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Carbon stocks, sequestration, and emissions of wetlands in south eastern Australia

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Abstract

Nontidal wetlands are estimated to contribute significantly to the soil carbon pool across the globe. However, our understanding of the occurrence and variability of carbon storage between wetland types and across regions represents a major impediment to the ability of nations to include wetlands in greenhouse gas inventories and carbon offset initiatives. We performed a large-scale survey of nontidal wetland soil carbon stocks and accretion rates from the state of Victoria in south-eastern Australia—a region spanning 237,000 km^2 and containing >35,000 temperate, alpine, and semi-arid wetlands. From an analysis of >1,600 samples across 103 wetlands, we found that alpine wetlands had the highest carbon stocks (290 ± 180 Mg C_{org} ha^{-1}), while permanent open freshwater wetlands and saline wetlands had the lowest carbon stocks (110 ± 120 and 60 ± 50 Mg C_{org} ha^{-1}, respectively). Permanent open freshwater sites sequestered on average three times more carbon per year over the last century than shallow freshwater marshes (2.50 ± 0.44 and 0.79 ± 0.45 Mg C_{org} ha^{-1} year^{-1}, respectively). Using this data, we estimate that wetlands in Victoria have a soil carbon stock in the upper 1 m of 68 million tons of C_{org}, with an annual soil carbon sequestration rate of 3 million tons of CO_2 eq. year^{-1}—equivalent to the annual emissions of about 3% of the state's population. Since European settlement (~1834), drainage and loss of 260,530 ha of wetlands may have released between 20 and 75 million tons CO_2 equivalents (based on 27%–90% of soil carbon converted to CO_2). Overall, we show that despite substantial spatial variability within wetland types, some wetland types differ in their carbon stocks and sequestration rates. The duration of water inundation, plant community composition, and allochthonous carbon inputs likely play an important role in influencing variation in carbon storage.

KEYWORDS
blue carbon, carbon profile, climate change, greenhouse gas emissions, soil carbon, wetland communities

1 | INTRODUCTION

Wetlands are among the most productive ecosystems in the world. They support a unique biodiversity (Strayer & Dudgeon, 2010) and provide vital ecosystem services, such as nutrient cycling, erosion control, and flood mitigation (Mitra, Wassmann, & Vlek, 2005; Mitsch & Gosselink, 2007; de Groot et al., 2012; Costanza et al., 2014; Russi et al., 2013). In addition, rising carbon dioxide levels in the atmosphere have brought attention to wetlands' capacity to sequester organic carbon. By taking up carbon during photosynthesis and burying undecomposed plant litter in anaerobic soils, freshwater wetlands sequester 1.38–2.26 t C_{org} ha $year^{-1}$ in soils, an average of 34–44 times more carbon than do terrestrial forests (Bernal & Mitsch, 2012; McCleod et al., 2011). Global estimates suggest that freshwater wetlands contain 20%–30% of the terrestrial soil carbon pool, a disproportionately high contribution given that they occupy a mere 6%–8% of the land surface (Lal, 2008; Lal, Follett, Stewart, & Kimble, 2007; Mitsch & Gosselink, 2007; Mitra et al., 2005).

Despite this estimated contribution, most studies on freshwater wetland carbon stocks and sequestration have been focused on just a few sites (Bernal & Mitsch, 2012; Whitaker et al., 2015). One limitation of making nation-wide or global estimates of wetland carbon stocks based off these values, is that the most carbon-rich sites have often been studied. As such, there has been a renewed push, to estimate broadscale wetland carbon stocks with broadscale sampling efforts (Nahlik & Fennessy, 2016). Even within a region, there is high variability in carbon sequestration capacity among wetland types, contributing even further to uncertainty in carbon stocks and sequestration. For example, in Ohio, US, depressional wetlands sequester two times more carbon than riverine communities (Bernal & Mitsch, 2012). Furthermore, different plant communities can also influence carbon sequestration rates within each wetland type (Mitsch et al., 2013; Villa & Mitsch, 2015). A high priority for research is therefore to quantify variability in carbon storage and sequestration among wetland types, within and between regions.

Quantifying the contribution of wetland ecosystems to carbon capture and storage is vital to justify the application of improved management strategies that ensure their continued contribution to a multitude of other ecosystem services. For despite the importance of wetlands, they have been historically underappreciated, and an estimated 87% of global wetland area has been lost since the early 1700s (Davidson, 2014). They are threatened by land-use change, pollution, water extraction, and landscape modification (MEA, 2005, Moser, Prentice, & Frazier, 1996; Van Asselen, Verburg, Vermaat, & Janse, 2013; Vörösmarty, McIntyre, Gessner, Dudgeon, & Prusevich, 2010). Such disturbances can undermine a wetland's ability to capture carbon, but critically, result in microbial breakdown, demineralization and ultimately release of significant amounts of carbon that had *already* been stored (Atwood et al., 2017; Lal et al., 2007; Page & Dalal, 2011; Pendleton et al., 2012). Conversion to agricultural land for cropping and grazing can lead to 80%–96% reduction in wetland soil organic carbon (Meyer, Baer, & Whiles, 2008; Sigua, Coleman, & Albano, 2009). By providing estimates on wetland carbon stocks, and wetland loss, we can use this information to estimate the carbon losses from wetland degradation.

This study was motivated by the Victorian Catchment Management Authorities and the Victorian State Government—the Department of Environment, Land, Water and Planning (DELWP)—to increase the understanding of the values of wetlands and their capacity to sequester carbon within the state of Victoria (hereafter referred to as "Victoria"), in south eastern Australia. Our specific objectives were to: (a) Undertake a landscape-scale assessment of wetland soil carbon stocks by **s**ampling wetland soil carbon from more than 100 wetlands spread across Victoria; (b) compare carbon stocks between wetland types and assess regional variability; (c) Utilize existing data on wetland soil accretion rates, combined with newly collected soil carbon data to estimate wetland soil carbon sequestration; and (d) based on this data, estimate soil carbon stocks and sequestration capacity for the entire region and the potential impact of historical wetland loss on carbon emissions from wetlands.

2 | MATERIALS AND METHODS

2.1 | Sample collection

This study was conducted across the state of Victoria in southeastern Australia (Figure 1). Victoria has approximately 530,400 hectares of wetlands, and is split up into ten catchment management regions. An estimated 147,053 ha of wetlands (or roughly 27%) have been lost since European settlement (~1834) (Papas & Moloney, 2012).

