The Distribution and Abundance of ‘Blue Carbon’ within Port Phillip and Westernport

A report for the Port Phillip & Westernport Catchment Management Authority

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Cover photo: Saltmarsh and Mangroves next to the West Gate Bridge, Yarraville
Executive Summary

Vegetated coastal habitats—seagrasses, saltmarshes and mangroves—have recently been identified as among the most effective carbon sinks on the planet. They can bury carbon at a rate 35-57 times faster than tropical rainforests and can store carbon for thousands of years. Recent global data estimate that vegetated coastal habitats contribute 50% of carbon burial in the oceans – termed “blue carbon”. These features make vegetated coastal habitats ideal candidates for carbon offset programs and nature-based climate mitigation initiatives.

In 2014, the Port Phillip and Westernport CMA identified a lack of information on the distribution and abundance of blue carbon within the catchment. Such information is critical for guiding the spatial prioritisation of conservation efforts. To address this knowledge gap, the Port Phillip and Westernport CMA commissioned researchers from Deakin University to conduct Port Phillip and Westernport’s first blue carbon stock assessment, focusing on sedimentary organic carbon. The major findings of this program are as follows:

- Port Phillip and Westernport have a total blue carbon stock of 1,025,203 Mg and a total carbon value of $15,378,048 over the top 30 cm of sediment at $15 Mg$^{-1}$.
- It should be noted that because current sampling was confined to the top 30 cm of sediment, the carbon estimates given here are highly conservative. In fact, since organic carbon is stored at depths up to several metres, the true value of these habitats will be even greater.
- The average soil carbon content is 5.77%, and 80.20 Mg carbon ha$^{-1}$ (over top 30 cm).
- Seagrass beds comprise 54% of the carbon stocks in Port Phillip and Westernport, which, despite their low percent carbon values, comes from the extensive seagrass beds in Westernport bay and to a lesser extent, Port Phillip.
- The seagrass maps used to calculate total carbon stock and value do not include the seagrass distribution outside of the bays, and are, therefore, underestimates of seagrass extent in the Port Phillip and Westernport catchment. This mapping was also conducted back in 2000 — 2001, with only smaller sections being completed more recently (French et al. 2014).
- Saltmarsh comprise 31% of the carbon stocks in Port Phillip and Westernport. Saltmarsh habitats were sampled the most of the vegetated coastal habitats and were found to have high carbon stocks ha$^{-1}$, with high (>10% carbon) values recorded at Hastings, Rhyll Inlet, Warneet, West Gate Bridge and Yaringa.
- Mangroves, while representing the smallest proportion of the carbon stock in Port Phillip & Westernport at 15%, had the second highest average carbon values, with an average mangrove carbon stock of 82.9 Mg carbon ha$^{-1}$.
- Across the sample sites in Port Phillip and Westernport, a trend that emerged was that areas higher in the estuaries (or closer to fluvial inputs) such as West Gate Bridge and Rhyll Inlet, were associated with higher carbon stocks (Fig 5c).
- The saltmarsh and mangroves at West Gate Bridge (Stony Creek) had the highest carbon stock values and therefore monetary worth at $2,535 ha$^{-1}$ for saltmarsh and $2,281 ha^{-1}$ for mangroves (Table 2). This site is located at the junction of two
estuaries (Stony Creek and the Yarra River) and is the likely reason for this high value. Interestingly, these mangroves were planted in 1984 and represent a significant addition to the carbon pool of Port Phillip and Westernport.

This report summarizes the valuable soil carbon stocks in blue carbon ecosystems across Port Phillip and Westernport. Saltmarsh makes up the second largest portion of blue carbon in Port Phillip and Westernport, this is due to the high sediment carbon content (almost 3 times that of seagrass), while only occupying 22% of the area of seagrass. Additionally, areas higher up in the estuary (or further away from the coast) tended to show the highest carbon stock, and therefore, the highest monetary value. Based on their previously mapped distribution, seagrasses account for just over half of Port Phillip and Westernport’s blue carbon. Updated distribution maps of seagrass are necessary to accurately estimate the seagrass carbon stock and identify highest-priority areas for seagrass conservation. We identified a number of areas of vegetated coastal habitat that should be prioritised for conservation because of their notably high carbon stocks. Further, the trends identified in blue carbon soil stocks provide valuable insight for identifying appropriate locations for revegetation (and potentially, carbon offset) programs. With carbon sequestration initiatives firmly on the national agenda, both protecting and improving these habitats will only become more important.

With a growing Australian push to ‘get blue carbon to market’, we recommend further research into opportunities for blue carbon offset projects within Port Phillip and Westernport, through strategic preservation or restoration of former blue carbon habitats (e.g. bund/dyke wall removals), and through management of catchment-level processes to enhance blue carbon sequestration within existing habitats (e.g. restore natural hydrology). Though the goal of such activities would be carbon enhancement, there would be broad environmental, social, and economic benefits (e.g. biodiversity and fisheries enhancement, shoreline stabilisation, climate change buffering, improved shoreline amenity).

