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RESEARCH

Vegetation carbon in New Zealand wetland ecosystems

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Abstract: Wetlands are a critical, though vulnerable, global carbon store, but the carbon stored in vegetation has not been quantified at a national scale for New Zealand wetlands. We undertook a literature review to assess vegetation carbon density in wetlands and used meta-analysis to estimate means and uncertainty for both vegetation structural classes and wetland type. We then used a combination of derived vegetation carbon densities alongside spatial extrapolation to estimate the amount of carbon stored in wetland vegetation in New Zealand. Our area of interest was the "Wetland - Vegetated non forest" class mapped by the Land Use Map (LUM), which is the authoritative layer for New Zealand carbon accounting. Within our area of interest, we used weighted aggregated land cover classes signifying vegetation to calculate a grand mean density (C ha^{-1}) for New Zealand wetlands mapped by the LUM in 2008, 2012, and 2016, and total vegetation carbon stocks for each of those years. We found that among wetland vegetation structural classes, total above-ground carbon varied from a mean of 4.8 Mg C ha^{-1} for sedgelands to 23.8 Mg C ha^{-1} for tall mangroves. For wetland types, total above-ground carbon varied from a group mean of 3.6 Mg C ha^{-1} for pakihi to 19.5 Mg C ha^{-1} for mangroves. Estimates of below-ground biomass were uncommon (n = ten estimates from four studies) but the limited data available suggest that below-ground carbon density was less than total above-ground carbon density for three herbaceous structural classes (below-ground divided by above-ground ranged from 0.70 for reedland to 0.86 for rushland), while for mangroves below-ground density varied: below-ground divided by total above-ground was 0.99 for tall mangroves and 2.09 for dwarf mangroves. We estimated a total carbon stock of 5 868 710 (1 467 178–10 270 243) Mg C for our area of interest, based on a density of 27.080 (6.68–47.48) Mg C ha^{-1}, and an area of 216 717 ha in 2016, the most recent estimate available. Using selected literature, we confirmed our estimates were broadly consistent with international values. There is a clear information gap regarding carbon densities in forested wetlands in New Zealand, and little ability to assess vegetation by wetland type due to the lack of spatial data around structural classes in New Zealand. We set out some recommendations to address this and other data gaps.

Keywords: bogs; carbon density; carbon stocks; climate change mitigation; fens; mangroves; swamps

Introduction

Wetlands, and particularly wetlands on a peat substrate, are known for climate regulation through carbon storage. Soil carbon storage is a major contributor, driven by wet anaerobic conditions that limit decomposition. Key peat-forming species in New Zealand include *Sphagnum* and Restionaceae species (Clarkson et al. 2017). While wetland plants build soil carbon, thus contributing to climate regulation (Millennium Ecosystem Assessment 2005; Clarkson et al. 2013; Junk et al. 2013), they also potentially contain substantial carbon themselves that merits systematic assessment. However, few studies to date have systematically assessed carbon stocks within wetland vegetation.

Wetlands are broadly defined by the periodic inundation of their substrate, as per Section 404 of the Clean Water Act (United States Congress, 1972) and the Ramsar Convention (1971), and can support a wide variety of vegetation structures, from turf vegetation to forest, including mangrove. As such, vegetation biomass is expected to vary widely across and within wetland types. An improved understanding of vegetation carbon stocks in wetlands will increase knowledge of the total contribution of wetlands to climate regulation services and will improve carbon emissions reporting under the Paris Agreement 2015.

Under the Paris Agreement, New Zealand is required to report on carbon emissions and sinks resulting from land management and land use change. Of the native ecosystems remaining, the most common are forests and scrublands (7 014 534 ha and 1 842 224 ha respectively), tussocklands (2 335 410 ha), lightly vegetated areas such as landslides, alpine herbfields, and gravel and rock areas (1 281 596 ha), and herbaceous vegetation (161 444 ha) (Land, Air, Water Aotearoa 2024; Stats NZ 2024). Current wetland extent, regardless of vegetation cover, is 249 214 ha (Dymond et al. 2021). While there is a comprehensive, spatially representative,

