland ecosystems however are an exception to the rule. Many mangrove forests, saltmarsh and seagrass can have minimal releases of GHG due to the inhibition of methanogenesis by high concentrations of sulfates in seawater (Magenheimer et al. 1996).

There is a wide range of variability in values of the emissions and removals of carbon by terrestrial Australian landscapes (and land uses) reported in the literature. This is partly due to variation in the methodologies used and partly due to variability related to variation in environmental factors (e.g. rainfall, vegetation) and levels of disturbance. For terrestrial ecosystems, existing estimates have been included in the look-up tables where available.

To gauge potential carbon emissions in Australia's coastal wetland ecosystems with disturbance or degradation, a similar methodology to that used by Pendleton et al. (2012) has been applied, using the best available range of data from the literature. Pendleton et al. (2012) use emissions in conjunction with estimates of habitat area, current conversion rate (% of area lost per year), and near-surface carbon stocks susceptible to loss in each of the three habitat types. While we acknowledge that each of the input multipliers has varying degrees of uncertainty owing to ranges reported in the literature or limited available data, unlike Pendleton et al. (2012), we were not able, within the scope of the current project, to undertake any sensitivity analysis to propagate uncertainties.

2.3 Data inputs

Stock change and carbon sequestration

In all cases, the most recent published or grey literature has been used to obtain estimates for Australian key terrestrial ecosystems and land uses, as well as global and regional estimates for coastal wetland ecosystems.

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[4] The net difference between carbon removal and carbon addition to or from the atmosphere or ocean resulting from the balance between carbon uptake by photosynthesis and its release as a result of decomposition.

When comparing the per hectare carbon stocks of ecosystems - for coastal ecosystems we have used values extracted from field studies and for terrestrial systems we have used field data as well as modelled data (see Appendix A, Haverd et al., 2012[a]). The modelled data (1990-2011) likely provides a more accurate picture of carbon stocks over the Australian landscape because it takes in to consideration climatic variability, but no model currently exists for coastal wetland ecosystems.

Following an extensive literature search, we have used available, published data for carbon stored estimates for living biomass and soil, as well as carbon soil sequestration rates for Australian coastal wetland ecosystems. As shown in Table 3 the number of studies available upon which our calculations are based is extremely limited and this has implications in terms of the representativeness of the data for all Australian regions. There are extensive data gaps within regions and habitat types that should be filled to improve our understanding of Australia's coastal ecosystem carbon pools. In addition, for seagrass, the methodologies used to calculate soil sequestration in published and grey literature (as outlined by Sifleet et al.. 2011) vary. For the Australian studies identified a metabolic pathway methodology has been used (Duarte et al.. 2010). This methodology compares annual primary production and community respiration and calculates sequestration from the difference of net photosynthesis and respiration, ie if carbon is captured (photosynthesis) and not respired it is assumed to be sequestered. Another common approach used in the literature is to examine the sedimentary record using radiocarbon dating, however no studies have used this method in Australian seagrasses to date.

While efforts were made to identify data for all key coastal wetland ecosystems, little data was available for *Melaleuca* wetlands and mudflats and so this report has concentrated on carbon stored in AGB and soils of mangroves, saltmarsh and seagrass.

Table 3. Australian ranges and averages of carbon sequestration rates and carbon stocks by coastal habitat type (n= number of studies)

Habitat type	**Living biomass carbon stock (Mg CO_2 eq ha^{-1})**		**Soil carbon stock (Mg CO_2 eq ha^{-1})**		**Living biomass sequestration (Mg CO_2 eq ha^{-1} y^{-1})**		**Soil sequestration (Mg CO_2 eq ha^{-1} y^{-1})**	
	Range	**n**	**Range**	**n**	**Range**	**n**	**Range**	**n**
Estuarine mangroves	63-848 (396)	3	1052-2349 (1710)	2	No data		1-12.3 (5.5)	4
Oceanic mangroves	546-2239 (896)	2	520-2438 (1190)	3	12.8 – 22.6 (17.7)	2	0.2-12.3 (5.2)	2
Saltmarsh	5-116 (52)	1	602-4184 (1646)	4	No data		1.7-7.6 (4.2)	3
Seagrass	0-31 (3)	1	423-1229 (846)	1	No data		-14.6 – 22.8 (-1.1 or 16.1 excluding degraded sites)	2

Area of habitat extent and loss over time

The areal extent of Australian coastal ecosystems was derived from international monitoring databases (Green and Shore 2003, Spalding et al. 2010) and the most recently published literature (Waycott et al. 2009, McKenzie et al. 2010, Creese et al. 2008, Saintilan and Williams 2000, Bucher and Saenger 1991, Sinclair and Boon 2012, Giri et al. 2011, Wilkes 2008,). In most cases, the age of data and the method used differed between States. No attempts were made to standardise these datasets, but rather the latest information identified has been used as the best estimate of areal extent for each habitat type.

Current rates of annual loss (land-use conversion) of Australian habitats were derived from the most recently published literature (Duke 2006, Valiela et al. 2001, Waycott et al. 2009, Saintilan and Williams 2000, West 1992, Sinclair and Boon 2012)

Flux and greenhouse gas emissions

In Australia, as is the case globally, carbon loss per hectare from land use conversion has not been well quantified in coastal wetland ecosystems, but can be estimated. As summaraised in Pendleton et al. (2012) and other literature, the most immediate result of conversion is the loss of vegetation biomass, but there are also losses from the surface sediment carbon pool (often assumed to be in the top 1 m of soil) as well as potentially large, but not well understood carbon losses from deep sediments, depending on the conversion type. As discussed above, given that the first top metre of soil carbon and carbon within the vegetation is most susceptible to land use changes, to be conservative, we have also only focused on stocks within these two main pools (biomass and top 1 m of soils). We were not in a position to undertake any sensitivity analysis for the uncertainty within the datasets available. Therefore, to be conservative, using the mean emissions rates derived from the best data available, and based on the calculations of Page and Dalal (2011) that 25% loss of organic carbon occurs from the top 1m in the first 50 years following drainage, we have assumed a scenario of 25% loss of emissions to the atmosphere and no re-burial of disturbed material from land use changes to coastal wetland ecosystems. A worst case scenario of 100% loss would apply if land use change resulted in conversion to a qualitatively different ecosystem state that removed and prevented soil carbon recovery. Using the current data it is not possible to assess the effects of variation in land use change on carbon stocks and emissions in Australia. It is likely however, that not all conversions have resulted in 100% loss of carbon, with some activities only having limited impact where retention, burial, or redistributed soil carbon has occurred, which would indicate emissions could be considerably less, hence our conservative approach.

In Australia, the effects of altering drainage and hydrology of saltmarsh for conversion to arable land for agricultural, industrial, port and residential development last for decades and can lead to the loss of several metres of sediment, along with its carbon, from oxidation (e.g. similar to that reported in the USA, Crooks et al. 2011). While many mangroves are located in sparsely populated regions in northern Australia and remain in a near pristine condition, reclamation landfill for expansion of population centres and port development can expose large portions of sediment carbon to oxygen. In seagrass systems, reduction in water quality due to excess nutrients or sediments from catchment-based sources is a leading cause of ecosystem decline and loss. Loss of living biomass results in exposure of sediment carbon to the water column and to resuspension where it can be oxidized and liberated to the atmosphere (Fourqurean et al. 2012). Direct impacts of activities such as dredging, trawling, and anchoring also affect seagrass beds and can result in emissions of stored carbon (Orth et al. 2006). Nutrient enrichment can affect decomposition of organic matter and thus may result in changes in sequestration and/or emissions. Although nutrient enrichment is a threat in many of Australia's estuaries there are currently no data to determine how this may influence sequestration or emissions from mangroves and this it is therefore not included in this report.

We have focused on the total amount of CO_2 that could be released from annual rates of conversion for each ecosystem, but have not attempted to estimate over what time scale these releases would be made. At the scale of the individual site, the rate of release likely follows a negative exponential curve with time—initially high and tapering in later years (Lovelock et al. 2011). Some studies suggest that the temporal dynamic of soil carbon pools after conversion may have a half-life on the order of 5–10 years (Lovelock et al. 2011), however any assumption of the temporal period of release within a degraded site is not critical to the results of this analysis. As indicated by Pendleton et al. (2012) globally, and assuming the scale of impact from land use conversion is at the low to medium end, and Australian coastal wetland ecosystem conversion rates are stable or increasing over time, the total amount of carbon released annually from all land use changes would be greater than or equal to our estimates.

Following Pendleton et al. (2012) we have also estimated the cost to the global economy of the estimated emissions resulting from coastal wetland ecosystem conversion for comparison to global estimates. In lieu of Australian estimates of the global social cost of carbon (SCC), national emissions estimates for each ecosystem were multiplied by the mean estimate of the global SCC of $41 per tonne of CO_2 (2007 U.S. dollars) (USG 2010). The social cost of carbon (SCC), is defined as the marginal value of economic damages of the climate change attributable to an additional tonne of CO_2 in the atmosphere in 2020 (2007 dollars) (USG 2010). The SCC calculation provides an estimate of the environmental damages that can be avoided by reducing emissions, but does not necessarily equal the price that the market will pay for reducing emissions, since that market price is determined by the avoided cost of regulatory controls on carbon and not avoided damages (Pendleton et al. (2012).

Although there are a few studies of CH_4 and N_2O in Australia under nutrient enrichment for coastal wetland ecosystems, we have included what information is available in the look-up table in Appendix A and summarised this in Section 3. In estimating emissions associated with land use changes to these ecosystems however, we have not included any changes in GHG such as methane (CH_4) or nitrous oxide (N_2O) as in highly saline wetlands (>18 ppt), as outlined above, these emissions are likely to be low.

2.4 Carbon stocks

Australia's coastal ecosystems store around 5 times the amount of carbon in their soils per hectare compared to terrestrial ecosystems including forests.

In ecosystems carbon is stored in both living and dead biomass and in soils. However, it is the location of stored carbon within an ecosystem that determines how efficient it is as a sink for atmospheric carbon. While great emphasis is placed at an international level on the role of living biomass[5] in storing atmospheric carbon, sustaining or enhancing soil carbon stocks is becoming a priority due to increasing land use change and land use intensification to meet global demands for food, water and energy. During the past 25 years, one-quarter of the global land area has suffered a decline in productivity and in the ability to provide ecosystem services due to soil carbon losses (Bai et al. 2008). Since the 19th century, around 60% of the carbon in the world's soils and vegetation, has been lost due to land use change (Houghton 1995). In Australia, conversion of natural ecosystems for agriculture has resulted in a 40-60% reduction in organic carbon stocks (Sanderman et al. 2009). Because soil carbon is central to agricultural productivity, climate stabilisation and other vital ecosystem services, creating policy incentives, such as through the Carbon Farming Initiative to encourage the sustainable management of soil carbon could deliver numerous short and long-term

[5] Defined by the IPCC as organic material both above-ground and below-ground, and both living and dead, e.g., trees, crops, grasses, tree litter, roots etc and includes above and below ground biomass

benefits to Australia.

Globally, it is recognised that soils contain the largest carbon reservoir on Earth, with the top metre of the world's soils storing approximately 2,200 billion Mg of carbon, two-thirds of it in the form of organic matter (Batjes 1996). This is more than three times the amount of carbon held in the atmosphere (IPCC 2007). For Australia, managing soils so that carbon stocks (and other ecosystem services provided) are sustained and even enhanced is of crucial importance if we are to meet near-term challenges and conserve this valuable resource for future generations (UNEP 2012).

It is estimated that converting land from native vegetation to agriculture dramatically reduces soil carbon (depending on the soil type this can be between 20–70% (Luo et al. 2010, Sanderman et al. 2010)). Much of this loss in soil organic carbon can be attributed to reduced inputs of organic matter (Post and Kwon, 2000). The nature of Australian soils (old and weathered) means they often have a naturally lower soil organic carbon content than European or North American counterparts (Spain, 1990).