In sum, we recommend the following actions be taken to maximize blue carbon stocks in Port Phillip and Westernport:

1)  Prioritize blue carbon hotspots for conservation
2)  Focus revegetation projects on saltmarsh and mangrove ecosystems and/or estuarine environments closer to fluvial inputs
3)  Restore natural hydrology (e.g. via bund wall removals) to enhance blue carbon sequestration
4)  Produce updated seagrass distribution maps
5)  Research into the distribution and carbon storage potential of freshwater wetland ecosystems in Port Phillip and Westernport
Introduction

Saltmarsh, mangroves, and seagrass meadows—collectively known as vegetated coastal habitats or “Blue Carbon” habitats—together sequester nearly equivalent quantities of organic carbon ($C_{org}$) as their terrestrial counterparts, in spite of their comparatively limited biomass (0.05% of terrestrial plant biomass). Blue Carbon habitats are reported to store organic carbon at almost 40 times the rate of terrestrial systems (Fourqurean et al. 2012a). Estimates from some parts of the world indicate that carbon is sequestered at a rate of up to 151.0 g C m$^{-2}$ yr$^{-1}$ in saltmarsh, 139.0 g C m$^{-2}$ yr$^{-1}$ in mangroves, and 83.0 g C m$^{-2}$ yr$^{-1}$ in seagrass (Smith 1981; Duarte et al. 2005; McLeod et al. 2011). The relatively anaerobic soils of vegetated coastal habitats prevent organic carbon remineralisation and tend to promote long-term sequestration (Mateo et al. 1997; Pedersen et al. 2011). As such, carbon may be stored for centuries to millennia, as opposed to the decadal scales typical for terrestrial systems, and never become saturated due to the vertical accretion of sediment in these habitats. Vegetated coastal habitats both produce and store their own carbon (autochthonous carbon), but also trap carbon produced from other locations (allochthonous carbon). Their ability to trap particles and suspended sediment means that vegetated coastal habitats may appropriate large quantities of the allochthonous organic carbon that originates from adjacent habitats, both terrestrial and marine (Gacia and Duarte 2001; Agawin and Duarte 2002; Hendriks et al. 2008; Kennedy et al. 2010).

However, degradation and loss of vegetated coastal habitats via mismanagement could shift them from carbon sinks to carbon sources, releasing atmospheric CO$_2$ equivalent to annual damages of US$6 to 42 billion globally (Pendleton et al. 2012). While natural disturbance events can lead to the loss of stored organic carbon (Macreadie et al. 2013), anthropogenic impacts including clearing of land, land fill, tidal restriction, stock grazing, and degradation of water quality have consistently driven more severe losses. The current global estimates of saltmarsh and mangrove habitat loss are around 25-35% (Valiela et al. 2001, Alongi 2002, IPCC 2007, Bridgham et al. 2006), though lower rates are estimated in Australasia (18% loss for mangroves). Total seagrass loss is similar, at an estimated 18-50% over the last 20 years (Green and Short 2003, Waycott et al. 2009). The rate at which such declines are occurring (based on multiple decades of data) was >1% y$^{-1}$ for seagrasses (Duarte 2002, Short and Green 2003, Duarte et al. 2005b), but have now accelerated to 7% y$^{-1}$ since the 1990s (Waycott et al. 2009).

In addition to their important role in carbon sequestration, vegetated coastal habitats are also worth trillions of dollars annually through the range of ecosystem services they provide (Costanza 1998). Vegetated coastal habitats serve as nursery habitat for many fisheries species, supplying valuable nutrition for around 3.5 billion people (Nellemann et al. 2009). Seagrasses are also the primary food source for endangered species of turtles and dugongs. Saltmarshes and mangroves play a critical role in shoreline stabilisation, which is increasingly important with respect to sea-level rise and increasing frequency and intensity of extreme weather events associated with climate change (King & Lester 1995, Gedan et al. 2011). This service was particularly highlighted through a number of recent catastrophic events such as the December 2004 Indian Ocean tsunami (Danielsen et al. 2005, Kathiresan & Rajendran 2005, Alongi 2008) and Haiyan, the November 2013 typhoon that hit the Philippines (Gross 2014).
While the ecosystem benefits of saltmarsh, mangroves and seagrasses are relatively well-known, reliable data on their stocks of soil organic carbon are limited to sites within the Mediterranean, Northern Atlantic, and eastern Indian Oceans. Thus, our ability to estimate global carbon sequestration may be heavily influenced by values from these geographic regions (Fourqurean et al. 2012a), making it difficult to predict carbon storage levels in regions that have never been sampled. In addition, even for areas that have been sampled, available data indicates that considerable variation in organic carbon storage exists among locations (Fourqurean et al. 2012a). Variation in organic carbon storage has been attributed to multiple biological and environmental factors that can strongly influence the rate of organic carbon deposition (Lavery et al. 2013).