To evenly spread the sampling effort across the state, we sampled at least 10 wetlands in each of the 10 catchment regions, with a final total of 103 wetlands (Figure 1). Here, wetlands are defined as surface waters, whether natural, modified or artificial, subject to permanent, periodic or intermittent inundation, which hold static or very slow moving water and support biota adapted to inundation and the aquatic environment. Wetlands were chosen across six key wetland categories defined by Corrick and Norman (1980), but we have also provided reference to the equivalent RAMSAR wetland classifications: **freshwater meadow** (Ts—Seasonal/intermittent freshwater marshes/pools on inorganic soils), **shallow freshwater marsh** (W—Shrub-dominated wetlands; Xf—Freshwater, tree-dominated wetlands; Xp—Forested peatlands), **deep freshwater marsh** (Tp—Permanent freshwater marshes/pools), **permanent open freshwater** (O—Permanent freshwater lakes (over 8 ha), **saline wetlands** (Q—Permanent saline/brackish/alkaline lakes; R—Seasonal/intermittent saline/brackish/alkaline lakes and flats), and **alpine wetlands** (Va—Alpine wetlands). While we tried to evenly sample the different wetland types in each of the 10 catchment regions, our sampling of each wetland type was proportional to the total area within catchment region, with some wetland types not present in some catchments (i.e. alpine or saline wetlands). Sampling was conducted between August 2015 and February 2016. We collected three soil cores at each of the wetlands. At each site we selected a single dominant vegetation stratum to sample within, to ensure soil was collected from areas with the same overlying litter material as well as similar inundation level. Cores were taken 50 m aside from one another to capture potential within site variability.

At each of the three sampling points, a 5 cm (inner-diameter) PVC pipe (the core) was hammered into the soil until 1 m was reached, or until core refusal (no further penetration). The depth to which wetlands could be sampled varied considerably, and was mostly determined by the depth at which the organic layer finished and a thick clay layer was reached. For some areas this was as shallow as 0.16 m (typically temporary open water bodies), while other wetlands contained organic peat material well past 1 m. Therefore,

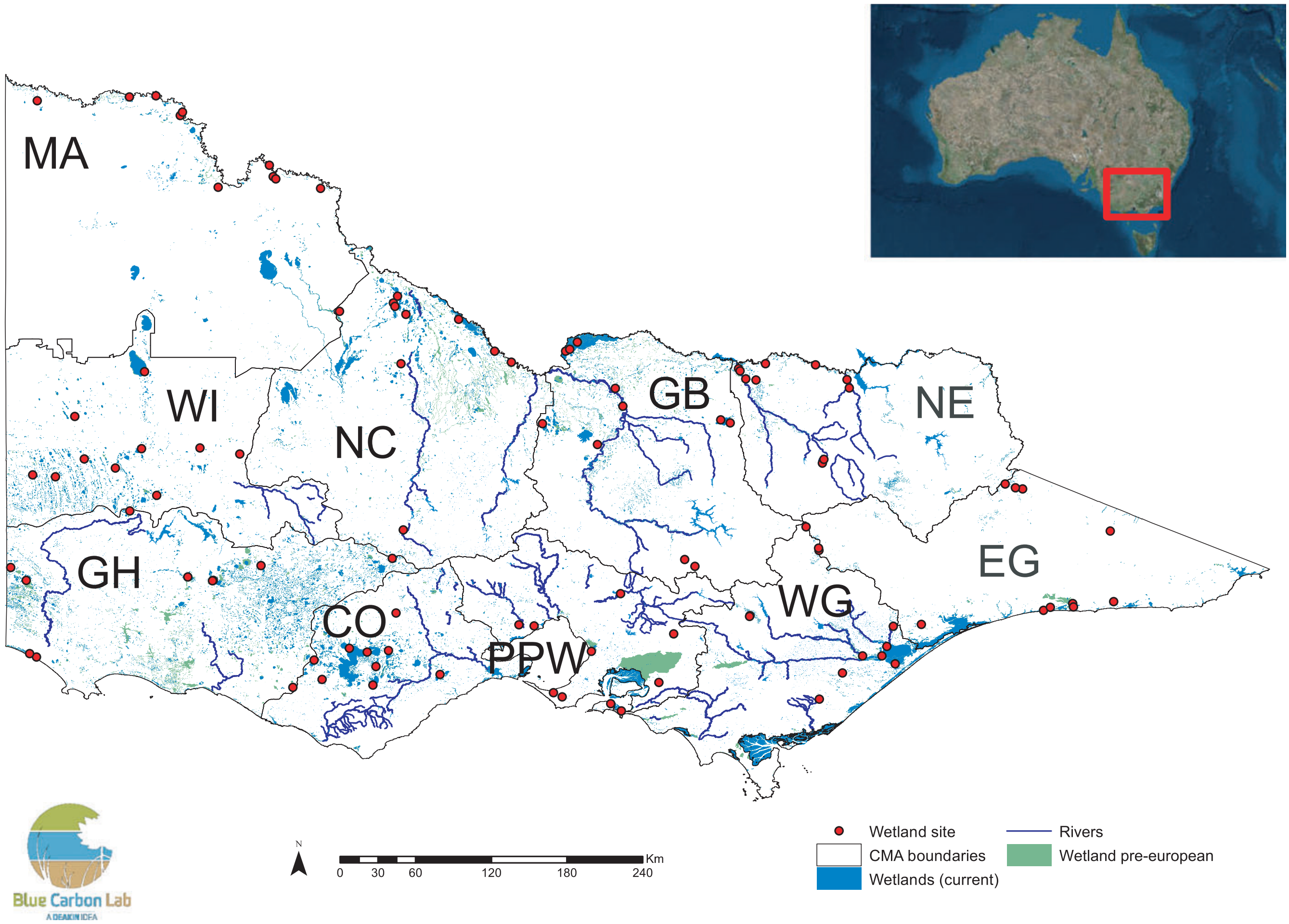

FIGURE 1 Extant wetlands and waterbodies in Victoria and the location of the 103 wetlands sampled in this study. The ten catchment regions are displayed: MA = Mallee, NC = North Central, GB = Goulburn Broken, NE = North East, EG = East Gippsland, WG = West Gippsland, PPW = Port Phillip & Westernport, CO = Corangamite, GH = Glenelg Hopkins

the carbon stock reported for each wetland was based on the actual core depth sampled to a maximum of 1 m, which includes all of the highly organic material.

We accounted for soil compaction during the coring process, by measuring the difference between the amount of soil sampled and the depth the core was hammered into the ground. Once depth at refusal was reached, a rubber plug was inserted into the top of the core to create a vacuum seal. After removing the core, a foam plug was inserted on top of the soil to stabilize sediment during transport. Both ends of the core were capped with a plastic cover and sealed with plastic tape to prevent moisture loss.

2.2 | Sediment carbon analysis

In the laboratory, soil, including dead plant material, from the three replicate cores from each site were extruded and sectioned at 0–2 cm, 12–14 cm, 28–30 cm, 48–50 cm, 74–76 cm, and 98–100 cm for carbon stock calculations. When core refusal was reached shallower than 1 m, an additional sample was taken at the deepest depth. For example if the core depth was 60 cm, the four samples from 0 to 50 were taken, plus one extra at 58–60 cm. The sections were dried to constant weight at 50°C for 48–72 hr. Dry weight was used to calculate sediment bulk density and the sectioned samples were homogenized with a stainless steel mortar and pestle (Retch RM200).