In Australia the decline in soil carbon on agricultural land is also exacerbated by conventional tillage practices (working the soil to remove the plant residue from the previous crop and produce a fine seedbed). A CSIRO study on the potential for agricultural soils to sequester carbon found that on average Australian cropping lands are still losing carbon, but that more modern forms of land management can increase soil carbon stocks (Sanderman et al. 2010). Larger soil carbon gains are possible with more radical schemes, such as conversion of cropping land to permanent pasture and restoration of natural vegetation, but these may adversely affect agricultural productivity. Studies suggest that it is unlikely that soil carbon can be returned to be equivalent to natural ecosystems with continued productive agriculture, but with changed practices they could reach 60-75% of the soil carbon originally present (Lal, 1999).

The recognition that increasing soil carbon can play a significant part in the reduction of GHG emissions, providing a low cost sink as well as other co-benefits for farm productivity has led policy makers to investigate schemes to restore soil carbon stocks. One of these, the Australian Government's Carbon Farming Initiative (CFI) provides opportunities for rural producers to earn additional income from carbon credits. However the CFI does not include mechanisms to protect or enhance other soils, such as those found in coastal wetland ecosystems, which also have important soil carbon stocks.

The total amount of carbon in terrestrial Australian landscapes (includes forests, woodlands, swamps, grasslands, farmland, soils, and derivatives of these carbon stocks, including biochar and biofuels but excluding coastal wetland ecosystems) is approximately 104 billion Mg CO_2 eq, which is partitioned almost equally between native forests and woodland and grassland and cropland (Wentworth Group 2009). In the case of forests and woodlands, the majority of this is stored in above ground biomass[6]. For example, the temperate forests of southern Australia, which have the highest recorded forest biomass, store nearly 80% of the carbon above ground (Keith et al. 2009). By contrast, we estimate that Australia's coastal wetland ecosystems store 5 billion – 22 billion Mg CO_2 eq[7], of which most of the carbon is stored in soils (5 billion – 20 billion Mg CO_2 eq, mean 11 billion Mg CO_2 eq).

As the majority of terrestrial tree carbon storage is generally above ground in Australia and subject to natural events such as drought and fire, and land use change, carbon loss both above and below ground can be high. Haverd et al. (2012[b]) found that annual mean net ecosystem productivity of Australian terrestrial ecosystems modelled over the period 1990-2011, was 294 Tg CO_2 eq yr^{-1}, which was accompanied by net losses from fire and land use change, of 180 Tg CO_2 eq yr^{-1}. The majority

6 Defined by the IPCC as all living biomass above the soil including stem, stump, branches, bark, seeds and foliage

7 Refer Appendix A for data used in these calculations

of coastal wetland ecosystem related carbon however, is stored in organic-rich sediments that may be several metres deep[8], which means even when the near surface soil is subject to some natural disturbance from cyclones and storms, the majority of carbon in the soil will remain locked up for millennium due to low-oxygen conditions and other factors that inhibit decomposition at depth (Kristensen et al. 2008). When healthy, coastal wetland ecosystems continuously store carbon in their soils over long time scales, unlike terrestrial soils, which tend to plateau over time (Schlesinger and Lichter 2001).

Howe et al. (2009) estimated that the estuary wide carbon storage capacity of the Hunter River estuarine habitats in NSW to be at 2570–3,670 Gg carbon, or about 0.005% of the worldwide estimates of tidal, saline wetlands.

When looking at storage in soils on a per hectare basis (Table 4), Australia's coastal wetland ecosystems appear to be superior to our terrestrial habitats, at around 5 times the amount stored per hectare in terrestrial habitats, including forests (refer Figure 5), however the variation about these values is high.

On a per hectare basis, the average estimates for carbon stocks for Australian coastal wetland ecosystems for living biomass and soil organic carbon based on the limited published data available, are generally similar to or slightly higher than global averages (Table 4). This is recognising however, that we have few studies and therefore do not necessarily provide representative estimates of carbon stocks from all regions. The majority of the data available for Australian mangroves has been collected within the north, east and southeast regions. For saltmarsh, only carbon stocks from the southeast region were found. Carbon stock data for seagrass is generally more widely geographically distributed than for the mangroves and saltmarsh. Further research is critically needed across all regions and there is a need to increase the sample size to improve estimates of the carbon stocks within coastal wetland ecosystems in Australia.

Table 4 also shows carbon stocks from studies undertaken at a range of terrestrial locations (see Appendix A). The values for terrestrial ecosystems are considerably higher than global averages in some cases. For example for Australian temperate forests, living biomass carbon stocks are considerably higher than the global average (Keith et al. 2009). Results from the model developed by Haverd et al. (2012 [a,b]) which uses a land surface model of Australian terrestrial carbon and water cycles to calculate carbon pools and fluxes over the Australian landscape for the period 1990-2011, show lower estimates for living biomass that those from field plot data, but are probably more representative of carbon stocks on a landscape scale (Havard et al. 2012). The differences in soil carbon between the observed and model data are likely due to model estimating total carbon while measured values are from the top 30 cm.

[8] Note that while the carbon stored in the soil can be many metres deep, only the first metre of soil depth has been considered so as to allow for consistent comparisons among habitat types and in recognition of the fact that the top metre of carbon is most at risk after conversion of the ecosystem to other uses.

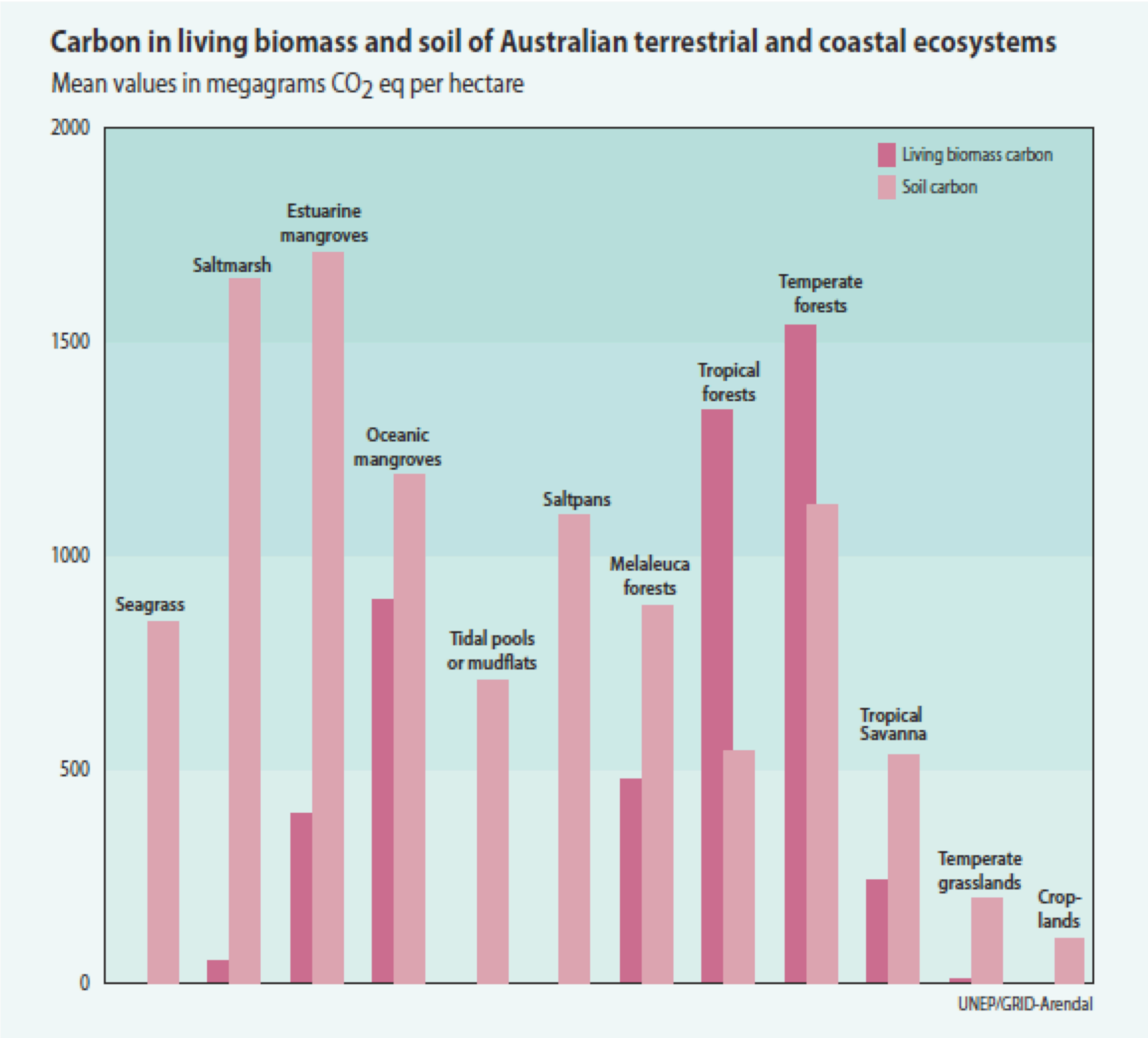

Figure 5 Carbon stocks (soil organic carbon and living biomass) for Australian terrestrial and coastal systems Soil carbon in coastal ecosystems has been standardised to 1 m depth. However in terrestrial ecosystems it has been standardised to 0.3m[9]. Terrestrial carbon mean values taken from plot based data (see Appendix A).

[9] The IPCC (2006) recommends sampling the top 0.3-m depth of soil for soil organic carbon stock assessments as changes in soil organic carbon due to land-use change or management are primarily confined to the top 0.1- or 0.3-m depths in most soils. (e.g. Wilson et al.et al., 2002, found that land use induced change was most apparent in the near surface layer at a site in NSW). In terrestrial soils, carbon content tends to decrease rapidly with depth (e.g. Chen et al.et al., 2005). In addition the carbon depth relationship in terrestrial soils is strongly influenced by vegetation type and can show a wide variability (Jábbogy and Jackson, 2000)

Table 4: Comparison of carbon storage capacity of Australian habitats against global ranges of carbon stocks

Habitat Type	Global range Living biomass (Mg CO_2 eq ha^{-1})	Global range Soil organic carbon (Mg CO_2 eq ha^{-1})	Australian range (mean) Living biomass (Mg CO_2 eq ha^{-1})	n	Australian range (mean) Soil organic carbon (Mg CO_2 eq ha^{-1})	n	Australian mean (modeled, ecoregions) Living biomass (Mg CO_2 eq ha^{-1})	Australian mean (modeled, ecoregions) Soil organic carbon (Mg CO_2 eq ha^{-1})
Seagrass	0.4 - 18.3	66 - 1467	0 – 31 (3)	1	423 - 1230 (846)	1	n/a	n/a
Saltmarsh	12 - 60	330 - 4436	5 – 116 (52)	1	602 - 4184 (1646)	4	n/a	n/a
Estuarine Mangroves	237 - 563	1060	63 – 848 (396)	3	1052 – 2349 (1710)	2	n/a	n/a
Oceanic mangroves	237 - 563	1690 - 2020	546 – 2239 (896)	2	520 – 2438 (1190)	3	n/a	n/a
Tidal pools/ mudflats			n/a		710	1	n/a	n/a
Saltpans			n/a		1095	1	n/a	n/a
Melaleuca forests			477	1	881	1	n/a	n/a
Tropical forests	442	450	1226-1457 (1341)	2	528-558 (543)	2	700	1178
Temperate forests	208	353	1061-2937* (1539)	8	451-1118 (1118)	8	457	1680
Tropical savannas	108	431	160-290 (240)	4	513-554 (534)	2	131	486
Temperate grasslands	26	866	5-15 (13)	3	83-341 (200)	10	97 (includes shrublands	373