While substantial efforts are being made to understand and capitalise on carbon sequestration on land, the status of carbon stocks in vegetated coastal habitats is simply unknown in many regions of the globe (Nellemann et al. 2009). Improving our understanding of the factors influencing variability in carbon storage requires expanding the global dataset of carbon inventories. This study aimed to i) quantify belowground carbon in vegetated coastal habitats and ii) identify ‘hotspots’ (areas of above-average organic carbon storage) across the Victorian coastline in south eastern Australia.

This report summarizes the findings for blue carbon habitat stocks in the Port Phillip and Westernport catchment for the Port Phillip and Westernport Catchment Management Authorities (CMA).

**Methods**

**Site selection for blue carbon sampling**

To quantify and characterise the carbon sequestration capacity of blue carbon habitats across Port Phillip and Westernport, we relied heavily on existing habitat mapping to appropriately sample these habitats across the state. Saltmarsh and mangroves have been mapped comprehensively across the entire state of Victoria (Boon et al. 2010). In contrast to these habitats, the mapping of seagrass across the rest of Victoria was conducted between 12-18 years ago and only included estuarine or large embayment’s (Roob and Ball 1997, Roob et al. 1998, Blake et al. 2000, Blake and Ball 2001a, Blake and Ball 2001b). Hence, our sampling approach was informed by the existing mapping and in the case of seagrass, supplemented by our own knowledge of where seagrass habitats are likely to exist.

Saltmarsh, seagrass, and mangroves were all represented in the Port Phillip and Westernport carbon stock assessment sampling (Figure 1, 2). Saltmarsh dominated the sampling scheme (n=96), followed by mangroves (n=75) and then by seagrasses (n=60; Table 2). Seagrass distribution has likely changed over the past 15 years since the last major mapping program was undertaken. Further, the seagrass area recorded for the catchment is
an underestimate, as only inlet and estuarine seagrass was included (leaving out offshore seagrass).

Figure 1. Representatives of blue carbon habitats sampled in Victoria (a-i) and sediment coring techniques utilized to analyse soil carbon (j-l). Seagrass (a-c) samples included Zostera muelleri (a) and Zostera nigracaulus (b, c - with high sedimentation). Mangrove samples represent the one species of mangrove present in Victoria, Avicennia marina (d - mangrove plants, e - flowers, and f - seeds). Several types of saltmarsh habitats were sampled, including wet saltmarsh herbland (g - Sarconia sp., h - Suaeda sp.) and wet saltmarsh shrubland (i - Tecticornia sp.). Sediment cores were taken to 30cm deep using 50cm length PVC pipes (j), cores were extracted and sectioned in the lab (k), and sediment samples were dried, weighed, and analysed for organic carbon content (l).
Blue carbon habitats in Port Phillip and Westernport catchment were generally clustered in a few main areas, notably north-western Port Phillip Bay, northern Westernport Bay and Phillip Island (Figure 2). The number of cores sampled at each site reflected the types of blue carbon habitats present. For example, a site with only seagrass would result in three replicate cores, while a site with all three blue carbon habitats would have three sets of three replicates for each, and thus a total of nine. All sites were sampled between 17 July and 23 September 2014. Geographic location and 1 m² quadrat photos were taken at all sites. Quadrat photos were used to calculate percentage cover using the image analysis program CPCe (Kohler and Gill).

Figure 2. Blue carbon habitat sampling sites within the Port Phillip Westernport catchment. The majority of samples were taken from the north-western Port Phillip Bay (A), northern Westernport (B) and Phillip Island. Number of samples collected in each area are represented by the size of the orange circles on the main map (larger circles equal more samples collected), while blue carbon habitat types are represented in the inset maps by blue triangles (seagrass), red triangles (saltmarsh), and green triangles (mangroves).

Within blue carbon habitats, carbon is stored in living plant biomass for relatively short time scales (years to decades), while carbon sequestered in soils can be extensive and remain trapped for very long periods of time (centuries to millennia) resulting in very large carbon stocks (Duarte et al. 2005; Lo Iacono et al. 2008). As such, we focused on the belowground carbon pool by collecting soil sediment cores. Sediment cores were collected haphazardly within a given habitat location (with at least 50 m between each habitat core). Cores were collected via a piston corer, which involved hammering a PVC tube (50 mm internal diameter) into the sediment until a depth of 300 mm was reached and using suction from
the tightened piston located within the tube to hold the sediment in place while the tube was extracted. Subsequent processing of the cores was performed back in the laboratory.

**Sediment carbon content analyses**

The sediment in the tube was extruded and divided into the following sections 0-2, 14-16, and 28-30 cm. These samples were then placed into sterile plastic tubes and dried at 60°C for at least 120 hours. To enable carbon stock to be calculated, we first calculated dry bulk density (g cm⁻³) for each sediment depth by dividing the mass of the dried sediment by the original (pre-dried) volume of the sample.