At five of the 103 sites, a fourth core was sectioned every centimeter from 0 to 20 cm for ^{210}Pb dating. The sections for ^{210}Pb dating were split in half, with one half to be age dated, and the other reserved for carbon analysis, so that carbon sequestration rates could be calculated.

As some of the samples were collected from declared Phylloxera Infested Zones (areas infected with an aphid-like insect pest of grapevines (*Daktulosphaira vitifoliae*), all cores were gamma irradiated at 50kGray for 60 hr (by Steritech, Dandenong, Victoria), before being sent to CSIRO in South Australia for analysis. This irradiation process has been demonstrated to have no major influence on carbon measurements in soil samples (Baldock et al., unpublished data).

Based on the protocols of Baldock, Hawke, Sanderman, and Macdonald (2013) (see Supporting Information Data S1 for more detail), a Thermo Nicolet 6700 FTIR spectrometer equipped with a Pike AutoDiff automated diffuse reflectance accessory was used to obtain diffuse reflectance Fourier-transform MIR spectra for all samples across a spectral range of 8000–400 cm^{-1} with a spectral resolution of 8 cm^{-1}. Using Unscrambler 10.3 software (CAMO softare, Olso, Norway), a principle component analysis was used to visualize spectral variability across the set of 1672 samples included in this study. A Kennard-Stone Algorithm (Kennard & Stone, 1969) was used to identify 286 samples that were most representative of the spectral variability exhibited across all samples.

The total carbon content of all 286 samples was determined by analyzing approximately 0.80 g of the dried and ground soil on a LECO Trumac CN analyzer. Among the 286 samples, 52 were identified to contain inorganic carbon by the presence of a reflectance peak at 2,500 cm^{-1} and were labeled as calcareous. The calcareous soils were pretreated by addition of 1 ml of 1 M HCl to a known mass of dried and ground soil, shaken and centrifuged with the supernatant being retained for each sample. The HCl pretreatment was repeated until no further effervescence was detected. The soils were then washed three times with 50 ml of deionized water with centrifuging and collection of the supernatant between washes. After the last wash the soils were frozen, freeze dried, and analyzed again on the LECO trumac CN analyzer. All supernatants (after 1 M HCl treatment and water washes) were accumulated for each sample and the amount of dissolved organic carbon present in the bulk supernatant was determined using a Thermalox DOC analyzer. The C_{org} content of the calcareous soils was determined as the total carbon measured by the LECO on the freeze dried HCl pretreated soil plus the organic carbon present in the accumulated supernatants corrected back to the original mass of soil that was pretreated.

The C_{org} content of all 286 samples were combined with their respective MIR spectra and a partial least squares regression (PLSR) analysis was used to develop an algorithm capable of predicting C_{org} content from MIR spectra. A square root transformation was applied to the C_{org} contents to correct for nonlinearity and an inhomogeneity of variance in the resultant model. A full cross validation process was used to validate the PLSR prediction algorithm derived for the square root of C_{org} content. The statistics associated with the PLSR prediction algorithm are provided in Table S1. The PLSR prediction algorithm was then applied to the MIR spectra derived for all 1,672 samples and the values derived for the square root of C_{org} content were squared to provide values for C_{org} content of the samples.

2.3 | Total C_{org} stock and C_{org} density

The depth of refusal at each site varied (some as shallow as 16 cm), but most cores reached at least 50 cm. For each core with more than three depth points (>30 cm depth) cubic splines were used to calculate the C_{org} density (g C_{org} cm^{-3}, calculated by multiplying the sample dry bulk density (DBD) (g/cm^3), by the C_{org} contents (%C_{org})) at intermediate depth ranges that were not analyzed. Although cubic splines are commonly preferred for soil carbon analysis (e.g. Minasny, McBratney, Mendoca-Santos, Odeh, & Guyon, 2006), linear splines are more appropriate for cores where there are only two or three data points per core. Thus, for cores that had C_{org} density for ≤3 depths (where cores only reached 16 or 30 cm) linear splines were used. Using the spline approach, C_{org} density was calculated for each 2 cm depth increment down the entire soil profile sampled. Splines were not used to extend carbon density estimates beyond the depths sampled.

The value of Victoria's total carbon stock was calculated through the following equation:

$$\mathcal{T} = \sum_i \bar{s}_i \times \alpha_i \tag{1}$$

where $\mathcal{T}$ represents the total carbon stored in Victorian wetlands to maximum 1 m depth, $\bar{s}_i$ represents the mean wetland type carbon stocks within each region (Mg C_{org} ha^{-1}, 1 m depth), and α_i represents the total area of each wetland type in each region. This study did not sample all wetlands types in all regions, so to calculate total stocks in these regions, the average carbon stock values of all regions were used as a proxy for $\bar{s}_i$ in the equation stated above.

2.4 | C_{org} sequestration rates

The profiles of ^{210}Pb concentrations were determined for one sediment core at five wetlands. Analysis of its decay product ^{210}Po, in equilibrium with ^{210}Pb, was used to determine the concentrations of ^{210}Pb (Sanchez-Cabeza, Masque, & Ani-Ragolta, 1998). Soil samples were spiked with known amounts of ^{209}Po, acid digested, and the polonium isotopes were plated onto silver disks and their emissions were measured by alpha spectrometry using Passivated Implanted Planar Silicon detectors (CANBERRA, Mod. PD-450.18 A.M). The concentrations of ^{210}Pb were calculated by applying appropriate decay corrections to sampling time and accounting for reagent blanks and detector backgrounds, which were both almost negligible. Analyses of replicate samples and reference materials were carried out in parallel. The supported ^{210}Pb for each core was determined from the average of the deepest samples analyzed where concentrations of ^{210}Pb were constant (ranging from 10 to 20 Bq/kg for the various cores). Excess ^{210}Pb was determined by subtraction of the supported concentration from the total ^{210}Pb at each section and the Constant Flux: Constant Sedimentation (CF:CS) and the Constant Rate of Supply (CRS) models were applied to determine the sedimentation rates over the last 100 years (Appleby & Oldfield, 1978; Krishnaswami, Lal, Martin, & Meybeck, 1971).

To determine carbon sequestration (g C m^{-2} $year^{-1}$), sediment vertical accretion rate (cm/year) was multiplied by the carbon stock (g C_{org} cm^{-3}) over the dated soil depth. The depths of dated samples were within the upper 20 cm, which represents an age range of over the last 76–116 years (Table S2). While this is currently the most common method for calculating carbon sequestration by wetlands (Chmura, Anisfeld, Cahoon, & Lynch, 2003; Macreadie et al., 2017; McCleod et al., 2011), the carbon in these surface layers is still potentially vulnerable to microbial attack and decomposition, meaning this is a potential overestimate compared to long-term rate of carbon accumulation (Bao et al., 2011).