Habitat Type	**Global range Living biomass** (Mg CO_2 eq ha^{-1})	**Global range Soil organic carbon** (Mg CO_2 eq ha^{-1})	**Australian range (mean) Living biomass** (Mg CO_2 eq ha^{-1})	**n**	**Australian range (mean) Soil organic carbon** (Mg CO_2 eq ha^{-1})	**n**	**Australian mean (modeled, ecoregions) Living biomass** (Mg CO_2 eq ha^{-1})	**Australian mean (modeled, ecoregions) Soil organic carbon** (Mg CO_2 eq ha^{-1})
Deserts and semideserts	6	154	n/a		n/a		30	129
Montane Grasslands and Shrublands	n/a	n/a	n/a		n/a		609	2128
Croplands	7	294	n/a		54-229 (106)	22	n/a	n/a
Australia			n/a		n/a		106	427

Source: Global estimates from Murray et al. 2010, and IPCC 2000. Refer Appendix A for Australian estimates data. * The maximum biomass recorded is in *Eucalyptus regnans* (Mountain Ash). These temperate forests have the highest recorded biomass carbon density in the world (Keith et al., 2.009). Note n = number of studies. Modeled data –source Haverd and Briggs see Appendix A, see Figure 6 for ecoregions

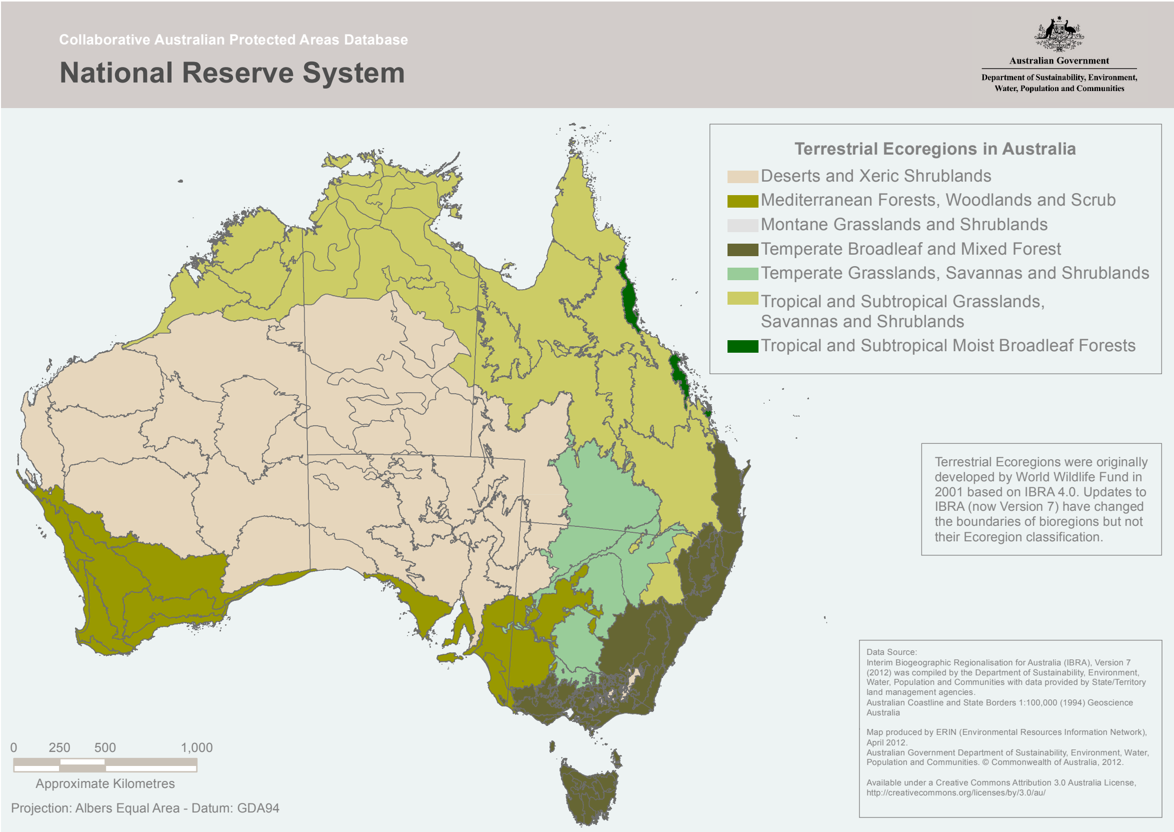

Figure 6 Map of Australian ecoregions used in Table 4 (Source Department of Sustainability, Environment, Water, Populations and Communities (adapted from WWF)[10]

2.5 Long term carbon burial

For Australia the annual rates of carbon burial in soils of all coastal wetland ecosystems is up to 66 times greater than the model results for Australian terrestrial ecosystems.

Carbon sequestration is the process of removing carbon dioxide (CO_2) from the atmosphere and depositing it within a reservoir[11]. Carbon influx and residence time are two key factors for determining the carbon sequestration capacity of an ecosystem (Luo et al. 2003). Ecosystems usually store more carbon if they have high rates of carbon influx via photosynthesis where carbon is incorporated into both above and belowground biomass (autochthanous) and eventually into soils through incorporation of detritus, and have longer carbon residence times, i.e. they can lock up the carbon through burial. Some ecosystems such as coastal wetlands can also trap external carbon that enters the ecosystem (allochthanous), for example, through deposition of upstream organic matter and sediments into downstream mangrove or saltmarsh soils. It is generally acknowledged that carbon stored in soil has a longer residence time than above ground biomass. The soil carbon stock can be divided into three compartments according to how fast the carbon breaks down and is replaced. These compartments are - fast (e.g. annual), slow (e.g. decadal) and passive (e.g. millennial). For carbon sequestration (long-term storage) and for carbon trading purposes, it is most effective to increase the total amount of carbon in the stocks that break down slowly (e.g. the slow and passive stocks) (Walcott et al., 2009).

[10] available from http://www.environment.gov.au/parks/nrs/science/bioregion-framework/terrestrial-habitats.html)

[11] IPCC glossary of climate change acronyms http://unfccc.int/essential_background/glossary/items/3666.php#S

The rate of net primary production of all ecosystems however varies based on vegetation size and age as well as the balance between carbon production and respiration. Carbon accumulation in terrestrial forests and soils eventually reaches a saturation point, beyond which additional sequestration is no longer possible or occurs at very low rates (Magnini et al. 2000). This happens, for example, when trees reach maturity or when soil nutrients are depleted, although some studies have shown that some old-growth forests continue to accumulate carbon in their soils (Guoyi Zhou 2006, Keith et al. 2009). In intertidal coastal wetland ecosystems, the amount of soil carbon increases over time as the elevation of the soil surface increases with rising sea level. Increases in the elevation of soil surfaces occur as sediments and organic matter are accreted on the soil surface as well as through addition of organic matter from roots. It is this characteristic that gives them an advantage over terrestrial systems in terms of their ability to continue to sequester carbon (Alongi 2011) with the rate of carbon sequestered and the size of the carbon pool potentially continuing to increase over long time frames (Chmura et al. 2003). Current data indicates that for Australian mangroves the rates of surface elevation (increase in the elevation of the soil surface) matches or exceeds the local rate of sea level rise (Rogers et al. 2006; Lovelock et al. 2011a), however it is likely that this will be at the expense of saltmarsh (Saintilan and Williams 2000, Wilton 2002, Rogers et al. 2006, Trail et al. 2011). On-going destruction and damage of coastal wetland ecosystems to make way for coastal development and expansion however, impacts on their ability to function as long-term carbon sinks.

The quantity of CO_2 removed from the atmosphere and trapped in natural habitats on an annual basis is known as an annual carbon sequestration or burial rate. Trees, plants and crops remove carbon from the atmosphere through photosynthesis by absorbing and storing it in their biomass (tree trunks, branches, foliage and roots) and within soils. In terms of above ground carbon sequestration, of the Blue Carbon sinks only mangroves provide a significant contribution to removing carbon from the atmosphere with long term rates of sequestration from wood production estimated between 12.8 - 22.6 Mg CO_2 eq ha^{-1} yr^{-1} (Table 5). This compares to terrestrial ecosystems which provide rates estimated between 3.7 – 40.9 Mg CO_2 eq ha^{-1} yr^{-1}.

While mangroves, saltmarsh and seagrass have a much smaller areal extent than terrestrial forests, grasslands and croplands in Australia, their total contribution to long-term carbon sequestration exceeds the size of carbon sinks in some terrestrial ecosystem types and to activities that aim to improve soil carbon within agricultural systems on an annual per hectare basis. On a per hectare basis, the rates of carbon sequestration in biomass and soils of mangroves are in line with mean global estimates, but below that for saltmarsh and seagrass (Table 6). Saltmarsh global data estimates a mean of around 8 Mg CO_2 eq ha^{-1} yr^{-1} and it is likely that the Australian data is not representative given the limited studies (n = 3) that are all in south eastern Australia where accretion rates are low. Global carbon sequestration data for seagrass range from -77 to 85 Mg CO_2 eq ha^{-1} yr^{-1} with and average around 5.1 Mg CO_2 eq ha^{-1} yr^{-1}. A large number of estimates of carbon sequestration in seagrass show annual net losses of carbon (McCleod et al.. 2011, Sifleet et al. 2011). The Australian data available also showed net loss of carbon, although this data comes from one study which sampled degraded coastal lagoons in Sydney (Eyre and Ferguson (2002), cited in Duarte et al. 2010), which brought the per hectare carbon sequestration rate of seagrass to an average of -1.1 Mg CO_2 eq ha^{-1} yr^{-1}t. There is an urgent need to increase understanding of carbon sequestration in a wide range of Australian seagrass ecosystems.

The total (biomass + soils) national carbon sequestered in mangroves is estimated at 22.9 Tg CO_2 eq y^{-1} but robust rates cannot be estimated for saltmarsh and seagrass as there are insufficient data available for carbon accumulated in living biomass in saltmarsh or seagrass. For soils, mangroves, saltmarsh and seagrass are estimated at 5.3 Tg CO_2 eq y^{-1}, 5.7 Tg CO_2 eq y^{-1}, and 154 Tg CO_2 eq y^{-1} (or -10.3 Tg CO_2 eq y^{-1} including degraded sites) respectively (Table 5). This compares to terrestrial ecosystem national total and soil carbon burials of 283 Tg CO_2 eq y^{-1} and 145 Tg CO_2 eq y^{-1} respectively.

While the total and soil carbon burial rate for terrestrial ecosystems are significant, Sanderman et al. (2010) suggested that Australian terrestrial soils may only be mitigating losses and not sequestering additional atmospheric carbon. For cropland, included within the Carbon Farming Initiative where estimates of potential increases in carbon sequestration from management of cropland vary, a review by Sanderman et al. (2010) found that a variety of options such as enhanced rotation and no-till or stubble retention resulted in relative gains of 1.1 -1.4 Mg CO_2 eq ha^{-1} y^{-1} compared to conventional management. However, when examined over time, even the improved management often showed declines in absolute soil carbon stocks. In addition they found that the relative gains diminished with trial duration, with the largest gains in the first 5-10 years.

For forests, Schlesinger and Lichter (2001) suggested that long-term carbon sequestration in forest soils was unlikely in a rising CO_2 atmosphere, with most of the CO_2 taken up in short lived tissues and that there was no net accumulation of carbon into the deeper soil layers.

The global soil rates for coastal wetland ecosystems are about two to four times greater than global rates observed in mature tropical forests (1.8–2.7 Mg CO_2 eq ha^{-1} y^{-1} (Lewis et al. 2009, Schlesinger and Lichter 2001). For Australia the annual rates of carbon burial in soils of all coastal wetland ecosystems reviewed ranged between 0.2 and 16.7 Mg CO_2 eq ha^{-1} y^{-1} (Table 5) which is up to 66 times greater than the model results from Harverd and Briggs (Table 5) that estimated soil carbon burial rates across Australian terrestrial ecosystems of between -0.5 - 0.3 Mg CO_2 eq ha^{-1} y^{-1} (Table 5).