After drying, all samples were homogenized by breaking up aggregates with an agate mortar and pestle. Samples were then quantitatively split down to 8 g subsamples which were finely ground on a Retch MM400 Mixer Mill using tungsten carbide grinding jars and balls. Samples were ground for 180 seconds at an oscillation frequency of 28 Hz, a duration determined to be necessary to produce a homogenous sample with repeatable mid infrared (MIR) spectra (Baldock et al. 2013).

Diffuse reflectance Fourier-transform MIR spectra across a spectral range of 8700-400 cm⁻¹ at 8 cm⁻¹ resolution were then obtained on all samples on a Thermo Nicolet 6700 FTIR spectrometer equipped with a Pike AutoDiff automated diffuse reflectance accessory following the protocols of Baldock et al. (2013). Spectra were then imported into the Unscrambler X ver 10.1 software as default OMNIC files. After Baseline Offset preprocessing and truncating the spectral range to 6000-1030 cm⁻¹, a principal components analysis (PCA) was used to visualize the variability in the total sample set. The Kennard-Stone Algorithm (Kennard and Stone 1969) was used on the first 6 principal components to pick the most representative 200 samples from the entire sample set.

Total carbon (TC), total organic carbon (TOC), total nitrogen (TN), and inorganic carbon (IC) were then determined in the laboratory on these 200 samples. All 200 samples were analysed for TC and TN by high temperature oxidative combustion on a LECO Trumac CN analyser at a combustion temperature of 1350°C and an extended purge and lance oxygen flows to ensure complete combustion of carbonate materials. For non-calcareous samples, determined by visual inspection of the MIR spectra (absence of a reflectance peak at 2500 cm⁻¹), TC = TOC and no further analyses were performed. For calcareous samples, carbonate removal was accomplished by acidification in 4% HCl. Two grams of sediment were weighed into 50 ml centrifuge tubes, 4% HCl was added slowly in 5 ml increments with vortexing between aliquots of acid. After 30 ml was added, tubes were capped and left on a shaker table overnight. Samples were then centrifuged at 3000 rpm for 8 minutes and supernatant was decanted. Samples were washed twice with 30 ml of ultrapure water followed by centrifuging before being lyophilized and reweighed. These acidified samples
were then run again on the elemental analyser for carbon and TOC data were reported back on original sample mass basis. For the calcareous samples, IC = TC – TOC.

The laboratory data were then used in a partial least squares regression (PLSR) to build algorithms which were then used to predict TC, TOC, TN and IC for the full set of samples. Full details of the MIR-PLSR procedure can be found in Baldock et al. (2013). Briefly, the PLSR models were built using the preprocessed MIR spectra and analytical data using a Random Cross Validation approach available in the Unscrambler 10.3 software (CAMO Software AS, Oslo, Norway). It was necessary to use square-root transformed TC, TOC, IC, and TN data in order to reduce non-linearity and improve homogeneity of residuals of calibration models. The quality of the derived PLSR models for predicting contents of TC, OC, IC, and TN was evaluated using a range of statistical parameters applied in the chemometric analysis of soil properties (Bellon-Maurel et al. 2010; Bellon-Maurel and McBratney 2011). The relationship between measured and PLSR predicted values was characterised by the slope, offset, correlation coefficient ($r$), R-squared, the root mean square error (RMSE), bias and the standard error (SE) of calibration (SEC) and validation (SEP). The ratio of performance to deviation (RPD) was used to define the quality of the derived models. To calculate RPD, the standard deviation ($s$) of measured samples used in the cross validation ($s_{CV}$) was calculated and divided by the appropriate standard error term (SE).

**Figure 3.** Relationship between measured and predicted data obtained for square-root transformed sediment OC contents.
Table 1. Summary statistics for MIR/PLSR models derived for square root transformed TC, OC, IC and TN content data (sqrt_TC, sqrt_OC, sqrt_IC and sqrt_TN, respectively) for Victorian coastline sediment samples.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Factors</th>
<th>n</th>
<th>Slope</th>
<th>Offset</th>
<th>r</th>
<th>$R^2$</th>
<th>RMSE $^2$</th>
<th>Bias</th>
<th>SE $^3$</th>
<th>s</th>
<th>RPD $^4$</th>
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<td>199</td>
<td>0.962</td>
<td>0.289</td>
<td>0.981</td>
<td>0.962</td>
<td>0.819</td>
<td>0.000</td>
<td>0.821</td>
<td>4.22</td>
</tr>
<tr>
<td></td>
<td>Validation</td>
<td></td>
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<td>0.354</td>
<td>0.978</td>
<td>0.956</td>
<td>0.887</td>
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<td>0.889</td>
<td>4.22</td>
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<td>0.735</td>
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<td></td>
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<td>199</td>
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<td>0.217</td>
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<td>0.086</td>
<td>0.974</td>
<td>0.949</td>
<td>0.251</td>
<td>0.005</td>
<td>0.252</td>
<td>1.11</td>
</tr>
</tbody>
</table>