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unbiased network of repeated-measures plots to monitor and report on changes in forest carbon (Holdaway et al. 2017; Paul et al. 2021), New Zealand lacks comparable monitoring systems for other natural ecosystems. Because of a lack of carbon density and ecosystem extent data for wetlands, until 2023 when our findings were reported to the Ministry for the Environment, national inventory submissions (i.e. reports) could not estimate the carbon stock and carbon stock change in vegetated wetlands, which meant any emissions and removals for land transitioning into and out of this class were also not estimated. A United Nations Framework Convention on Climate Change review of New Zealand's reporting recommended ameliorating this situation. National estimates of vegetation carbon for New Zealand were compiled in Tate et al. (1997). Of the 47 vegetation types for which biomass was estimated, only 12 types were supported by values derived from research literature. Wetland communities were aggregated to two coarse types ("Wetland communities" and "Pakihi heathland communities") and biomass estimates were based on "author estimate/interpolation" (see Table 2 in Tate et al. 1997). Subsequent carbon research has primarily focussed on improving estimates of forested carbon stocks in New Zealand (Mason et al. 2012; Holdaway et al. 2017; Paul et al. 2021).

A critical driver of variation in vegetation biomass (and therefore carbon) is vegetation structure (Catchpole & Wheeler 1992). As such, it follows that estimates of wetland vegetation carbon should be weighted by structural class abundance where possible. International guidance permits country-specific methodologies for carbon accounting in wetlands where this retains consistency and comparability between countries (Hiraishi et al. 2014). In this context, New Zealand provides an instructive case study, with published estimates of vegetation biomass (and therefore, vegetation carbon) from a range of wetland ecosystems, described such that structural class can be inferred, and spatial layers of land cover and land use available to scale up to a national estimate of wetland vegetation for carbon. We sought to quantify vegetation carbon within the relevant area. The New Zealand Land Use and Accounting System (LUCAS) Land Use Map (LUM) (Ministry for the Environment 2012) defines the area for which New Zealand reports carbon values for wetlands using two classes, only one of which is vegetated: "Wetland - Vegetated non-forest". The area covered by the most recent estimate of the LUM class "Wetland - Vegetated non-forest" is 216 717 ha, and as such is c. 13% less than the estimated total wetland extent in New Zealand noted above. We sought to quantify vegetation carbon:

(1) For all reported structural classes of wetland vegetation and wetland types (per Johnson & Gerbeaux 2004) reported in New Zealand, using a literature review and meta-analysis;

(2) For additional structural classes, inferred from land cover types mapped within the LUM class "Wetland - Vegetated non-forest", which had not been found in our literature review in (1);

(3) At the national scale, for the entire "Wetland - Vegetated non-forest" area, by using the values derived from (1) and (2) to calculate wetland vegetation carbon stocks, using the three most recent LUM mapping years (2008, 2012, 2016).

This approach has been adopted into New Zealand's Greenhouse Gas Inventory (Ministry for the Environment 2024).

Methods

Overview of methodological approach

The LUM (Ministry for the Environment 2012) defines the area in which New Zealand might report carbon values for wetlands using two classes: "Wetland - Vegetated non-forest" and "Wetland - Open water". Because New Zealand does not distinguish wetland forests from non-wetland forests within its natural forest class, the only vegetated wetlands class relevant for reporting under the Paris Agreement is the LUM class "Wetland - Vegetated non forest". New Zealand also has a more detailed land cover layer captured within the New Zealand Land Cover Database (LCDB) (MWLR 2020) that can be used to disaggregate the broad range of vegetation that is expected to be present within the LUM class "Wetland - Vegetated non forest".

We reviewed literature on vegetation carbon densities, assigned reported densities to vegetation structural classes following Atkinson (1985), and undertook a meta-analysis with the resulting data. We supplemented the literature review and meta-analysis with a targeted data search, which we describe further below. This was in part necessary because the area mapped as LUM class "Wetland - Vegetated non-forest" includes LCDB forest classes, both native and exotic. The meta-analysis results were stratified by structural class and then aggregated to land cover classes using LCDB. We then aggregated the land cover classes to a grand mean (mean weighted by the area of each land cover class that fell within the LUM class "Wetland - Vegetated non-forest"). We calculated uncertainty using quadrature (discussed below) weighted by the area of each land cover class. We understand New Zealand's carbon accounting system currently takes just one weighted mean of carbon density. As such, the weightings and grand means were calculated for the three mapping years in two ways: once where all three mapping years used the relative proportions of land cover class areas in 2012 for weights, and a second time where each mapping year used the relative proportions of land cover class area for its mapping year for weights. Where one value is used, we use the 2012 mapping year, as this was the period where the LCDB and LUM estimate dates aligned most closely: for other mapping years, the estimate dates (or mapping period) did not quite align. All numbers are reported to two decimal places, except in the case of density estimates which we have reported to three decimal places for greater accuracy given the large sizes of the areas concerned.