Coastal wetland ecosystems comprise less than 1% of the total area examined in Table 5, however they contribute around 36% of the annual carbon sequestered into soil. Therefore, even with the smaller above ground biomass and areal coverage of coastal wetland ecosystems in Australia, they have the potential to contribute substantially to long-term carbon sequestration of Australia, mainly resulting from the higher rate of organic carbon sequestration in soils (Figure 7).

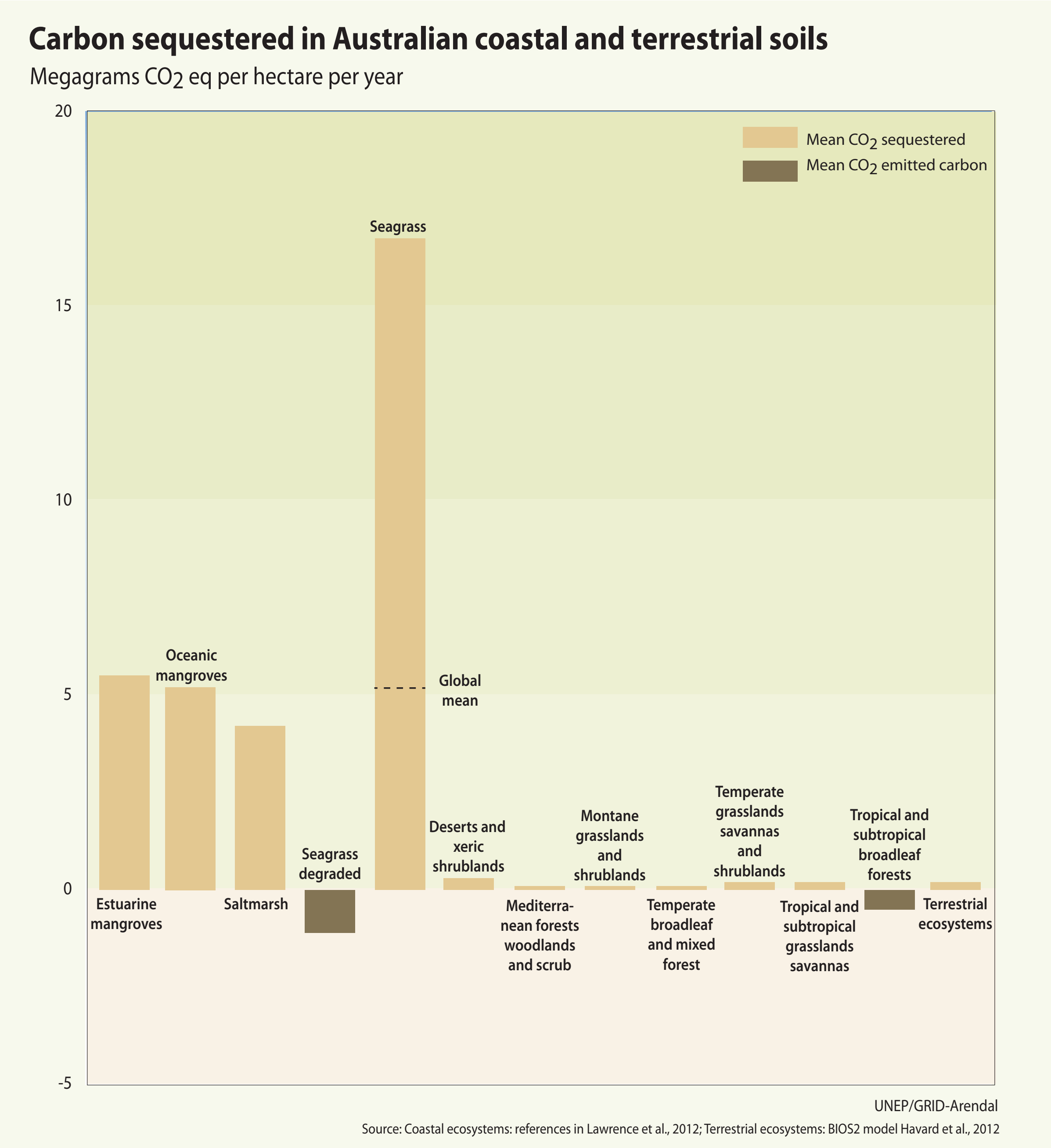

Figure 7. Coastal wetland ecosystems sequester large amounts of carbon. Australian mean long-term rates of carbon sequestration in soils in terrestrial forests and coastal wetland ecosystems. Note terrestrial values are modeled values. Because seagrass carbon sequestration values are highly variable in the Australian data set the global mean has also been included.

Table 5. Carbon burial for Australian ecosystems and land-uses Terrestrial ecosystem values have been provided by Vanessa Haverd and Peter Briggs for the period 1990-2011 (see Haverd et al.. (2012[a]) for details of model methodology)

Habitat type	**CO_2 accumulated in Living Biomass** **Mg CO_2 eq ha^{-1} y^{-1}**		**National CO_2 accumulated in living biomass** **Tg CO_2 eq y^{-1}**	**Carbon burial rate (Soils)** **Mg CO_2 eq ha^{-1} y^{-1}**		**National carbon burial (Soils)** **Tg CO_2 eq y^{-1}**		**Total national carbon burial** **Tg CO_2 eq y^{-1}**
	Range (Mean)	**n**	**Range (Mean)**	**Range (Mean)**	**n**	**Area (ha)**	**Range (Mean)**	**Range (Mean)**
Estuarine mangroves	No data		No data	1 - 12.3 (5.5)	4	990,800	0.2 – 12.2 (5.3)	12.9 – 34.6 (22.9)
Oceanic mangroves	12.8 - 22.6 (17.7)	12	12.7 – 22.4 (17.6)	0.2 - 12.3 (5.2)	2			
Saltmarsh	No data		No data	1.7 - 7.6 (4.2)	3	1,376,500	2.3 – 10.5 (5.7)	No data
Seagrass	No data		No data	-14.6 – 22.8 (-1.1 or 16.7 excluding degraded sites) *Global rate 5.1 (McCleod et al. 2011)	2	9,256,900	-134.7 – 210.8 (-10.3 or 154.6 excluding degraded sites)	No data
Deserts and Xeric	2.5		(69.30)	0.6		356,980,528	(90.8)	(160.1)

Habitat type	**CO_2 accumulated in Living Biomass** **Mg CO_2 eq ha^{-1} y^{-1}**		**National CO_2 accumulated in living biomass** **Tg CO_2 eq y^{-1}**	**Carbon burial rate (Soils)** **Mg CO_2 eq ha^{-1} y^{-1}**		**National carbon burial (Soils)** **Tg CO_2 eq y^{-1}**		**Total national carbon burial** **Tg CO_2 eq y^{-1}**
	Range (Mean)	**n**	**Range (Mean)**	**Range (Mean)**	**n**	**Area (ha)**	**Range (Mean)**	**Range (Mean)**
Shrublands	(3.7)			(0.3)				

Habitat type	**CO_2 accumulated in Living Biomass $Mg\ CO_2\ eq\ ha^{-1}\ y^{-1}$** Range (Mean)	n	**National CO_2 accumulated in living biomass $Tg\ CO_2\ eq\ y^{-1}$** Range (Mean)	**Carbon burial rate (Soils) $Mg\ CO_2\ eq\ ha^{-1}\ y^{-1}$** Range (Mean)	n	**National carbon burial (Soils) $Tg\ CO_2\ eq\ y^{-1}$** Area (ha)	Range (Mean)	**Total national carbon burial $Tg\ CO_2\ eq\ y^{-1}$** Range (Mean)
Mediterranean Forests Woodlands and Scrub	6.5 (10.5)		(12.30)	1.5 (0.1)		78,295,156	(5.3)	(17.6)
Montane Grasslands and Shrublands	16.2 (32.2)		(0.62)	2.1 (0.1)		1,232,981	(0.1)	(0.72)
Temperate Broadleaf and Mixed Forest	6.7 (28.7)		(0.07)	2.0 (0.1)		55,264,938	(4.7)	(4.77)
Temperate Grasslands Savannas and Shrublands	11.9 (10.9)		(9.84)	2.1 (0.2)		52,977,918	(9.1)	(18.94)
Tropical and Subtropical Grasslands Savannas	9.5 (16.1)		(45.70)	2.5 (0.2)		220,624,008	(38.6)	(84.3)

Habitat type	CO_2 accumulated in Living Biomass Mg CO_2 eq ha^{-1} y^{-1}		National CO_2 accumulated in living biomass Tg CO_2 eq y^{-1}	Carbon burial rate (Soils) Mg CO_2 eq ha^{-1} y^{-1}		National carbon burial (Soils) Tg CO_2 eq y^{-1}		Total national carbon burial Tg CO_2 eq y^{-1}
	Range (Mean)	**n**	**Range (Mean)**	**Range (Mean)**	**n**	**Area (ha)**	**Range (Mean)**	**Range (Mean)**
Tropical and Subtropical Broadleaf Forests	10.2 (40.9)		(0.05)	3.7 (-0.5)		3,453,315	(-1.9)	(-1.85)
Terrestrial Australia	4.6 (10.6)		(138.00)	1.2 (0.2)		768,828,843	(145.0)	(283)

Note – mean estimates in brackets and terrestrial systems living biomass values represent NPP (provided by Vanessa Haverd and Peter Briggs at CSIRO)

Table 6 Comparison of global and annual soil carbon sequestration rates for coastal wetland ecosystems (Source: Sifleet et al. 2011 and McLeod et al.. 2011 for global estimates and Appendix A for Australian estimates)

Habitat type	**Global annual soil carbon sequestration mean rate (Mg CO_2 eq ha^{-1} yr^{-1})**	**Australian annual carbon sequestration rate (Mg CO_2 eq ha^{-1} yr^{-1})**
Estuarine mangroves	6.3	5.5
Oceanic mangroves	6.3	5.2
Saltmarsh	8.0	4.2
Seagrass	5.1	-1.1 or 16.7 (excluding degraded sites)

3 Optimising coastal carbon

3.1 Managing coastal carbon

Degraded coastal ecosystem contribution to climate change

While the combustion of fossil fuels is primarily responsible for anthropogenic contributions to atmospheric greenhouse gases (GHG), land-use activities, especially deforestation, are also a major source of GHG, accounting for ~8–20% of all global emissions (van der Werf et al. 2009). New evidence indicates that it is not just landuse conversion associated with terrestrial forests that can act as a source of GHG, but also land-use conversion of saltmarsh, mangroves, and seagrass. These emissions are so far relatively unappreciated or even neglected in most policies relating to climate change mitigation (Climate Focus 2011) and this is certainly the case in Australia. The potential magnitude and economic impact of these previously unaccounted emissions have been estimated below for Australia and compared with global estimates summarised in Pendleton et al. (2012).

Areal extent of habitat loss

Fundamental to identifying opportunities to prevent the release of buried carbon from coastal wetland ecosystems is an understanding of the location, areal extent, and conversion rates. Global data shows that seagrass, saltmarsh and mangroves are being degraded or destroyed at a rapid pace, around 1-2% per year (Murray et al. 2010). Estimating the rate of loss for Australian mangroves, saltmarsh and seagrass is difficult given the paucity of up to date information available on the extent of loss. Based on published data, Australia seems unexpectedly to fall within the annual global loss range for saltmarsh and mangroves (1-2% and 2% respectively), but as expected, is lower for seagrass (0.05%) (Table 7).