$^1$ Only samples that gave a positive fizz test were used to generate the IC model.
$^2$ RMSE = RMSEC for calibration samples and RMSEP for validation samples
$^3$ SE = SEC for calibration samples and SEP for validation samples
$^4$ RPD = RPD$_c$ for calibration samples and RPD$_p$ for validation samples.
Chang et al. (2001) suggested that RPD values >2, between 1.4 and 2, and <1.4 could be used to distinguish excellent, fair, and non-reliable models, respectively. The Unscrambler software determined the optimum numbers of factors for each calibration model and $n$ refers to the number of observations included in each analysis. Each calibration model was then used to predict values of sqrt TC, sqrt OC, sqrt IC, and sqrt TN for all sediment samples. It was necessary to back transform all data to obtain values for TC, OC, IC and TN. All data are reported as g C (or N) per kg sediment mass. An indication of the confidence of the prediction is given by the deviation statistic. This was used with the back transformed data to produce an upper and lower limit value for prediction which approximates a 95% confidence interval. In addition, each predicted value’s status as an outlier was assessed by two measures: Inlier distance and Hotelling’s $T^2$ distance. The inlier distance assessed the distance of the predicted value to the nearest calibration value, while the Hotelling’s $T^2$ distance assessed the distance to the centre of the calibration values. If the ratio of the inlier distance to the inlier limit for a predicted value exceeded 1.0, the predicted value was identified as being an outlier in the sense that it was too far distant from the nearest calibration point. Similarly, if the ratio of the Hotelling’s $T^2$ distance to the Hotelling’s $T^2$ limit exceeded 1.0, the predicted value was identified as being an outlier in the sense that it was too distant from the centre of the calibration set.

**Total carbon stock calculations**

We followed the approach for calculating total sedimentary carbon stock as outlined by Howard et al. (2014). For each interval of the core analysed, we calculated the sedimentary organic carbon density as follows:

**Step 1.**
Soil carbon density ($g/cm^3$) = dry bulk density ($g/cm^3$) $\times$ (% $C_{org}$/100)

We then calculated the amount of carbon present in each section of the core by multiplying the soil carbon density value obtained in step 1 by the thickness of the core section (2 cm):

**Step 2.**
Carbon content in core section ($g/cm^3$) = Soil carbon density ($g/cm^3$) $\times$ Thickness of core section (2 cm)

As subsamples were taken along the core, we averaged the amount of carbon in each of the sections and then multiplied over the total depth sampled to get the total carbon stock. We then converted the total core carbon into MgC/hectare using the following unit conversion factors: 1,000,000 g = 1 Mg (megagram), and 100,000,000 cm$^2$ = 1 hectare:

**Step 3.**
Total sedimentary carbon (MgC ha$^{-1}$) = Averaged core carbon ($g/cm^3$) $\times$ (1 Mg/1,000,000 g) $\times$ (100,000,000 cm$^2$/1 hectare)
Replicate cores for each habitat within a single location were averaged to obtain an estimate of the carbon stock within the habitat at a given location. These carbon stock estimates were then averaged by habitat across all locations to estimate the average amount of carbon per habitat within the catchment. To calculate the total carbon stock across the catchment, we multiplied the average carbon value for each habitat (MgC ha\(^{-1}\)) by the total area of each habitat (in hectares) in the catchment, then summed the total carbon values for each habitat to determine the total sedimentary carbon stock in all the blue carbon habitats in Corangamite.

**Carbon Hotspot Analysis**

Hotspots were identified in Port Phillip and Westernport catchment and generally reflect trends in organic soil carbon stock related to habitat type, patch size, and location. Maps conveying estimates of organic carbon (tonnes, i.e. Mg) by habitat patch were derived using field samples of average carbon stock by habitat, combined with existing habitat maps of coastal and aquatic vegetation in the Port Phillip and Westernport catchment. Geographic distribution of intertidal vegetation including saltmarsh and other wetland habitat was extracted from Boon *et al.* (2010) describing the condition and extent of Ecological Vegetation Classes (EVCs). The distribution of seagrass vegetation in Port Phillip and Westernport was extracted from habitat maps produced by Blake and Ball (2001a, 2001b). Mean organic carbon stock was calculated from sediment core replicates in the Port Phillip and Westernport catchment (See locations - Figure 2). Using core samples for each habitat type organic carbon values (Mg C ha\(^{-1}\)) were extracted at different locations within north-western Port Phillip Bay, northern Westernport Bay and Phillip Island. ArcMap 10.1 was used to georeference sample cores with habitat types at different locations within each inlet. Carbon stock was then extrapolated to all habitat patches and an estimate of organic carbon tonnes for each patch was calculated by multiplying carbon density estimates by patch area.
Table 2. Summary of site locations, including site region, shortened site name, habitat (SM=Saltmarsh, MG=mangrove, SG=Seagrass (Zostera), SG*= Lepilaena, BANK=Eroding bank), GPS coordinates, average percent cover of salt marsh habitat within a square-meter quadrat, number of cores taken, average carbon soil stock, and average monetary value of soil carbon stocks.