In order to estimate sequestration rate at sites we did not select for ^{210}Pb age dating, we used sediment accretion rates (mm/year) from wetlands in Victoria and wetlands nearby (New South Wales and South Australia) in the published literature (Table S2). These previous studies mostly calculated sediment accretion rates in wetlands (with reviews of this by Gell et al. (2009) and Gell and Reid (2014)). These studies used various age dating methods, that included ^{210}Pb, ^{137}Cs, ^{14}C, and pollen. Here, core lengths ranged from 10 to 200 cm, with ages ranging between 40 and 200 years, representing recent rates of carbon accumulation or RERCA (Bao et al., 2011). We combined this sediment accretion data with data collected in this study on average DBD and percent carbon from the same wetland or from an average of the three closest wetlands of the same type to assign a value for sequestration for each wetland (Table S2).

2.5 | Statistical analysis

Carbon stocks present within the sampled soil profiles were transformed ($\log_{10}$(Mg C_{org} ha^{-1})) prior to statistical analysis to meet assumptions of normality and homogeneity of variance. Carbon stocks were compared across ecosystems using a one-way ANOVA, with carbon stock as the response and wetland type as the factor. A post hoc Tukey Pairwise Comparison was performed to distinguish differences in means. A fully nested ANOVA was performed to compare carbon stocks in each ecosystem across the six CMA regions where they co-occurred, using carbon stock as the response and site nested within CMA as factors.

To test for differences between wetland types in carbon sequestration rate, we used an ANCOVA, with wetland type as a fixed factor and wetland size (ha) as a covariate. Tukey HSD was then used to determine differences among wetland types. Statistical analyses were performed with IBM SPSS Statistics version 24.0 for Windows (SPSS Inc.).

2.6 | Emissions from previous, current, and future ecosystem loss

Carbon emissions from wetland loss were calculated based on carbon stock estimates from this study in combination with estimates of resultant carbon emission due to disturbance and data on pre-European (pre-1778s) distribution of wetlands (DELWP 2016). The area of ecosystem loss from each CMA region was calculated by comparing pre-European and current wetland extents. Next, average carbon stocks for each wetland type, within each CMA region, were multiplied by the area of wetland loss, and then by a demineralization rate. Where no sediment cores were collected for a particular wetland type within a CMA region, the average carbon stock value for the wetland type in the CMA was based on the average of that wetland type in all of Victoria.

Upper, lower, and intermediate estimates of carbon demineralization were calculated based on previous research. Murray, Pendleton, Jenkins, and Sifleet (2011) estimated that 90% of the C_{org} in the top meter of sediment was released as CO_2 emissions, while Donato et al. (2011) estimated that 50% of the top 30 cm and 25% from the remainder of the top meter of sediment would be converted into CO_2. Siikamäki, Sanchirico, Jardine, McLaughlin, and Morris (2013) combined these estimates to calculate a high (90%), low (27.25%), and intermediate (58.625%) estimate of the carbon stock lost as CO_2 equivalents (eq.). These percentages defined by Siikamäki et al. (2013) were used here to estimate the demineralization of C_{org} from the surface to the depth that was sampled in this study, converted from C_{org} to CO_2 eq. by multiplying by 3.67.

3 | RESULTS

3.1 | Carbon density, DBD, and percent carbon

The percentage C_{org} values across all samples ranged from 0% to 56%, with an average of 7.7 ± 0.3%. The organic carbon density of all samples ranged from 0 to 260 mg C_{org} cm^{-3}, with a mean ± SEM of 31 ± 1 mg C_{org} cm^{-3}. Percent organic carbon was generally highest at the surface (0–2 cm and 14–16 cm), with a 25% decline by 28–30 cm and a 50% decline by 48–50 cm (Figure S1a). However, there were a number of sites that did not follow this trend. The DBD of all samples ranged from 0.03 to 2.4 g/cm^3, with an average of 1.00 ± 0.02 g/cm^3 (Figure S1b).

Using the carbon stock data collected in this study, carbon stocks associated with wetlands were calculated within each region and then added together to calculate total wetland carbon stock across Victoria (Table 1). We determined that permanent open freshwater wetlands are responsible for 18% of Victoria's wetland carbon stock, mostly due to their extensive area in Victoria (about 30% of total wetland area). Freshwater meadows store a further 16% of Victoria's nontidal wetland soil carbon stocks, while saline wetlands account for only 9% of soil carbon stocks despite accounting for almost 25% of the state's wetlands by area. Conversely, despite their high soil carbon stocks, due to their limited distribution (0.73% of total nontidal wetland area), alpine wetlands store just 1.14% of nontidal wetland soil carbon stocks in Victoria.

3.2 | Wetland comparisons

When comparing soil carbon stocks across the five freshwater wetland types most represented, permanent open freshwater wetlands had significantly lower carbon stocks (110 ± 120 Mg C_{org} ha^{-1}) compared to shallow freshwater marsh (200 ± 200 Mg C_{org} ha^{-1}), deep freshwater marsh (230 ± 190 Mg C_{org} ha$^{-1)}$, and alpine wetlands (290 ± 180 Mg C_{org} ha^{-1}) (ANOVA, $F_{2,857}$ = 140.20, p < 0.001; Table 1, Figure 2). Freshwater meadows also had lower carbon stocks (130 ± 100 Mg C_{org} ha$^{-1)}$ when compared to deep freshwater marsh and alpine wetlands (Figure 2).

TABLE 1 Carbon stock estimates for Victoria

Wetland type	Number of cores	Average of Mg C_{org} ha^{-1} ± *SD*	Average depth (cm) ± *SD*	Total Area (Hectares)	Total stocks (Mg C_{org})
Alpine wetlands	36	290 ± 180	75 ± 29	4,476	190,400
Shallow freshwater marsh	57	200 ± 200	61 ± 23	69,061	8,891,000
Saline	21	64 ± 48	63 ± 30	155,719	10,006,000
Freshwater meadow	32	130 ± 100	50 ± 26	144,180	14,200,000
Deep freshwater marsh	100	230 ± 190	72 ± 34	55,790	15,000,000
Permanent open freshwater	67	110 ± 120	62 ± 21	185,035	19,830,000
All wetlands combined	313	186 ± 176	64 ± 28	614,259	68,120,000

While the average carbon stock for each wetland type across all samples is presented, total carbon stocks are calculated within each region and combined to estimate the total stocks. The wetlands are ordered from smallest to largest in estimated total carbon stocks.