Table 7. Coastal wetland ecosystems: National areal extent and conversion rates (Global information included for comparison extracted from Murray et al. 2010)

Habitat type	Extent (ha)	Conversion drivers	Annual loss rate	Total historic loss (%)
Global seagrass	30 million – 60 million	Water quality degradation, mechanical damage	1.2%-2% (≈1980-2000)	29%
Australian seagrass	**Total 9,256,900** WA 2,500,000 (Green and Shore 2003) NT/Gulf of Carpentaria (west) 77,900 (Green and Shore 2003) Gulf of Carpentaria (east) 40,900 (Green and Shore 2003) Torres Strait 1,720,600	Increasing human population densities - Industrial development, nutrient loading near population centres, port development, coastal agriculture and fisheries. (Duarte et al. 2008)	0.05% (1930-2000 – (Waycott et al. 2009 supp data)	20,030 ha (Waycott et al. 2009 supp data 1930-2005 datasets) 45,000 ha from human loss and 100,000 ha from natural disasters between 1987-

Habitat type	Extent (ha)	Conversion drivers	Annual loss rate	Total historic loss (%)
	(McKenzie et al. 2010) QLD east coast 3,807,900 (McKenzie et al. 2010)[12] NSW 16,100 (Creese et al. 2008) VIC 47,000 (Green and Shore 2003) TAS 84,500 (Green and Shore 2003) SA 962,000 (Green and Shore 2003)			1997 (Kirkman 1997)
Global tidal marsh	40 million	Historical reclamation for agriculture and salt ponds, real estate development	1%-2% (≈1980-2000)	Centuries of conversion
Saltmarsh	**Total 1,376,500** WA 296,500 Bucher and Saenger 1991) NT 500,500 Bucher and Saenger 1991) QLD 532,200 Bucher and Saenger 1991) NSW 7,300 (Creese et al. 2008) VIC 27,900 (Sinclair and Boon 2012) SA 8,400 Bucher and Saenger 1991) TAS 3,700 Bucher and Saenger 1991)	Tidal restriction, fragmentation, access to waterways, offroad vehicles, mowing and watering, dumping of litter, stormwater, pollution, invasive species agricultural practices, reclamation for agricultural, industrial, port and residential development (Saintlan 2009)	1.2% (Saintilan 2000 between 1930 - 1994)	Eastern Australia 52% (Saintilan 2000) Victoria 5-15% Sinclair and Boon 2012) NSW 12 - 97% (West et al. 1992)
Global mangroves	13.7 million - 17 million	Aquaculture, forestry uses and agriculture	0.8%-2.1% (≈1980-2000)	35% (Valiela et al. 1998, Alongi 2002)
Mangroves	**Total 990,800** WA 164,000 (MIG 2008) NT 359,000 (MIG 2008) QLD 436,000 (MIG 2008)	Direct loss or alternation of trees from conversion and land-use changes associated with coastal development, indirect agricultural chemicals and sediment runoff	0.01 - 2% (Beeton et al.. 2006 and Valiela et al. 2001 between	8% loss in Victoria (Sinclair and Boon 2012) 23,100ha (Valiela et al. 2001)

[12] Noting 306 hectares in Great Barrier Reef World Heritage Area waters shallower than 15m and in locations that can potentially be influenced by adjacent land use practices (McKenzie et al.et al. 2010). An additional 3178 hectares of the sea floor within the GBRWHA has some seagrass present (Coles et al.et al. 2009). (McKenzie et al.et al.. 2012)

Habitat type	Extent (ha)	Conversion drivers	Annual loss rate	Total historic loss (%)
	NSW 12,600 (Creese et al. 2008) VIC 5,200 (Sinclair and Boon 2012) SA 14,000 (MIG 2008) TAS 0 (MIG 2008)	(Duke 2006)	1983-1990)	

CO_2 emissions

The rate of carbon captured by both terrestrial and coastal ecosystems is influenced by local environmental drivers. For example, some of the key environmental drivers controlling mangrove primary productivity are temperature (latitude), rainfall (influencing salinity) and nutrient availability (Feller et al. 2010, Morrissey et al. 2010). For savanna it is water and nutrient availability, vapour pressure deficit, solar radiation and fire (Kanniah et al. 2010). The carbon balance of these ecosystems can be modified with a change in environmental factors that influence productivity of plant communities. The draining, conversion or destruction of coastal ecosystems for other uses can disrupt the carbon sequestration by coastal ecosystems and may switch these ecosystems from being net sinks to net sources of carbon (McCleod et al. 2011) - effectively, stopping CO_2 sequestration and releasing the carbon stored back into the atmosphere (Lovelock et al. 2011). The process by which CO_2 is released back into the atmosphere when coastal wetland ecosystems are disturbed is an area needing further research. For mangroves and saltmarsh, it is thought that stored carbon is released directly to the atmosphere through a process of oxidation whereby soil carbon is metabolised by bacteria into CO_2 when exposed to oxygen. In seagrass systems the processes by which carbon is emitted is less well understood. It may be that carbon is mineralised or oxidised in the water column when living biomass dies or when carbon in previously anaerobic soils is eroded and exposed to aerobic water. Carbon released into the water column could then be released into the atmosphere (Sifleet et al. 2011).

The amount of CO_2 released is dependent on the type of disturbance, how deep into the soils the disturbance penetrates, and the type of ecosystem being disturbed. Disturbing the top layer only, e.g. the first metre, may mean that only the carbon to that depth is released to the atmosphere, provided the lower layers remain intact. Drainage to greater depths tends to increase carbon loss (Armentano and Menges 1986; Furukawa et al. 2005). Replacing native vegetation with lower biomass agricultural species or cultivation or burning can also result in relatively large losses of carbon (Nykanen et al. 1995; Hirano et al. 2007; Anda et al. 2009; Howe et al. 2009). Pendleton et al. (2012) estimated global carbon emissions from deforestation and land use changes to mangroves, saltmarsh and seagrass are in the order of 0.15–1.02 Pg (billion tonnes) of CO_2 released annually. These emissions are equivalent to 3–19% of those from deforestation globally, and result in economic damages of $US 6–42 billion annually (Pendleton et al. 2012).

Terrestrial carbon emissions (primarily from land clearing, with 95% occurring in QLD and NSW) are responsible for 14% of Australia's annual greenhouse gas emissions (Wentworth Group, 2009) and other land management activities such as biomass burning, decomposition of soil organic carbon from tillage practices and microbial activity related to fertilizer application directly contribute to emissions. Sixty-five percent of emissions from the agricultural sector are methane from livestock (Wentworth Group, 2009). What is unclear however is the contribution to emissions from degraded, damaged and conversion of coastal wetland ecosystems.

Changes to carbon following coastal wetland ecosystem drainage (Page and Dalal 2011)

Howe et al. (2009) reported that a disturbed wetland, drained for pasture production since the 1950s in the Hunter River estuary, NSW recorded total losses of 136 and 180 Mg CO_2 eq/ha from the top 0.2 m of mangrove and saltmarsh profiles, respectively. This represented a loss of nearly 40% of organic carbon over a period of around 50 years.

At a mangrove/saltmarsh site drained for around 20 years for sugarcane production near Cairns, Queensland, Hicks et al. (1999b) recorded large carbon losses of 2,605 t CO_2 eq/ha (around 45% of total carbon over 20 years) from the top 4 m of the profile of a mid-tidal mangrove area and losses of 807 t CO_2 eq/ha (26% of total carbon over 20 years) from a high-tidal mangrove area. Minimal losses were observed from a saltmarsh location. It should be noted that the soil at this site was an acid sulfate soil, and therefore the values for carbon loss may be overestimated as part of the carbon loss may have been due to the dissolution of carbonates.

Using the limited data available the conversion and degradation of Australian coastal wetland ecosystems each year coastal wetlands may be releasing up to 0.2 Tg CO_2 eq yr^{-1} into the atmosphere (Table 8). Mangroves contain the largest per-hectare carbon stocks and contribute the majority of the estimated total carbon emissions (0.1-93.1 Mg CO_2 eq ha^{-1} yr^{-1}). Our saltmarsh calculation only includes saltmarsh data for Victorian and some NSW estuaries, as loss information from other states was not available and therefore does not provide a national estimate. However, on a per hectare basis, Victoria and NSW carbon emissions are 7.1-50.3 Mg CO_2 eq ha^{-1} yr^{-1} which is relatively high given this represents only some of the losses in two states. These estimates are in line with the global range reported for mangroves (14-105 Mg CO_2 eq ha^{-1} yr^{-1} with mean of 44.5 Mg CO_2 eq ha^{-1} yr^{-1}) and saltmarsh (4-174 Mg CO_2 eq ha^{-1} yr^{-1} with mean of 32 Mg CO_2 eq ha^{-1} yr^{-1}) (Chmura et al. 2003, Silfleet et al., 2011, Donato et al., 2011; Lovelock et al. 2011; Fourqurean et al., 2012). Seagrasses, as expected are contributing the least in terms of national carbon emissions (0.2-0.6 Mg CO_2 eq ha^{-1} yr^{-1}). It is likely, given the lack of data at a national level of areal loss of saltmarsh and seagrass and the variability in numbers for mangroves, greater accuracy in estimates would be possible following improved mapping of changes to areal extent. It should be noted that work is under way to update the national extent of seagrass (J. Udy pers. com.).

Our estimates did not consider lost annual sequestration potential of coastal wetland ecosystems (Page and Dalal, 2011), but rather focused on the loss of carbon stocks in coastal ecosystem sediments that have accumulated over hundreds to thousands of years that have been lost, upon disturbance, within a period of decades (Crooks et al. 2011). The lost annual sequestration potential of coastal wetland ecosystems, which is considerable, would push these estimates to the high end of the spectrum, to around an additional 0.1Tg CO_2 yr^{-1}. Our estimates also account only for changes in ecosystem carbon and do not consider the transfer and deposition of carbon from one habitat to another. While the amount of carbon transferred to other habitats is likely to be small compared to the carbon gas emissions described above, caution should be taken when aggregating carbon budgets across multiple habitats.

Putting our estimates in perspective (and noting that saltmarsh only includes Victoria and NSW), the upper estimate for national annual emissions from converted or degraded coastal wetland ecosystems equates to an additional 4,397 cars on Australian roads or 0.04% of national emissions, 0.3% of national transport emissions, and 1% of annual national agricultural emissions (excluding livestock) and 37% of Tasmania's energy generation (DCCEE 2012). Compared to other ecosystem carbon fluxes, the loss of vegetated coastal wetland ecosystems may contribute an additional 0-0.5% above the most recent estimates of national emissions from deforestation (43.8Tg CO_2 per yr) (DCCEE 2012).

Given the small areal coverage of Australia's mangroves, saltmarsh and seagrass, these ecosystems play a disproportionally large role in land-use carbon gas emissions. Globally it is reported that disturbing the near surface carbon susceptible from conversion or degradation of a hectare of mangroves can contribute as much emissions as three to five hectares of tropical forest (Donato et al. 2011, Donato et al. 2012, Kauffman et al. 2012, Pan et al. 2011) and a hectare of near surface carbon from seagrass could contribute around the same amount of emissions as a hectare of tropical forests (Fourqurean et al. 2012, Pan et al. 2011). Our estimates, even with the limited data available are consistent with these global comparisons.

Economic impacts

Applying the same approach as Pendleton et al. (2012) we have estimated that the current global cost of Australian coastal ecosystem conversion to be between US$0-9 million incurred annually (Table 8). These estimates have used the mean value used by Pendleton et al. (2012) for the social cost of carbon (SCC) of $US 41 per tonne of CO_2 (USGS 2010) however, this range would be even wider if the full range of SCC values from $US7–81 had been applied. Regardless, there is a relatively high economic value in maintaining sediment carbon beneath coastal wetland ecosystems and keeping it out of the atmosphere. The high ongoing cost of coastal ecosystem loss also supports the conclusion of Irving et al. (2011), that management efforts focused on reducing coastal habitat loss may be more beneficial than the extensive restoration efforts being conducted in many regions which have smaller carbon benefits.