<table>
<thead>
<tr>
<th>Site Name</th>
<th>Region</th>
<th>Shortened site name</th>
<th>Habitats Sampled</th>
<th>Latitude (°N)</th>
<th>Longitude (°E)</th>
<th>Saltmarsh % cover</th>
<th># Cores</th>
<th>Mg C$_{org}$ ha$^{-1}$ (top 30cm)</th>
<th>Average $$ value ha$^{-1}$ (at $15$ Mg$^{-1}$)</th>
</tr>
</thead>
<tbody>
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<td>Port Phillip</td>
<td>WBS</td>
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<td>41.653</td>
<td>625</td>
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<td>Port Phillip</td>
<td>ALT</td>
<td>SM</td>
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<td>88.56</td>
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<td>Port Phillip</td>
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<td>160.577</td>
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<td>Northern Western Port</td>
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<td>MG, SM</td>
<td>-38.21373333</td>
<td>145.3775694</td>
<td>-</td>
<td>6</td>
<td>70.375</td>
<td>1,056</td>
</tr>
<tr>
<td>Lang Lang</td>
<td>Northern Western Port</td>
<td>LANG</td>
<td>BANK, SM</td>
<td>-38.26080833</td>
<td>145.4950083</td>
<td>96.45</td>
<td>4</td>
<td>97.775</td>
<td>1,467</td>
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<tr>
<td>Churchill Island</td>
<td>Phillip Island</td>
<td>CHU</td>
<td>MG, SG, SM</td>
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<td>145.3446426</td>
<td>-</td>
<td>9</td>
<td>58.532</td>
<td>878</td>
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<tr>
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<td>Phillip Island</td>
<td>RHY</td>
<td>SG</td>
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<td>145.3102111</td>
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<tr>
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<td>RHYI</td>
<td>MG, SG, SM</td>
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<td>145.2895833</td>
<td>-</td>
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Results and Discussion

Soil sample analysis

The median dry bulk density (DBD) of the sediment samples (0.78 g cm$^{-3}$) was lower than the global median (0.92 g cm$^{-3}$) (Figure 4a). Visually, samples varied across depth, habitat, and site in amount of plant material (including peat), sediment grain size, compaction, and soil composition.

Figure 4. Frequency of soil sample (a) dry bulk density (DBD, g cm$^{-3}$, n=237) and (b) organic carbon content (% carbon, n=237) samples across Port Phillip Westernport catchment (pooled by depth and habitat). The global median for DBD and percent carbon are indicated by dashed lines, correspondingly, according to Campbell et al. (2014).
The median percent organic carbon of all samples pooled (3.62%) was higher than the global median (1.4%, Figure 4b), with the median saltmarsh and mangrove percent organic carbon much higher at 5.52% and 4.23%, respectively. Although there was a large range in soil carbon levels by depth, the median values showed a step-wise decrease in soil carbon with depth (Figure 5a). There were clear differences in soil carbon levels based on habitat type, with saltmarsh habitats having the highest average and the largest range of soil carbon (Figure 5b). Saltmarsh is closely followed by mangroves and then seagrass in average percent carbon. This is consistent with the results from other catchments within Victoria.

**Carbon stocks across sites and carbon hotspots**

Carbon stocks varied across sampling site locations in the Port Phillip and Westernport catchment, but all vegetated coastal habitats sampled represent valuable carbon sinks, providing both ecological and economic benefits (Table 3). The differences in carbon storage (and the resulting differences in monetary value) across adjacent sampling sites reflects the high variability in carbon stocks within small geographic scales. Notably, the highest soil carbon values were often associated with sites located higher up in estuarine areas (or closer to fluvial inputs), such as Westgate Bridge, Warneet, Yaringa, Hastings and Rhyll Inlet (Figure 5c). Interestingly, the mangroves at the West Gate Bridge location were planted in 1984. While the surface samples have similar carbon densities as adjacent, natural mangroves (Figure 7), the deeper sections of the cores (14-16 cm and 28-30 cm) at this site have the highest carbon density values of any mangroves in the catchment. This suggests that these sites have been operating similar to one another in recent times, but in the past, the West Gate Bridge location had much higher carbon storage than surrounding mangroves. The position of these mangroves at the junction of two estuaries (Stony Creek and the Yarra River) is the likely reason for the high carbon values. Unfortunately, this site is also often impacted by chemical or waste discharges.

**Table 3.** Summary of habitats sampled, including total habitat area within Port Phillip Westernport, total habitat area within Victoria, and percentage of the habitat in Port Phillip Westernport relative to all of Victoria. Carbon soil stocks (in the top 30 cm) in Port Phillip Westernport across each habitat are based on the average carbon storage multiplied by the total habitat area within Port Phillip and Westernport. Habitat area estimates are based on state-wide mapping performed by Boon et al. (2010), Blake and Ball (2001a, b) and French et al. (2014). Seagrass mapping has not been completed across the entire state of Victoria. Estimates of seagrass distribution in Port Phillip and Westernport do not include coastal seagrass, and are, therefore, likely to be underestimates of seagrass extent.