Literature review of wetland vegetation carbon stocks

We used the following search tools to conduct our literature review: Web of Science, a New Zealand database of theses from all tertiary academic institutions (nzresearch.org.nz), and the Manaaki Whenua - Landcare Research library catalogue of unpublished reports. These were complemented with Google Scholar searches to discover articles that had cited key papers identified in the previous step. We searched for any studies that contained both the terms "wetland" and "New Zealand" in conjunction with either of the terms "biomass" or "carbon". Search outputs were screened to identify and classify publications relevant to our objective of quantifying carbon in wetland vegetation. We first screened by publication title, and then by abstract and methods. Our focus was vegetation biomass carbon, so we discarded studies on soil carbon, soluble organic carbon, litter decomposition, wastewater treatment, methane fluxes, or biomass/carbon of individual plants (although we retained those that sampled vegetation stands or vegetation per unit area). We did not limit our search by vegetation structural class or exotic/native status.

We extracted key information from relevant studies: location, wetland type, which biomass pools were reported, the vegetation carbon stock, and information on sampling methods. Where biomass estimates were only presented graphically, we used Plot Digitiser 2.6.6 (Huwaldt 2014) to extract the values plotted either as means and uncertainties (two studies) or as individual measurements (one study).

For each study we compiled reported estimates of live, dead, and total above-ground biomass pools and included below-ground pools where available. Total above-ground biomass pools can include both live and dead plant material (reported in some sources) as well as coarse deadwood in woody vegetation; soil litter was generally not reported in the literature so was not included in our assessment. Soil carbon and historical buried deadwood (e.g. swamp kauri) were out-of-scope for this exercise. In the few instances of vegetation dominated by deciduous species (as in some rhizomatous macrophytes) where biomass values were reported at different times in a year, we computed and relied on the inter-seasonal mean. For each study, we compiled the reported sources of errors (either standard deviations or standard errors). In some cases, studies lack data to allow us to use them fully. One study reported fresh plant biomass only and was used to match wetland structural classes to land cover classes but was otherwise excluded. Other studies reported only live above-ground biomass or did not state whether dead plant material was accounted for; these were excluded from the meta-analysis but were used as a supplement where data were entirely lacking (e.g. Bassett et al. 2010; Burge et al. 2020). Most studies did not involve experimental manipulations but for the few that were experimental studies (Clarkson et al. 2015; Burge et al. 2020), we used the biomass reported from untreated control plots.

Meta-analysis

Data preparation

First, we standardised wetland types and structural classes of vegetation following definitions from Johnson and Gerbeaux (2004), who adopt the structural class definitions of Atkinson (1985). Second, we transformed biomass density estimates into standard carbon units (Mg C ha^{-1}). We did this by assuming a carbon content in dry plant biomass of 44% for herbaceous species and 42% for mangroves. The 44% content was based on approximated 0.2-0.5-0.3 stem-leaf-root biomass fractions for herbaceous plants (Poorter et al. 2012) and a 0.424-0.447-0.425 stem-leaf-root carbon concentration for herbaceous plants (Ma et al. 2018). For mangroves we relied on the 42% carbon concentration determined for *Avicennia marina* wood (Bulmer et al. 2016a). Third, we converted any measures of uncertainty reported as standard errors into standard deviations. Fourth, we used the location data of study sites to determine their LCDB v.5 class as of 2012. Study locations are shown, with LCDB v.5 land cover class and assigned structural class (Appendix S1 in Supplementary Material).

Analysis

We used a random-effects meta-analysis to summarise the pooled mean carbon densities for groups, where our groups were vegetation structural class and wetland type. The method combines and weights the estimates from individual studies according to their precision (individual means are weighted by the inverse of associated variances) and accounts for the within- and between-study variances. A random-effects meta-analysis accounts for two variance components (within- and between-study variances) by allowing the means from different studies to differ from one another as a random sample from a population of outcomes (Gurevitch et al. 2018; Harrer et al. 2021).