Coastal ecosystems are lost because market forces give landowners incentive to profitably convert habitat or regulations are ineffective at preventing catchment runoff and contamination. In Australia, while our coastal wetland ecosystems receive strong legislative protection, the reality is that cumulative impacts from development, population growth and urban expansion are slowly reducing the extent and health of these ecosystems, which impacts the other ecosystem services they provide, such as fisheries nurseries, coastal protection, biodiversity and cultural values (Barbier et al. 2011, Duke et al. 2007). With at least 50% of mangroves and saltmarsh found on private or leasehold land (Saintilan pers com, MIG 2008), there could be strong economic incentives provided for landowners or managers to protect the carbon stored in coastal ecosystems.

Table 8. National estimates of carbon released by landuse change in coastal wetland ecosystems and associated economic impact

Ecosystem	National reported loss (ha)	Current conversion rate (% y^{-1})	Near surface carbon (biomass + top metre of soil) susceptible - range (mean) Mg CO_2 ha^{-1}	Carbon emissions - range (mean) Mg CO_2 ha^{-1} y^{-1}	Carbon emissions - range Mg CO_2 y^{-1}	Total emissions Tg CO_2 - range (million Mg)	Economic cost $US million per year
Mangroves	168,607-236,935	0.01-1.99	583-4677 (2106)	0.1-93.1 (41.9)	11-214,997	0-22	0-8.8
Saltmarsh (VIC and NSW only)	2,162-8,852	1.17	607-4300 (1698)	7.1-50.3 (19.9)	156-2,415	0-0.5	0-0.1
Seagrass	20,029-45000	0.05	423-1261 (849)	0.2-0.6 (0.4)	159-2,837	0-0.0	0-0.1
Total				**7.4-144 (62.2)**	**327-220,249**	**0-22.5**	**0-9.0**

Notes – Saltmarsh loss range derived from Sinclair and Boon (2012) for Victoria and West (1992) and NSW DPI (pers comms) for some estuaries in NSW only as insufficient data for other states. National loss for seagrass derived from Waycott et al. (2009) and Kirkman (1997) for anthropogenic related loss only and Cofinas and Creighton (2001) and Beeton et al. (2006) for mangroves.

Conversion rates derived from Waycott et al. (2009) supplementary data for seagrass, Saintlan and Williams (2000) for saltmarsh and Valiela et al. (2001) and Beeton et al. (2006) for mangroves. Near surface carbon derived from data analysed in this report (Living biomass carbon + Soil carbon) and carbon emissions assumes 25% loss in first 50 years based on Page and Dalal (2011). No sensitivity analysis has been undertaken to account for uncertainties.

Economic estimates apply a multiplier of $US41 per tonne of CO_2 to upper and lower estimates of CO_2 emissions. Using the same approach as Pendleton et al. (2012) we multiplied the national emissions estimates for each ecosystem by a recent estimate of the global economic cost of new atmospheric carbon of $41 per ton of CO_2 (2007 U.S. dollars) (United States Government (USG) 2010)

Other GHG emissions

While all habitats can produce methane and other GHG in their soils through respiration, in addition to CO_2, drainage may induce several changes to CH_4, and N_2O fluxes from coastal wetland ecosystems. Marine based coastal wetland ecosystem soils however, have high concentrations of sulphate which hinders methane (CH_4) production, so these ecosystems are considered to be negligible sources of CH_4, if not CH_4 sinks (Bartlett and Harris 1993; Magenheimer et al.. 1996; Giani et al.. 1996). Research also suggests emissions of N_2O are also low (Smith et al.. 1983; DeLaune et al.. 1990) except where nutrient enrichment is very high (Allen et al. 2007). Table 9 shows a comparison of the level of other GHG produced by Australian ecosystems and land-uses.

Table 9. Comparison of CH_4 and N_2O emissions

Habitat/landuse type	**CH_4 (Mg $ha^{-1}yr^{-1}$)**	**N_2O (Mg $ha^{-1}yr^{-1}$)**
Mangroves[a]	<0.008-0.121 for temperate systems 0-1.522 for sub tropical degraded systems 0.113 for tropical systems 0.027 global average	0.000 - 0.006 0.007 global average
Saltmarsh[b]	0.002	<0.003
Seagrass	No data	No data
Forest[c]	-0.0876 to – 4.38 (sink)	0.438 to – 4.38 (sink)
Savanna[d]	- 0.002 to -0.0016 (sink)	very low
Pasture[e]	+0.867 to –2.61 (sink)	0.94-1.17, + 0.0876 to +8.76
Cropland[f]	-	3.75
Melaleuca forest[g]	-0.008-0.000 (Sink)	<0.003
Freshwater wetlands[h]	0.70	0.0009 *
Livestock[i]	0.14	0.025

Source: [a,b] Kreuzwieser et al. (2003), Livesley and Andrusiak (2012), Allen et al. (2007), Page and Dalal (2011), [c d,] Livesley et al., 2011, [e] Allen et al.,(2009) , [,f] CSIRO, 2009, [g,h] Boon and Sorrell (1995), Page and Dalal (2011), [i] calculated from Table 10.

Terrestrial forest and woodland systems are often a considerable CH_4 sink, as long as soil moisture conditions prevent the development of anaerobic conditions (Dalal et al., 2008; Livesley et al., 2009). Allen et al. (2009) examined effects of land use change of grazed pastures on the in situ fluxes of nitrous oxide (N_2O) and methane (CH_4) from soil across 3 forest types in Australian temperate, Mediterranean, and subtropical ecoregions. During the 12-month study, rates of N_2O flux ranged between in + 0.0876 and +8.76 Mg ha^{-1} yr^{-1} in pasture soils and from -0.438 to +4.38 Mg ha^{-1} yr^{-1} in forest soils. Rates of CH_4 flux varied from -0.0876 to - 4.38 Mg ha^{-1} yr^{-1} in forest soil and from +0.867 to –2.61 Mg ha^{-1} yr^{-1} in pasture soils.

Scheer et al. (2011) assessed the effect of biochar application (10 Mg ha^{-1}) on the emissions of GHG from subtropical ferrosol pasture in NSW and found, contrary to expectations, that biochar did not lead to a reduction in GHG emissions from the soil.

A significant proportion of Australia's land based emissions occur as non-carbon dioxide gases, in particular CH_4 from livestock production (58.1 Mt CO_2 eq), N_2O from agricultural soils (14.2 Mt CO_2 eq) and CH_4 and N_2O from savanna burning (12.5 Mt CO_2 eq). This represented 9.7%, 2.4% and 2.1% of Australia's net emissions in 2009 (ABARES 2011).

Methane is emitted directly from livestock and CH_4 and N_2O are also released from manure and urine[13] (Table 10).

Table 10. Estimates of livestock GHG emissions for Australia and Queensland Mg CO_2 eq per year per animal (Source: Department of Climate Change, 2008)

Source	CH_4	N_2O	Estimated number of animals, million's *
Cattle	46.08	1.04	26.5
Sheep	13.55	-	68
Other livestock	1.54	5.45	2.3
Total	61.2	10.58	96.8
Grazing area**	430,100,800 ha		

* Source Agricultural Commodities, Australia, 2009-2010, http://www.abs.gov.au/ausstats/

** http://www.anra.gov.au/topics/land/landuse/index.html#lands

[13] Reductions in emissions from livestock are being investigated through waste management (aeration and composting may reduce the amount of methane that is produced from manure stockpiles), sheep and cattle genetics and feed alternatives (DAFF, 2012).

Nitrogen additions to cropland soils are the largest source of anthropogenic nitrous oxide (N_2O) emissions (Table 9). Average rates of fertilizer application for a range of crops in Queensland are estimated to be between 120 kg/ha (for sugarcane) and 60 kg/ha for cereals (CSIRO, 2009). From 0.7 to 16% of applied inorganic fertiliser N is lost as N_2O from intensive cropping systems. Emissions from flooded rice fields tend to be lower than from other crops due to the anaerobic conditions that lower N_2O production ratios in denitrification (Stehfest and Bouwman, 2006). Most tropical and subtropical cropping soils provide a small sink for CH_4. In areas where CH_4 is emitted it is small proportion of CH_4 emitted from livestock.

Savanna regions are the most fire-prone biome, with up to half or more of many landscapes burnt each year (CSIRO, 2009). Savanna fires release both CH_4 and N_2O and calculations show that improved management could abate approximately 5 Mt/yr CO_2 eq. (CSIRO, 2009). Accountable emissions of CH_4 and N_2O from savanna burning contribute about 6–8% of global carbon emissions from biomass burning (Meyer et al. 2008) but fluctuate considerably between years. Most of this variation is due to seasonal conditions, with extensive fires following periods of high rainfall in arid and semiarid Australia (Cook et al., 2010).

The West Arnhem Land Fire Abatement (WALFA) project aimed to reduce non-CO_2 emissions from savanna burning in a 2,800,000 ha area, by 0.1 Mt CO_2-eq y^{-1} in 5 years to 2010. The project involved implementing a programme of early dry season burning, thereby reducing the extent of unmanaged late season, intense fires. The change in fire regime was very successful and exceeded expectations with an estimated 0.7 Mt CO_2 eq y^{-1} reduction in GHG emissions (Heckbert et al., 2011).

There are only a few studies that measure CH_4 and N_20 emissions from undisturbed and disturbed Australian coastal wetland ecosystems over an extended period. Given the lack of data and therefore baseline emissions rates for Australia, accurately assessing the effect of wetland drainage on GHG emissions in Australia is difficult Two studies, as reported by Page and Dalal (2011) consist of a body of work that measured CH_4 flux from a Victorian freshwater floodplain (*Melaleuca*) wetland (Boon and Sorrell 1995), and work conducted in central and southern Queensland to quantify N_2O and CH_4 flux from mangrove ecosystems (Kreuzwieser et al. 2003). More recently, Livesley et al. (2012) examined N_2O and CH_4 flux across mangroves, saltmarsh and *Melaleuca* forest near Mornington Peninsula, Victoria and Allen et al. (2007) examined N_2O and CH_4 flux in subtropical mangrove sediments along the Brisbane River, at a site located adjacent to a treated sewage outlet.

Page and Dalal (2011) used data collated from studies in other countries to provide a preliminary estimate of likely Australian emission rates in lieu of Australian data and noted that CH_4 emissions reported by Kreuzwieser et al. (2003) from Australian mangroves are similar to their worldwide average of 0.027 Mg ha^{-1} for mangrove wetlands. The values reported by the additional studies of mangroves covering temperate (<0.008-0.121 Mg ha^{-1}), sub tropical (0-1.522 Mg ha^{-1}) and tropical (0.113 Mg ha^{-1}) subzones were also consistent with the global estimates (Table 11). The N_2O flux data collected from Australian mangrove ecosystems (0.000 - 0.006 Mg ha^{-1}) was also similar to the only other measurement of N_2O in undisturbed mangrove ecosystems identified of 0.0007 tonnes per hectare (Barnes et al. 2006). Livesley et al. (2012) reported that saltmarsh was a weak (negligible) source of CH_4, and N_20 (0.002 Mg ha^{-1} and <0.003 Mg ha^{-1} respectively) and *Melaleuca* woodland soil was a constant moderate CH_4 sink (-0.008-0.000 Mg ha^{-1}) and weak source of N_20 (<0.003 Mg ha^{-1}) (Table 10). There is no data available for seagrass.

It is important to note that across the studies undertaken fluxes of N_2O and CH_4 differed significantly between sampling seasons, as well as between different hydrological zones within mangrove forests. In addition, N_2O flux differed significantly over diurnal cycles. Higher bulk density and total carbon content in sediment were significant associated with decreasing N_2O emissions. Livesley et al. (2012) reported that on the basis of their global warming potentials, CH_4 emissions dominated in summer and autumn seasons, whereas N_2O emissions dominated in winter when overall CO_2-eq emissions were low. Allen et al. (2007) however noted that Purvaja and Ramesh (2001) observed several human-induced factors that enhance CH_4 emissions from mangroves to the atmosphere, and that there is evidence that additional nitrogen inputs in mangroves increased N_2O emissions (Kreuzwieser et al., 2003). Increasingly, riverine mangrove sediments are considered to contribute to N_2O and CH_4 emissions (Sotomayor et al., 1994; Corredor et al., 1999; Purvaja and Ramesh, 2001; Kreuzwieser et al., 2003). Thus as human expansion continues along riverine and coastal shorelines, mangroves may be subject to anthropogenic inputs including sewage, aquaculture and agriculture, which potentially increase nutrients in mangrove ecosystems (Alongi, 2002) and thereby potentially increasing the CH_4 and N_2O emissions.