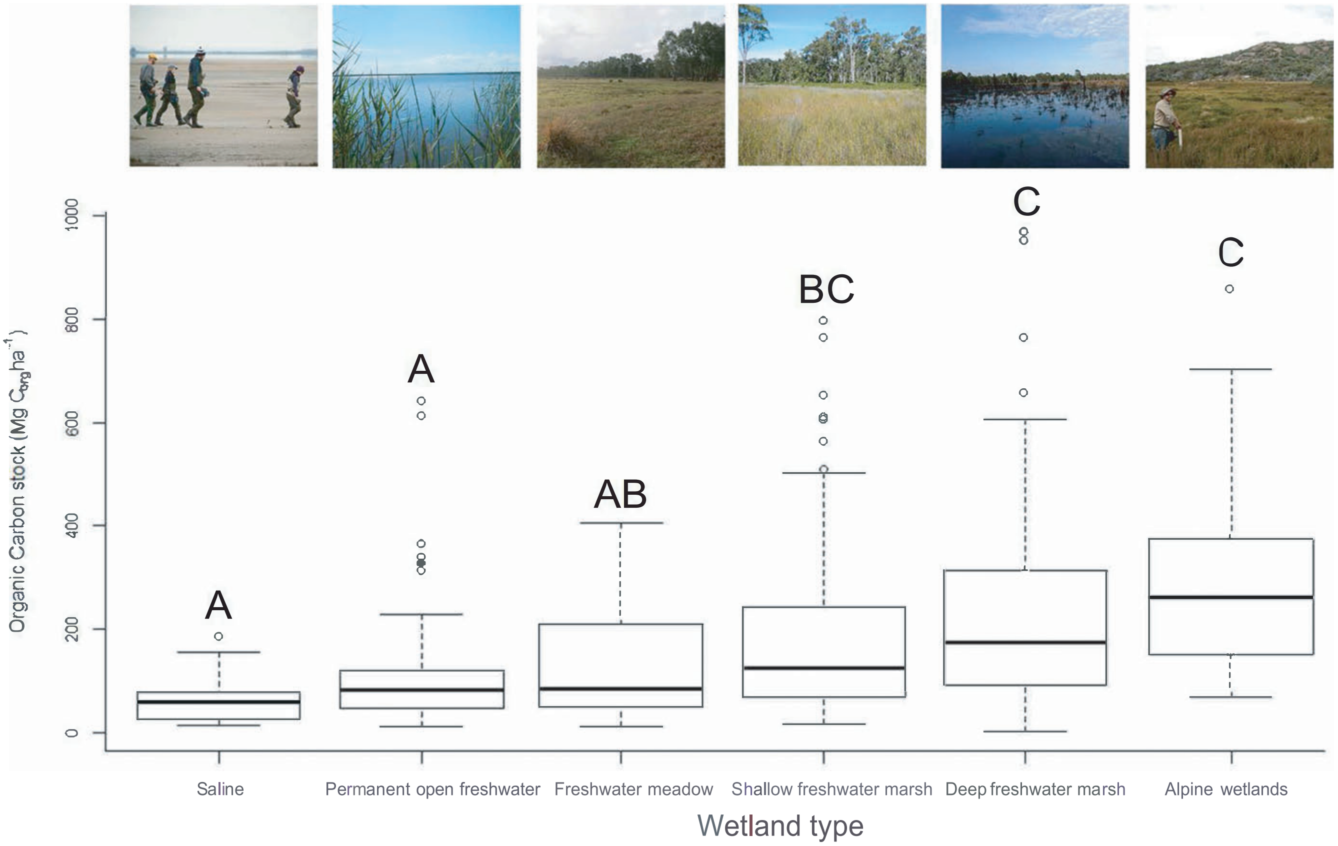

FIGURE 2 C_{org} stock (Mg C_{org} ha^{-1}) to depth at refusal by wetland type. The center horizontal line represents the median and the box and vertical lines represent quartiles. Letters above bars represent Tukey post hoc results (ANOVA, $F_{2,857} = 140.20$, $p < 0.001$)

Carbon stocks varied significantly between catchment regions, but these differences also depended on the wetland type. For example, in deep freshwater marshes, Glenelg Hopkins (GH) had the highest carbon stocks with an average of 376 ± 269 Mg C_{org} ha^{-1} (Figure 3). In contrast, for shallow freshwater marshes, Port Phillip and Western Port had the highest average carbon stocks with an average of 426 ± 194 Mg C_{org} ha^{-1} (Figure 3). In comparison, for alpine wetlands, there was no significant difference among regions. This suggests that the drivers of wetland carbon stock variability (or lack thereof) are different for each wetland type.

So the carbon stock information can more readily be taken up into conservation and management planning, we mapped the catchment region specific wetland carbon stock averages from Figure 3, back onto the wetland maps that exist for Victoria (Figure 4). This provides a quick and easy way to digest the wetland type and spatial variability in the dataset across this broad region. Here, we can see