<table>
<thead>
<tr>
<th>Habitat type</th>
<th>Port Phillip Westernport Area (Ha)</th>
<th>Victorian Total Area (Ha)</th>
<th>Port Phillip Westernport as % of state</th>
<th>Port Phillip Westernport Mg C&lt;sub&gt;org&lt;/sub&gt;</th>
<th>Total $ value (at $15 per Mg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Saltmarsh</td>
<td>3,108.3</td>
<td>20,625.74</td>
<td>15.07</td>
<td>322,060.17</td>
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<td>Mangrove</td>
<td>1,827.7</td>
<td>5,186.58</td>
<td>35.24</td>
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<td>Seagrass</td>
<td>14,456.9</td>
<td>42,812.63</td>
<td>33.76</td>
<td>551,552.17</td>
<td>8,273,282</td>
</tr>
<tr>
<td><strong>Grand Total</strong></td>
<td><strong>19,392.9</strong></td>
<td><strong>68,624.96</strong></td>
<td><strong>100%</strong></td>
<td><strong>1,025,203.23</strong></td>
<td><strong>15,378,048</strong></td>
</tr>
</tbody>
</table>
Figure 5. Carbon stock (mg C$_{org}$ cm$^{-3}$) in blue carbon habitat soils by (a) depth, (b) habitat type, and (c) sampling location (west to east from left to right) within Port Phillip and Westernport catchment. Box plots represent the minimum and maximum values (tails), the middle 50% range (box), and the median (line bisecting box) for each sample group. Mean carbon stock values are represented by “x” symbols.
Figure 6. Carbon heat maps for two regions of blue carbon habitats in Port Phillip Westernport catchment, (a) northern Port Phillip and (b) northern Westernport. Blue carbon values represent tonnes of organic carbon stored in the top 30cm of the soil in seagrass, mangrove, and saltmarsh habitat areas. Note different scales between (a) and (b).
The Carbon Hotspot Analysis is a combination of both the soil carbon content and the size of particular habitat patches. It provides a method for identifying high value habitat patches using spatial information. The current analysis highlights that the seagrass, mangroves, and saltmarsh in the upper regions of Westernport (Warneet) are the highest value blue carbon locations (Figure 6b). In Port Phillip Bay, the Cheetham wetlands and Laverton Creek saltmarsh represent the most significant carbon stocks (Figure 6a). Of these two locations, the Laverton Creek saltmarsh is the most vulnerable, as it is currently unprotected and is hemmed in by railway tracks, housing, and industrial development.

![Average (± SEM) organic carbon density (mg C\textsubscript{org} cm\textsuperscript{-3}) at the revegetated West Gate Bridge site (blue) and the nearest site at Jawbone Marine Sanctuary (green) at the three depths sampled (0-2 cm, 14-16 cm, and 28-30 cm).](image)

**Figure 7.** Average (± SEM) organic carbon density (mg C\textsubscript{org} cm\textsuperscript{-3}) at the revegetated West Gate Bridge site (blue) and the nearest site at Jawbone Marine Sanctuary (green) at the three depths sampled (0-2 cm, 14-16 cm, and 28-30 cm).

Many of the vegetated coastal habitats in the Port Phillip and Westernport catchment are vulnerable to disturbance or other anthropogenic influences (Keough *et al.* 2011). High visitation pressure leading to habitat degradation, loss or fragmentation, nutrient and sediment inputs from the catchment, and hydrological modification can all lead to losses of vegetated coastal habitats.

The Lang Lang site in north-eastern Westernport appears to be the most vulnerable to losses of sediment carbon stock. A large portion of the bank in this area is eroding (Figure 8a), with an estimated 270,000 Mg of sediment released into Westernport since 1947 at an annual rate of 4200 ± 2900 Mg (Tomkins *et al.* 2014). Using measurements of percent organic carbon from our sampling, both in the saltmarsh and at the eroding edge of the bank at Lang Lang, we calculated the loss of carbon associated with the eroded sediment. Based on these estimates, 15,724 Mg of carbon have eroded with the sediment since 1947, at a rate of 244 ± 168 Mg of carbon per year. While the fate of sediment carbon once it has eroded into Westernport Bay is unclear, a large proportion of it is likely to be released into the environment as carbon dioxide. Applying the current market value of $15 per Mg C to that which has eroded at Lang Lang, the monetary carbon loss equates to a total of $235,860, or $3,660 ± 2520 per year. Efforts to curb this erosion through
mangrove planting and erecting physical barriers have thus far been unsuccessful (Hurst 2013, Tomkins et al. 2014).

Figure 8. The eroding Lang Lang coast in the north-eastern section of Westernport Bay (a) and dyke/bund walls reinforced with concrete rubble on Cannons Creek in Westernport (b).