3.2 Capturing the full potential of coastal carbon

Conservation and restoration opportunities

The comparatively high carbon sequestration rates and emissions estimated for coastal wetland ecosystems in Section 2.5 and Section 3.1 respectively indicate that coastal wetland ecosystem restoration can not only have positive benefits for regulating atmospheric carbon concentrations (Irving et al. 2011), but also will deliver other ecosystem services such as fisheries productivity and coastal protection (Barbier et al. 2011). While draining wetland soils can result in substantial losses of soil carbon stocks through oxidation and decomposition, reintroducing tidal flows can reverse this decline. The rate of carbon sequestration in rehabilitated wetlands can exceed that of natural wetlands, mainly through an increase in the rate of soil vertical accretion. The ability of estuarine wetlands to continue sequestering carbon depends on their ability to adapt to changes in environmental conditions (Howe et al. 2009). The inclusion of coastal wetlands in carbon accounting, trading and incentive schemes will be contingent upon improved estimates of the carbon stocks and rates of flux in coastal wetlands, and an improved understanding of the drivers of variability in carbon sequestration potential at regional scales.

Globally, the conversion of wetland ecosystems to croplands and pastures and the burning of peat for fuel can reduce carbon stocks in the soils by up to 50%, mostly within the first decade following land use change (Armentano and Menges 1986). Since European settlement Australia is estimated to have converted 50% of its wetlands (Commonwealth Government of Australia, 1997) to other landuses. Page and Dalal (2011) estimated for Australia, that drained coastal wetland soils will lose on average 25% of the organic carbon from the top 1m in the first 50 years following drainage. These values are within the range (20–60%) of those reported for conversion of relatively undisturbed lands to arable agriculture over the first 50-years (Dalal and Chan 2001), meaning that Australia's converted or degraded coastal wetlands are also significant contributors of greenhouse emissions.

Coastal wetland ecosystems across Australia, and particularly along the east coast, have been affected by extensive modifications to tidal flows or have been reclaimed for alternate land-uses. In NSW alone Williams and Watford (1997) identified around 4,300 barriers to tidal flow, of which 1,398, if removed or regulated, would provide opportunities for wetland restoration (Williams and Watford 1996, Williams and Watford 1997). While a number of restoration projects in the US have indicated some recovery of the carbon stocks of tidal wetlands following reintroduction of tidal flows (Craft, 2001) there is only one example of a project in Australia to date (Howe et al. 2009).

Howe et al. (2009) quantified the soil carbon storage and sequestration rates of undisturbed natural wetlands and disturbed wetlands subject to restricted tidal flow and subsequent rehabilitation in the Hunter River estuary in NSW. The wetlands comprised saltmarsh and mangrove habitats, characterised by temperate climate, small tidal range, low sediment supply and small stream discharge. Howe et al. (2009) found that the carbon sequestration rate of undisturbed mangroves and saltmarsh was lower than for disturbed mangroves and saltmarsh, but the carbon store was higher in undisturbed mangroves and saltmarsh. The increased carbon sequestration rate of the disturbed wetlands was driven by substantially higher rates of vertical accretion. Osland et al. (2012) also showed that rates of carbon accumulation in the top 10 cm of soil at a created mangrove was in line with the rate of other natural, restored, and created wetlands and the sequestration rate was very close to the global mean for natural mangroves and saltmarsh. They indicated that the time to equivalence for the soil carbon stock in the upper 10-cm of created mangrove wetlands can be relatively rapid (<20 years) and faster than most other wetland types. These findings support the potential for substantial gains in carbon sequestration associated with reinstatement of tidal flows to degraded estuarine wetlands in a relative short time.

Incentives for coastal wetland ecosystem restoration and/or establishment could increase in the context of an emissions trading scheme for Australia. With the current Clean Energy Futures package where carbon pollution is currently taxed at $23 per tonne, the costs associated with the preservation and management of land for the multiple benefits associated with wetland restoration become affordable and potentially attractive under the Carbon Farming Initiative that regulates the development of carbon offsets associated with altered land management practices. For example, the Hexham Swamp Rehabilitation program aims to restore 1,946 ha of estuarine wetland, primarily saltmarsh, an endangered ecological community in NSW (Figure 7). Saintilan and Rogers in prep estimate that the value of the wetland for carbon sequestration once restoration is complete would be $US150 000 per annum. This figure is conservative given that the pre-existing wetland, a freshwater *Phragmites* reed swamp, is likely to have been a net GHG emitter over management timescales of up to 100 years (Brix et al. 2001).

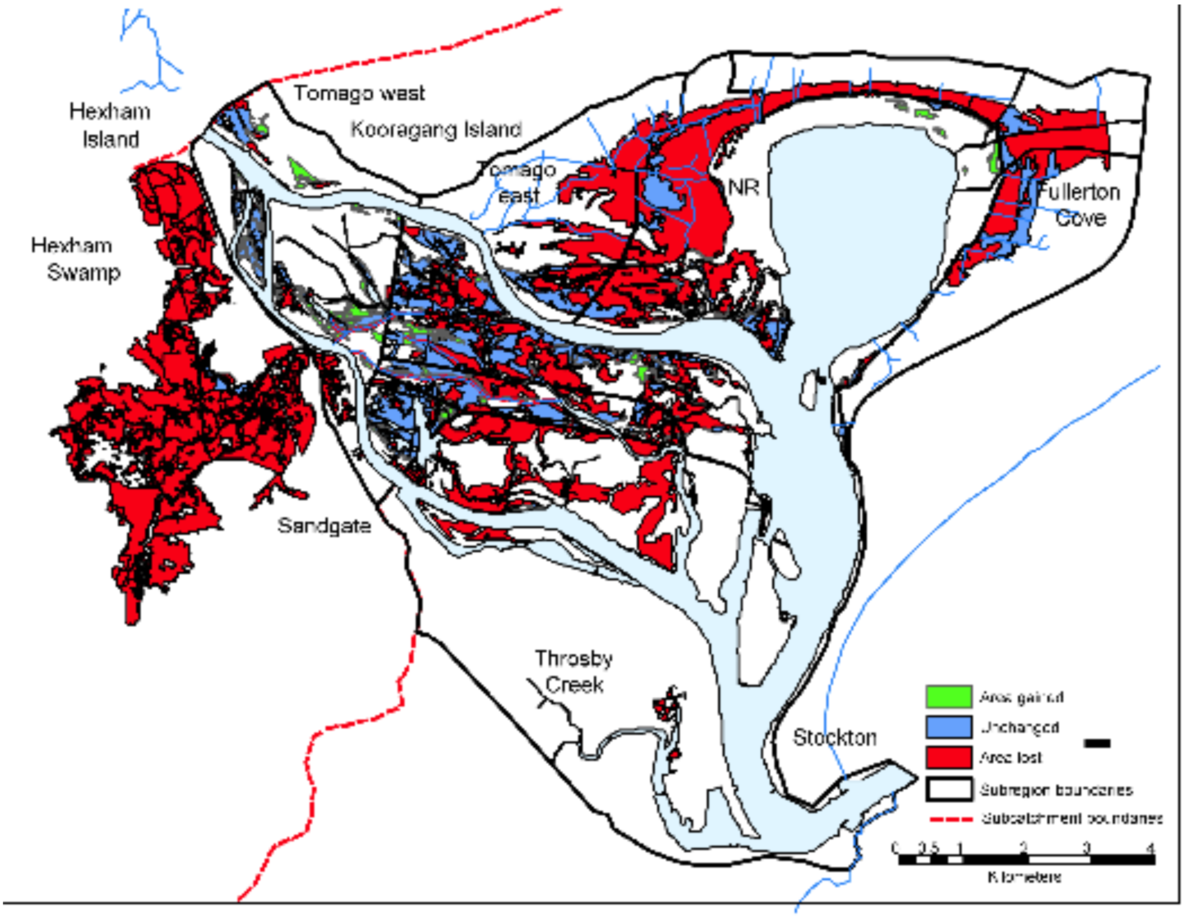

Figure 7. Coastal wetland ecosystem loss in Hexham Swamp (Source NSW DPI)

Removing impediments to tidal inundation for coastal wetland ecosystems will not only provide benefits in terms of climate change mitigation, but also for adaptation. Rogers et al. (2012) demonstrated that at a landscape scale, planning for sea-level rise should be directed towards facilitating wetland adaptation by promoting tidal exchange to mangrove and saltmarsh and providing land for wetland migration, as under moderate rates of sea-level rise (3.65 mm y^{-1}) by 2050, natural coastal wetlands have the ability to self-regulate to perturbations. They suggested that removal of impediments to tidal flow may promote the maintenance of these ecosystems under rising sea levels. In fact, their study indicates that these impediments may be having a greater impact on wetland extent and ecosystem function than that attributed to sea level increases.

It is important to note however that careful site selection for restoration is important as some mangroves and saltmarsh can be net exporters or emitters of carbon as rates of carbon accumulation and loss and gaseous flux of CH_4 and N_2O vary both within and among estuaries (Alongi et al. 2005, Poffenbarger et al. 2011). The success of restoration projects is dependent on the long-term stability of a site and the soil carbon stocks being accumulated (Alongi 2012). Alongi (2012) noted that upstream sites within estuaries represent a higher risk, with more dynamic fluvial drivers of geomorphic change and greater likelihood of methanogenesis. Restoring mangrove and saltmarsh wetland within reclaimed lowland floodplains protected by floodgates presents a high likelihood of success in controlled environments amenable to experimental manipulation. Careful monitoring of carbon benefits derived from such projects will increase confidence in the market over time (Rogers et al. 2012).

The latest science (Alongi 2012) shows that there remains large uncertainties concerning the extent to which these mangroves are a natural sink for carbon. This uncertainty is coupled to the natural dynamics of mangrove ecosystems. Any carbon or payments for ecosystems service schemes must account for this level of uncertainty. To maximise carbon payments for mangroves, schemes must therefore be restricted to choosing sites conducive to net accumulation of carbon, that is, primarily at the sea-forest boundary, unless payments are being made simply to preserve existing mangroves or to preserve biodiversity. In order to maximise the efficiency of schemes, close collaboration between managers and scientists is required. A combination of approaches, such as ecological modelling, field testing of ecosystem services and filling of existing information gaps to improve the accuracy of modelling estimates is necessary for the successful sustainable management of mangrove ecosystems (Alongi 2012). It should be noted that it is likely that natural variability in carbon sequestration will also be a challenge for seagrass and saltmarsh carbon projects.

The co-benefits of coastal carbon

Ecosystem services provided by coastal and marine habitats are of crucial importance for global food security and poverty eradication, as well as many of the sectors currently driving the economies of coastal nations.

In Australia, saltmarsh, seagrass, mangroves and other estuarine and coastal environments provide ecosystem services including food, habitat shoreline protection, water filtration, recreation and cultural benefits. Although they clearly provide economic, ecological and cultural benefits to communities, most coastal ecosystem services have not been systematically evaluated. Coastal wetland environments in Australia are under increasing pressure from development and their function is often underestimated and the benefits of their services underpriced in policy settings and therefore likely to be ignored in land use decision making (Barbier 2007). However around the globe the significance of coastal wetland ecosystems to environmental well-being and community prosperity is increasingly recognised.