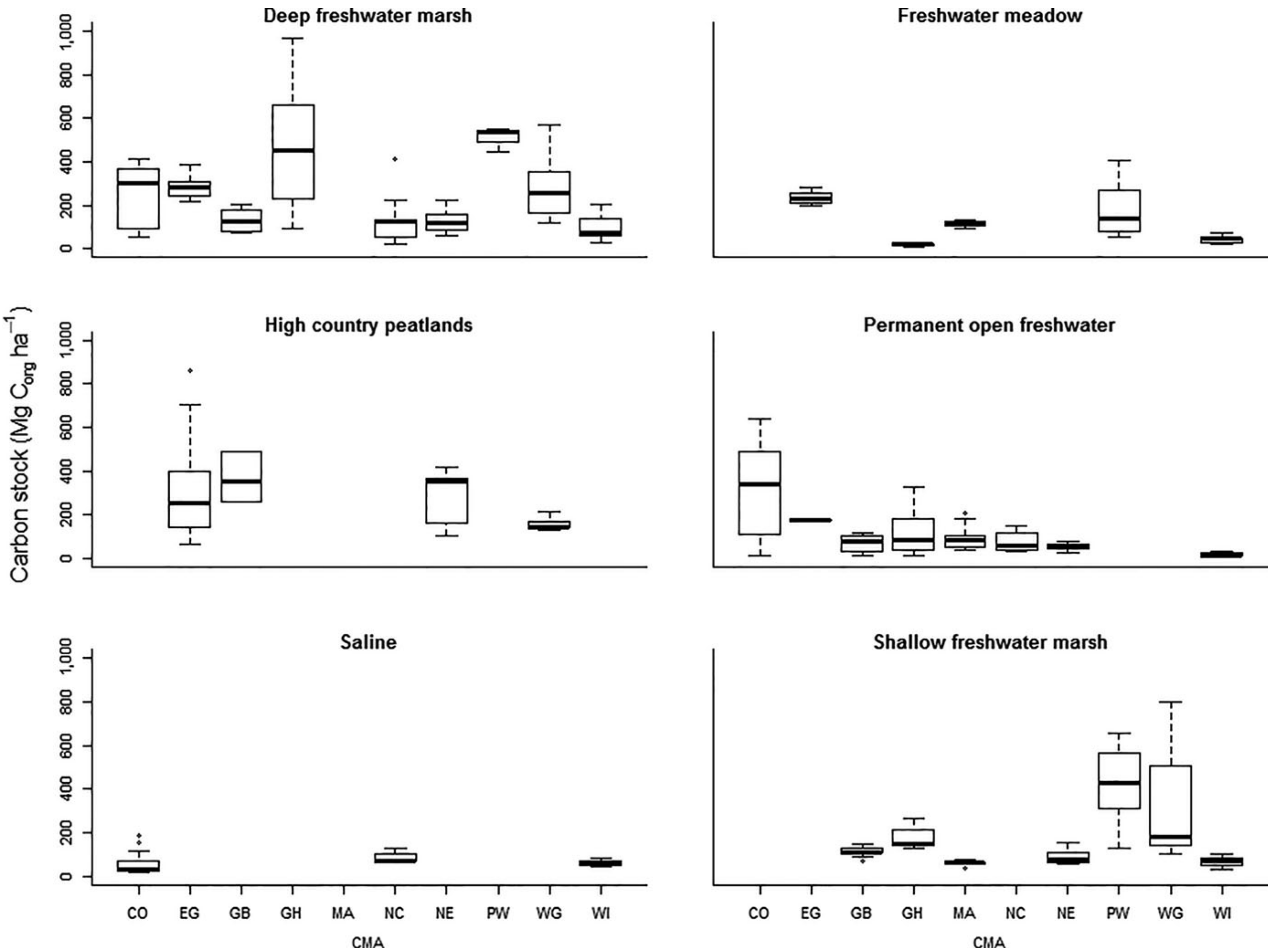

FIGURE 3 C_{org} stock (Mg C_{org} ha^{-1}) to depth at refusal by wetland type. The center horizontal line represents the median and the box and vertical lines represent quartiles. Letters above bars represent Tukey posthoc results

that the regions of GH and Corangamite (CO) have wetlands with high carbon stocks. In particular, the contrast in carbon stocks between different between wetland types that are spatially clustered, is highlighted in these regions. The Gippsland Lakes in West Gippsland (WG) is also an interesting example, as here it suggests that wetland carbon stocks get lower the closer to the coast through this large freshwater lakes system.

3.3 | Carbon sequestration rates

The ^{210}Pb analyses of the cores collected in this study allowed estimating soil accretion rates of four wetlands, that ranged from 0.4 to 1.45 mm/year[1] (Table S2). For the core at Ewing's Morass in East Gippsland CMA region, no pattern of decreasing concentrations of ^{210}Pb with depth was observed, indicating that this sediment core was mixed, at least in the upper 11 cm, and thus it was not possible to estimate a sediment accumulation rate. The concentrations of ^{210}Pb in the cores at Seaford wetlands in Port Phillip and Westernport and McCullum swamp in GH were constant in the upper 3 and 4 cm, respectively, which is interpreted as evidence of mixing. Below these depths the concentrations of ^{210}Pb decreased steadily, allowing estimates of the average sedimentation rates for the last decades to be derived using the CF:CS model. The CRS model was also applied to these cores, obtaining comparable results: 1.45 ± 0.06 mm/year for Seaford wetlands and 0.57 ± 0.03 mm/year for McCullum swamp. No mixing was apparent in the core collected from Doctors Swamp in Goulburn Broken CMA region, and both the CF:CS and the CRS models yielded equivalent sedimentation rates (0.91 ± 0.05 mm/year). The upper 3 cm of core Redgum swamp in the Wimmera CMA consisted of dense vegetation, but the ^{210}Pb concentration profile also allowed estimating an average sedimentation rate of 0.40 ± 0.05 mm/year. These results add to the sediment accretion rates compiled from a number of published studies in southeast Australian wetlands (Table S2). Overall, soil accretion rates ranged from 0.4 to 30 mm/year, with a mean ± *SEM* of 6.7 ± 1.3 mm/year. Carbon sequestration rates can be calculated by multiplying sediment accretion rate by the average carbon density over the depth sampled. Using carbon density data from our carbon measurements, average carbon sequestration rates were 1.9 ± 0.4 Mg C_{org} ha^{-1} $year^{-1}$ or 6.9 ± 1.4 Mg CO_2 eq. ha^{-1} $year^{-1}$ (Figure 5).