Many other vegetated coastal habitat locations are under threat from anthropogenic influences. The saltmarsh and mangroves at Hastings were among the highest sediment carbon stocks within the catchment. Yet, saltmarsh, mangroves, and seagrass at Hastings are potentially under threat from the planned Port of Hastings development. While the project is in the planning and approval phase, any proposal to remove or negatively impact these vegetated coastal habitats (and thus their carbon stocks) should be taken into account before proceeding.

Further, the effects of climate change on these valuable carbon stocks should be noted. There are a number of locations within Port Phillip and Westernport where the saltmarsh and mangroves have limited capacity to move shoreward with sea-level rise (Figure 8b), which is typical of much of northern and western Westernport. This should be a major consideration to ensure the preservation of this carbon stock into the future. These are just a few of the many threats that these vegetated coastal habitats face in Westernport, which were recently reviewed by Keough et al. (2011). Disturbance and loss of blue carbon habitats can shift them from serving as powerful carbon sinks to major sources of carbon emissions to the atmosphere.

Despite the relatively lower carbon values per hectare of seagrass, the large area of this habitat type in Port Phillip and Westernport means it makes up a substantial portion of the carbon stock in the catchment (Table 3, Figure 5). However, the long time period since the habitat was last mapped at the bay-wide scale and the fact that no offshore seagrass has been mapped in the region means that updated mapping information is required for the total carbon stock estimate to be accurate. While loss of this habitat results in decreased potential for future carbon sequestration, we have little understanding of how belowground carbon stock is affected by seagrass loss. With significant seagrass decline in Westernport in the late 1970s and early 1980s (Bulthuis 1984), this would have had major implications for the carbon storage and stock in
Westernport, including the potential release of large amounts of previously stored carbon. Recovering these lost habitats, or at least avoiding future declines, will have large benefits for the Port Phillip and Westernport catchment’s carbon stock.

**Major conclusions**

The Port Phillip and Westernport catchment contains a significant portion of the blue carbon ecosystems present across Victoria. The information gathered for this report can help understand the current blue carbon stocks within vegetated coastal habitats in the Port Phillip and Westernport catchment. This can be used to conserve high-priority areas, and plan for future protection and restoration of these habitats for carbon sequestration and myriad ecological purposes. The highest-value carbon stock areas (‘hotspots’) in the Port Phillip and Westernport catchment have been identified as either areas of dense soil carbon stock (e.g. mangroves at the West Gate Bridge) and/or areas with extensive ecosystem distribution (e.g. seagrass in Westernport). Saltmarshes have the highest carbon stock per unit area of all the blue carbon habitats, followed closely by mangroves. However, seagrasses in Port Phillip and Westernport store 54% of the catchment’s sedimentary carbon due to their large distribution. Therefore, the largest potential for soil carbon loss is through decline in seagrass habitat. Because most of these area estimates are based on reports of seagrass distribution from 14 or more years ago (Blake and Ball 2001a, Blake and Ball 2001b, but see French et al. 2014), updated mapping is essential to estimate seagrass carbon stocks and ensure effective management of the blue carbon in Port Phillip and Westernport.

Given the high efficiency of saltmarsh and mangrove ecosystems in Port Phillip and Westernport to sequester high densities of carbon, revegetation or protection programs centred on these habitats may be the most cost efficient (in terms of potential carbon storage per area). Indeed, there are a number of “at risk” locations or large areas of habitat that could be restored to their pre-European distribution (Boon et al. 2010). These include sections of north-western Port Phillip and much of the northern coastline of Westernport.

With a growing Australian push to ‘get blue carbon to market’, we recommend further research into opportunities for blue carbon offset projects within the Port Phillip and Westernport catchment through strategic restoration of former blue carbon habitats (e.g. bund/dyke wall removals), and through management of catchment-level processes to enhance blue carbon sequestration within existing habitats (e.g. restoring natural hydrology). Though the goal of such activities would be carbon enhancement, there would be broad environmental, social, and economic benefits (e.g. biodiversity and fisheries enhancement, shoreline stabilisation, climate change buffering, and improved shoreline amenity).

While not covered as part of this report, freshwater wetlands (which include alpine peatland, freshwater wetland and coastal wetlands) are also thought to be significant carbon sinks. Though they only represent about 4% of terrestrial land area, it’s estimated that freshwater wetlands are storing about 33% of the carbon in terrestrial soils (Euliss et
However, there is currently little known of carbon storage in these habitats in Port Phillip & Westernport. Sampling in the Glenelg-Hopkins CMA suggests that these habitats store similar quantities of carbon to saltmarsh and mangroves. Some concerns regarding natural methane release in freshwater wetlands have arisen, but in the long run, the benefits of carbon dioxide storage appear to outweigh the costs associated with carbon release in the form of methane (Euliss et al. 2006). Further research into the distribution and carbon storage of these habitats is required to determine the role these habitats are playing in Port Phillip & Westernport and how they might add to the already significant blue carbon stock.
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