Functioning coastal wetland ecosystems are ranked among the most economically valuable of all ecosystems and vital for the food security of coastal communities, providing nurseries and fishing grounds for inshore fisheries and a critical natural defence against storms and coastal erosion. Coastal wetland ecosystem services[14] globally have been valued at US$25,783 billion (Martinez et al., 2007) or just over $2,800 ha-1 yr-1 (Brander et al. 2006) and yet we are losing our global coastal wetland ecosystems at rates exceeding terrestrial systems in some locations. The Millennium Ecosystem Assessment estimates the market value of seafood from mangroves at $7,500 to $167,500 per km per year (MEA 2005) and Bann (1997) estimated annual commercial fish harvests from mangroves at $6,200 per km in the United States to $60,000 per km in Indonesia (Bann, 1997).

There is some data available providing an economic valuation of Australia's coastal wetland ecosystems, however these tend to be location specific, eg Moreton Bay or Great Barrier Reef, and mostly focus on fisheries and recreational values (Manson et al. 2005, Meynecke et al. 2008, Prayaga et al. 2010). Little information is available in relation to other ecosystem services provided, such as cultural services (H.Yorkston pers com.) To date, reliance has been placed on the international estimates such as those provided by Costanza et al. (1989), Barbier et al. (2011) or in the Millennium Ecosystem Assessment (MEA 2005) to generate estimates for Australia.

Food security and fisheries productivity

Coastal wetland ecosystems such as seagrass, saltmarsh, and mangroves are particularly valued for their extremely high productivity, which supports a great abundance and diversity of fish as well as shrimp, oysters, crabs, and other invertebrates. The vegetation provides both carbon and nutrient resources (e.g. biomass is broken down by fungi and bacteria or are eaten by small crustaceans and incorporated into the food web) and increases the physical structure of the ecosystem providing refuge from predation for many species,. The abundance of detrital food webs attracts breeding fish and shellfish. Coastal wetland ecosystems are widely considered as "nurseries" (Morton 1990). Mangroves, seagrass and coral reefs are linked by water masses that move with the tides and current. Around 70% of Australia's commercially and recreationally significant fish and prawns found offshore, inhabit mangrove and seagrass areas during part of their life cycle (Newell and Barber 1975, Pollard 1976, Staples 1980, Morton 1990). Mangroves are important nursery grounds for commercial species such as king prawns, barramundi, snapper, bream and mackerel. Seagrass also shelters the larval and juvenile stages of many fish and other sea creatures including leatherjackets, mullet, whiting, tailor, buffalo bream, flathead, seacucmbers and seahorses (Bloomfield and Gillanders 2005, Connolly 1994). Some smaller, non-commercial species also spend their vulnerable juvenile stages in the mangroves and seagrass and become the food source of larger fish when they migrate to the open ocean. Other species, such as mud crabs, spend most of their lives in the mangroves and move to the open sea to spawn.

Food provisioning in the form of fisheries catch is one of the most important services derived from coastal wetland ecosystems. Loss of mangroves, saltmarsh and seagrass can result in threatened food security for coastal communities and loss of habitat, which in turn may damage to other industries such as tourism and recreational fishing. Many people depend on wetlands for food and employment. In Australia just over 15,000 people are employed in commercial fisheries and aquaculture, 90,000 in recreational fisheries (ABARES 2011) and more than 37,000 indigenous Australians use coastal wetland ecosystems.[15] Ridge Partners (2010) estimates that about 3.4 million Australians engage in recreational fishing each year.

[14] Ecosystem services are the benefits provided to people from nature that play a vital role in livelihoods and economies at all scales – these benefits can only be realised if the capacity of natural processes is retained. (Munang et al.. 2010)

[15] Only covers indigenous Australians living in coastal communities across the north of Australia from Broome in Western Australia to Cairns in Queensland (excluding those living in the Torres Strait) and represents 91.7 per cent of the Indigenous population in northern Australia.

Coastal protection

Mangroves, seagrasses and saltmarsh play an important role in preventing coastal erosion (Barbier et al. 2007; Spalding et al. in press, Shepard et al. 2011). Mangrove clearing, in particular, results in low-lying coastal areas being more susceptible to damage from cyclones and storm surges (Barbier 2007, Danielsen et al. 2005). Coastal vegetation helps stabilize soils. Compared with modified coastlines with artificial shoreline protection structures, natural systems are more adaptive to both routine and irregular changes in the dynamic coastal system. For example, the tangle of roots, pneumatophores and trunks of mangroves acts to reduce current speeds and traps sediment and nutrients. Additionally the structure of the vegetation helps reduce siltation in adjacent marine habitats. Similarly, river-borne nutrients (Adame and Lovelock, 2011), including agricultural chemicals (Ewel et al. 1998) are trapped and recycled within mangroves.

4 Conclusion and recommendations

Coastal wetland ecosystems such as mangroves, saltmarsh and seagrass provide fundamental ecosystem services such as fisheries production, coastal protection and nutrient, sediment and pollution filtering to improve water quality. To date, many coastal wetland ecosystem rehabilitation projects across Australia have been driven by species conservation, land management or water quality improvement. This report provides a summary of the growing body of evidence demonstrating the significant contribution of these wetlands to carbon capture and burial within Australia, In south eastern Australia the carbon value of mangrove and saltmarsh may outweigh the value of for commercial fisheries productivity (Saintilan and Rogers in prep). The conservation and restoration of mangrove, saltmarsh and seagrass therefore represent a win-win proposition for Australia.

Blue Carbon is a new concept in climate change mitigation and there are considerable knowledge gaps regarding the carbon services of coastal wetland ecosystems. For Australia, the scientific understanding of coastal wetland ecosystems, their extent and carbon reservoirs and burial potential varies greatly, and there are limitations in available data and methodological guidance. To move forward we need a comprehensive approach that leads to improve management and restoration for coastal wetland ecosystems in Australia (Figure 8).

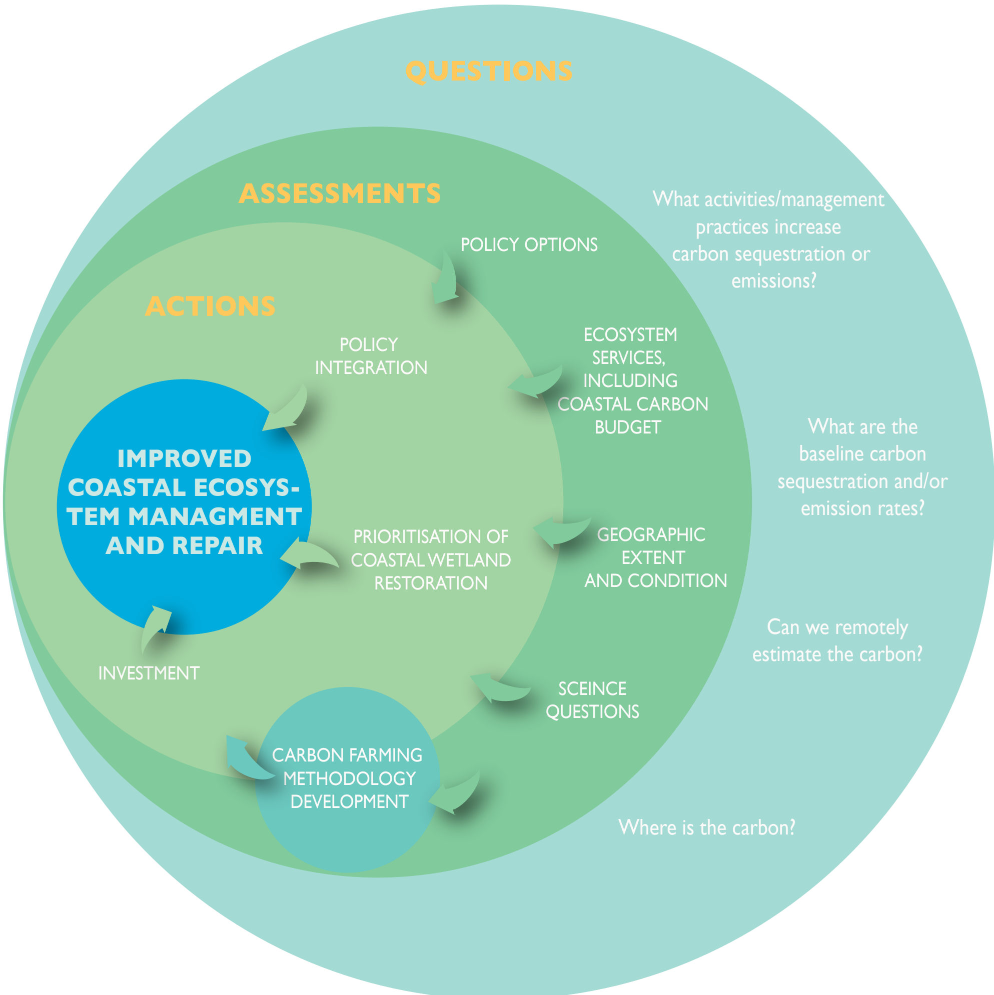

Figure 8. Comprehensive approach for improved coastal wetland ecosystem management and repair for Australia.

We make the following recommendations:

Policy and management

Australia should set in place a timetable and processes to integrate *Blue Carbon* into national climate policy. There would be a range of follow on implications including:

- ensuring coastal ecosystems are a priority within implementation initiatives such as the Carbon Farming Initiative and Biodiversity Fund, commensurate with their high carbon values;
- identifying site and landscape scale restoration priorities to deliver improvements to coastal wetland ecosystems providing (or with the potential for) high carbon and other values such as fisheries habitat repair;
- updating national datasets for mangroves and seagrass ecosystems, and in the case of saltmarsh, developing national datasets, mapping the areal extent and assessing condition to provide a comprehensive understanding of status and land-use changes, comparable across regions and states and contributing to National Carbon Accounting;
- recognising the multiple values, including CO_2 mitigation values of coastal ecosystems to develop and implement a National Action Plan for the Conservation and Restoration of Australia's coastal ecosystems, that seeks to standardise conservation and management regulations and measures across regions and States and supports restoration and rehabilitation of priority coastal wetlands ecosystems; and
- exploring the feasibility of community monitoring approaches, management intervention and providing incentives for maintaining carbon rich ecosystems. Participation of key stakeholder groups such as commercial and recreational fishing groups, coastal farmers and indigenous communities in projects to generate new revenue streams related to coastal wetland repair would be important in this process.

Internationally, Australia should lead policy development with partners (e.g. Indonesia) as part of the United Nations Framework Convention on Climate Change (UNFCCC) and its related processes and mechanisms, to incorporate blue carbon into the UNFCCC.

Scientific understanding

Recognising that coastal ecosystems, when compared to terrestrial ecosystems, are a very significant part of Australia's carbon stores and carbon management opportunities, we need to build on existing scientific data, analysis and available technologies to develop a coherent Australia-wide data gathering and assessment initiative that should:

- assemble sufficient data to support the development of policy and management activities;
- address gaps in knowledge in relation to carbon storage and sequestration for Australian coastal wetland ecosystems, utilising consistent internationally accepted measurement and assessment methodologies that are comparable across coastal and terrestrial ecosystems;
- undertake detailed baseline carbon inventories of coastal wetland ecosystems and incorporate coastal carbon into the Australian Terrestrial Carbon Budget (being undertaken by CSIRO) to quantify national coastal carbon storage, sequestration and losses;
- undertake a baseline assessment of related Australian coastal wetland ecosystem services (the need for a bundled/layered/stacked Blue Carbon plus other ecosystem services approach);
- conduct targeted research and monitoring to more accurately quantify the greenhouse gas emissions resulting from degradation, conversion and destruction of coastal ecosystems;

- establish a network of targeted field projects that assesses the capacity for carbon storage for coastal wetland ecosystems and the emissions resulting from degradation, conversion and destruction of coastal ecosystems; and
- conduct research quantifying the consequences of different coastal restoration and management approaches on carbon storage and emissions in coastal wetland ecosystems.

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Appendix A Lookup tables