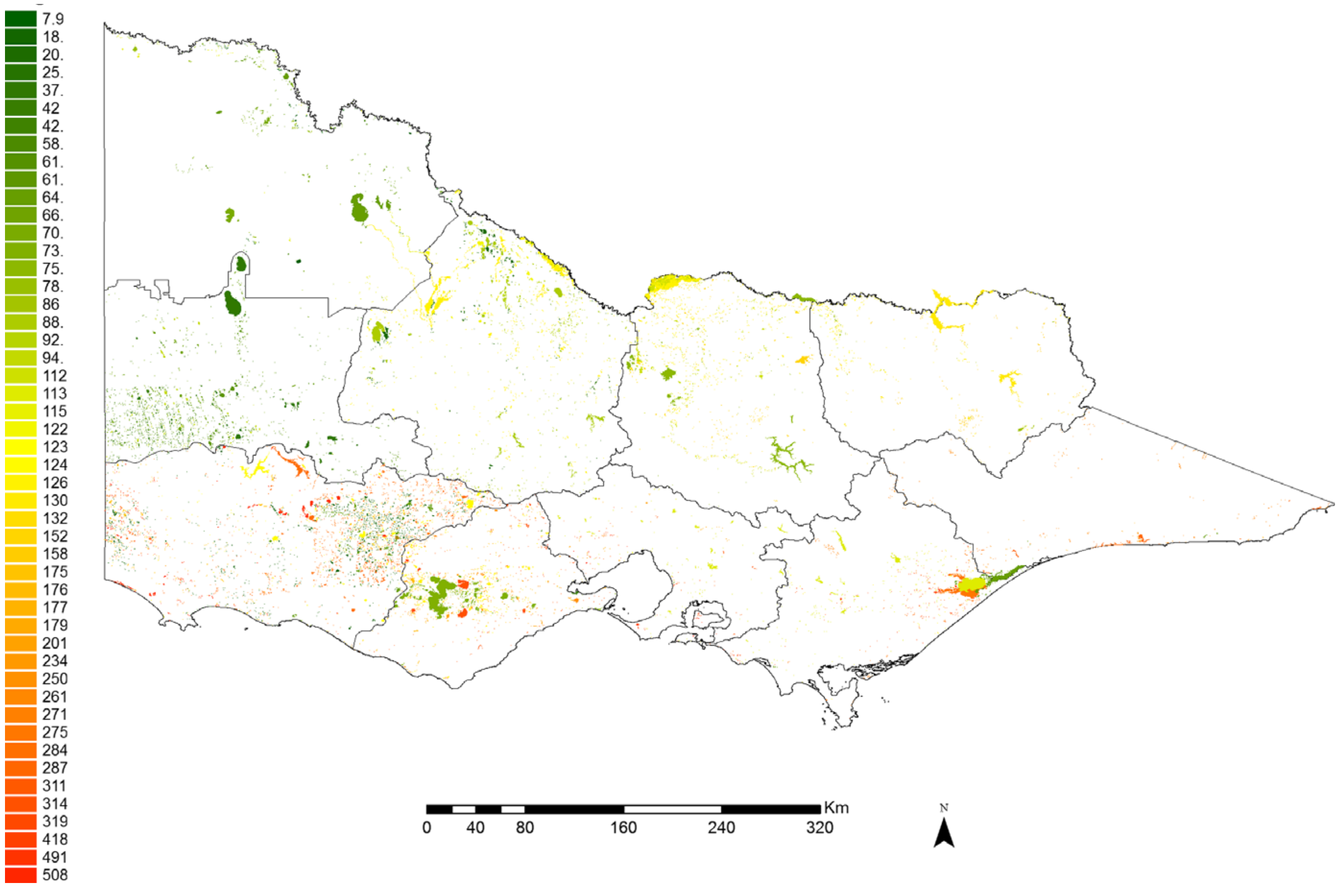

FIGURE 4 Average soil organic carbon stock (Mg C_{org} ha^{-1}) from the wetland types sampled, by each catchment region across the state of Victoria. Where we did not have data for a wetland type in a catchment region, we used the state average value instead. The ten catchment regions are displayed: MA = Mallee, NC = North Central, GB = Goulburn Broken, NE = North East, EG = East Gippsland, WG = West Gippsland, PPW = Port Phillip & Westernport, CO = Corangamite, GH = Glenelg Hopkins

When the carbon sequestration rate data collected from this project is combined with data from the literature (Table S2), results from the ANCOVA showed a significant difference between wetland types (ANOVA, $F_{2,26} = 4.104$, $p = 0.028$): Post hoc pairwise comparisons show that permanent open freshwater sites sequestered on average 2.5 times more carbon per year (2.3 ± 0.7 Mg C_{org} ha^{-1} $year^{-1}$) than shallow freshwater marshes (0.91 ± 0.27 Mg C_{org} ha^{-1} $year^{-1}$) ($p = 0.041$). The carbon sequestration rate of deep freshwater marsh's (1.6 ± 0.5 Mg C_{org} ha^{-1} $year^{-1}$) was not significantly different to permanent open freshwater or shallow freshwater marshes.

To demonstrate the capacity of these wetlands to store carbon and put this data in a global context, we can estimate the potential carbon sequestration of wetlands in Victoria. Using the average carbon sequestration value for freshwater wetlands (6.9 ± 1.4 Mg CO_2 eq. ha^{-1} $year^{-1}$), along with the total area of freshwater wetlands in Victoria (458,500 ha), the estimated carbon sequestration of Victoria's freshwater wetlands is 3,200,000 ± 630,000 Mg CO_2 equivalents per year. This has a potential value of \$AUD 38 ± 7.4 million per year (at \$11.83 Mg CO_2e, the weighted average carbon price from the last five Australian Emission Reduction Fund Auctions), and is equivalent to the annual CO_2 emissions of about 200,000 Australians. This estimate, however, does not include saline nontidal wetlands, as there was no data available on sediment accretion in these systems.

3.4 | Estimated emissions from wetland losses since European settlement

Based on our carbon stock measurements, wetland loss since the time of European settlement in Victoria (~1,834) has resulted in carbon emissions between 22.5 and 74.2 million Mg CO_2 equivalent (Table 2). More than 40% of carbon stocks in these lost ecosystems were associated with permanent open freshwater wetland loss, while deep freshwater marshes and shallow freshwater marsh account for just slightly more than 20% of carbon stocks each. The largest estimated carbon stock losses from deep freshwater marshes would occurred in WG CMA, with substantial losses of permanent open freshwater in CO CMA, totaling to almost 40% of the emissions resulting from wetland loss.

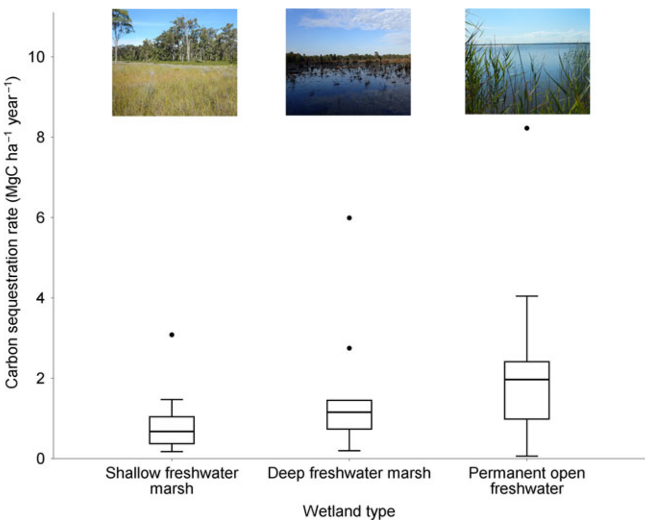

FIGURE 5 Carbon sequestration rate (Mg C ha^{-1} y^{-1}) of wetland types included in this study and from published data. Shallow freshwater marsh (n = 10), deep freshwater marsh (n = 9), and permanent open freshwater (n = 11). The center horizontal line represents the median and the box and vertical lines represent quartiles. Letters above bars represent Tukey post hoc results (ANCOVA, $F_{2,26}$ = 4.104, p = 0.028)

4 | DISCUSSION

4.1 | Wetland carbon stocks

This study represents the most comprehensive regional assessment of soil carbon stocks and sequestration rates of nontidal wetlands in Australia to date. Victoria's wetlands are storing substantial amounts of carbon in the soil, with a total estimated carbon stock for Victoria of 68 ± 44 million tons of C_{org} in the upper meter. By converting the estimated carbon stocks for Victoria to CO_2 equivalents and using an Australian Government's 2013–14 carbon price of $AUD 11.83/ton CO_2e (the weighted average carbon price from the last five Australian Emission Reduction Fund Auctions) Victoria's carbon stocks would be valued at $AUD 2.9 billion. However, this value is only realized if all the wetlands are at risk of disturbance or loss. The value of the carbon stocks has been calculated here to help place the value of these wetlands into a broader context. In addition, while this could be an underestimate as not all sites could be sampled to the full 1 m depth, it does encompasses the highest organic carbon layer and therefore most of the organic carbon stock.

The Victorian wetlands sampled show a wide range of C_{org} contents, from almost negligible amounts in some permanent open wetlands to more than 50% in alpine wetlands, resulting in carbon stocks ranging from 4 to 1,000 Mg C_{org} ha^{-1}. It is likely that the diversity of climatic regions (semi-arid, temperate, and alpine), which influences rainfall and temperature contributes to this substantial variation (Figures 3 and 4). The differing contribution of autochthonous vs. allochthonous carbon inputs is also likely to vary spatially because of differences in surrounding land use and the degree of water regulation in the catchment. This highlights the importance of both broad and fine spatial-scale data in understanding differences in carbon stocks between wetland types and to assist in estimating regional wetland carbon stocks.

While wetlands overall did show regional variation in carbon stocks (4–1,000), there were also differences between some wetland types. Alpine wetlands, deep freshwater marshes, and shallow freshwater marshes had higher soil carbon stocks than saline wetlands and permanent open freshwater wetlands. While permanent open freshwater wetlands can be sinks for catchment carbon, a recent study by Villa and Bernal (2017) suggests a quadratic relationship between inundation time and organic carbon stock and sequestration rates. This could also be because permanent open freshwater wetlands tend to have less macrophytes or tress compared to marshes or swamps, and therefore less primary productivity and addition of carbon from within the wetland itself.

We found the highest average carbon stock values in alpine wetlands, highlighting the importance of their protection. Grover, Baldock, and Jacobsen (2012) similarly found high carbon stocks (496 Mg C_{org} ha^{-1}) in alpine wetlands in Victoria, while boreal and arctic peatlands are known globally for storing substantial amounts of carbon (Kayranli, Scholz, Mustafa, & Hedmark, 2010; Yu, 2012). However, while alpine wetlands are the smallest in total wetland

TABLE 2 Estimated CO_2 emissions from wetland loss since European settlement of Victoria (~1,834)

			CO_2 equivalent loss		
Sites	Area lost (Ha)	Stock Mg C_{org} Ha^{-1} Avg.	27%	59%	90%
Alpine	Unknown	290 ± 180	NA	NA	NA
Saline	5,872	64 ± 48	381,500	820,800	1,260,000
Deep freshwater marsh	13,312	230 ± 190	5,112,000	11,000,000	16,884,000
Freshwater meadow	24,749	130 ± 100	2,163,000	4,653,000	7,143,000
Shallow freshwater marsh	31,595	200 ± 200	5,059,000	10,890,000	16,709,000
Permanent open freshwater	71,525	110 ± 120	9,752,000	20,980,000	32,209,000
All wetlands combined	260,530		22,467,500	48,340,000	74,205,000

Estimates of wetland area loss come from DELWP 2016. While wetland area lost, and average carbon stocks for each wetland type are presented, CO_2 equivalent loss is calculated based off the regional carbon stocks for each wetland type. C_{org} is converted to CO_2 by multiplying by 3.67. Wetland types are ordered from least to most area of wetland loss.

area in Australia, they are one of the most vulnerable to loss due to climate change (Williams & Wahren, 2005; Xue et al., 2016). Moreover, alpine wetlands are vulnerable to grazing and trampling disturbance by introduced herbivores such as horses and deer (Wahren, Williams, & Papst, 2001).

4.2 | Comparison of soil carbon sequestration by wetland types

Soil carbon sequestration rates were higher in permanent open freshwater wetlands compared to shallow freshwater marshes. This is opposite to what was observed in carbon stocks for these wetlands types, where permanent open freshwater wetlands had the lowest carbon stocks. This higher rate of carbon sequestration is linked to the higher rate of sediment accretion at these sites compared to the shallow freshwater marshes. This can occur because sites with high organic carbon soils (high carbon stocks), may accumulate at slow rates, such as alpine bogs (Grover et al., 2012)) or some tidal marsh systems (Ewers Lewis, Carnell, Sanderman, Baldock, & Macreadie, 2018). In comparison, sites which are accumulating sediment rapidly, while high in percent organic carbon, may have low bulk density, and potentially low carbon stocks when compared over similar depth ranges. This may also suggest that while accumulation of carbon is faster in permanent open freshwater systems over the time-scale of hundreds of years, they experience higher rates of organic matter decomposition over longer time scales.

The highest sediment accretion values were associated with wetlands located along major rivers (or closer to fluvial inputs) (Table S2). In south-eastern Australia, there has been a substantial increase in sediment accretion rate in floodplain billabongs since European settlement (Gell et al., 2009). This suggests that the high rates of sedimentation in these wetlands, and potentially carbon sequestration, has been anthropogenically increased due to clearing of terrestrial vegetation in the catchment and changes in river hydrology. Additional research investigating the allochthonous vs. autochthonous carbon contributions to sequestration rates in these wetlands is required to understand this dynamic.

4.3 | Potential emissions from previous wetland loss

We estimate that degradation and loss of wetlands since European settlement in Victoria (from 1,834) has resulted in the emission of 22–74 million Mg CO_2 equivalents. Since there is no available data to construct a timeline of wetland loss between 1,834 and now, it is difficult to put this loss into perspective, but the loss over this ~170 year period is comparable to the emissions of 1–4 million Australians or Americans (who release on average 16 Mg CO_2e year^{-1}) over 1 year. These emissions from wetland loss in Victoria, are similar to those estimated to have been lost from the loss of Mexican or Malaysian mangroves (~0.025 Pg CO_2 equivalents), which are some of the most carbon-rich soils in the world (Alongi, 2012; Murdiyarso et al., 2015).

One way to guard against continuing carbon losses is by protecting wetlands that may be used for agricultural activities. The impact of agricultural practices in wetlands is stark. Degradation by agriculture, either through wetland drainage, or direct disturbance from grazing herbivores or cropping machinery, can promote soil moisture fluctuations, which then stimulates decomposition and re-release of already stored carbon (Sigua et al., 2009). It is here that wetland rehabilitation works may help to increase wetlands' carbon sequestration capacity (Meyer et al., 2008; Ballantine & Schneider, 2009). In the study of rehabilitated wetlands in the Canadian prairie pothole region, Badiou, McDougal, Pennock, and Clark (2011) estimated soil organic carbon stocks to 30 cm of newly re-habilitated (1–3 years) and long-term rehabilitated wetlands (5–12 years) to be 60% (121 Mg C_{org} ha^{-1}) and 80% (165 Mg C_{org} ha^{-1}) of reference wetlands (205 Mg C_{org} ha^{-1}), respectively.

With guidance provided by IPCC (2014), the pathway forward for wetland inclusion into Greenhouse Gas Inventories is much clearer. The potential high emissions associated with wetland loss, highlights the need to ensure sufficient protection exists if ongoing carbon emissions from wetland degradation is to be reduced or avoided. In addition, a loss of wetlands will also reduce their future carbon sequestration potential. As a result, implementing appropriate management strategies that maintain current wetland carbon stocks and future sequestration potentials could offer mechanisms to mitigate greenhouse gas emissions when compared to current business as usual scenarios. However, appropriate and recognized methodologies for measurement will be required to allow the avoided emission and future sequestration to be assessed. If governments are to work towards reducing net carbon emissions, protection and rehabilitation of wetlands might be an important tool. However, their capacity to act as net carbon sinks will not be fully realized until emissions (namely methane and nitrous oxide) are quantified.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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