We applied the meta-analysis for each group for both above-ground and below-ground carbon. The meta-analyses results are presented via standard forest diagrams (Gurevitch et al. 2018; Harrer et al. 2021) and account for the variation between source studies. Two measures of between-study heterogeneity are presented: τ, an estimate of the standard deviation between individual means (that excludes within study variances), and the I^2 statistic, a measure of the percentage of variability in effect sizes that is not caused by sampling error. As a rule of thumb, I^2 values of 25%, 50%, and 75% are interpreted as low, moderate, and substantial between-study heterogeneity respectively (Harrer et al. 2021).

At the level of vegetation structural class, we obtained estimated uncertainties from the meta-analysis, where and the corresponding τ^2 provide measures of between-study heterogeneity (Harrer et al. 2021). Specifically, τ is an estimate of the standard deviation of effect sizes, and τ^2 is an estimate of the variance of effect sizes. Most of the compiled studies correspond to single site estimates and so these measures of between-study heterogeneity may be interpreted as capturing some of the spatial heterogeneity in carbon densities. Only two of the herbaceous structural classes (rushlands and mixed rushlands) had sufficient data to allow estimates of τ and thus we assumed equivalent levels of heterogeneity for other herbaceous structural classes and scaled those by the corresponding coefficient of variation (CV, where CV = SD/mean = τ/mean). Comparable approaches that assume equivalent levels of deviation for families of variables that are expected to behave similarly are used in ecosystem modelling (e.g. Håkanson 2003). Structural class means are presented in Table 1.

Table 1. Structural class mean total carbon density (Mg C ha^{-1}). For above-ground total carbon density, estimate uncertainty is presented in brackets. This uncertainty is τ (equivalent of standard error), and where replication did not allow τ to be estimated (for fernland, reedland, and tussockland), it is an estimate of standard error based on the mean for the structural class and the coefficient of variability for the rushlands classes (for details, see Table 2). As per main text, these are considered to be equivalent. Insufficient data meant uncertainty could not be assessed for below-ground carbon. Structural classes that are synonymous with land cover (LCDB) classes can be found in Appendix S2 in Supplementary Material.

Structural class	Above-ground density	Below-ground density
Dwarf mangrove	11.80 (0.88)	24.70
Fernland	6.14 (0.81)	4.27
Mixed rushland	5.36 (0.75)	6.72
Reedland	10.52 (1.38)	8.76
Rushland	7.04 (0.86)	6.03
Sedgeland	4.82 (0.63)	8.20
Tall mangrove	23.81 (0.61)	23.66
Tussockland	27.16 (3.57)	8.35

Scaling up from structural class to land cover classes within the relevant land use class

Scaling up: mean carbon stocks for each land cover class

We assigned structural classes to land cover classes (Fig. 1) based on the LCDB land cover class definitions (MWLR 2020). Some structural classes were not represented in our literature review. Others were not included in wetland-specific datasets and therefore we undertook a targeted literature search. The sources for each structural class, or land cover class where the class was not disaggregated into structural classes, are shown in Table 2. The structural class means and uncertainty are provided in Table 1. There is no map of structural classes in New Zealand, therefore, a non-weighted average of all relevant structural classes was used to derive the estimate for each land cover class. Mangroves are an exception; the relative extent of dwarf and tall mangroves has been assessed for a large region of northern New Zealand (Suyadi et al. 2020) and thus weighted averaging could account for the relative abundance of these two structural classes.

Scaling up: uncertainty around land cover-level estimates

In the case of estimated carbon densities for shrublands and forests, which were already at the LCDB-scale of land cover class (Table 2), we relied on the standard error of estimates as a measure of uncertainty. Since the standard error is an estimate of the standard deviation for the population mean (Gotelli & Ellison 2004) we interpret it to be at the same level as τ (a measure of deviation of the population mean around individual sample means). As noted in the introduction, there is a comprehensive, spatially representative, unbiased network of repeated-measures plots to monitor and report on changes in forest carbon (Holdaway et al. 2017; Paul et al. 2021); these plots are often referred to as "LUCAS plots" or "LUCAS [vegetation type] plots". Given the systematic and wide distribution of LUCAS plots, these estimates are interpreted to partly capture the spatial variability in carbon densities within vegetation covered by these plots (Table 2).

In the case of structural classes, we used quadrature to scale from the relevant structural classes to land cover classes. Quadrature is the square root of the mean of the squares, where the variance for a mean estimated from n independent random variables can simply be estimated from the mean of their variances (Morrison 2021). In all cases except mangroves this was unweighted, as per Equation (1), as we have no data that give the spatial extent of each structural class within land cover classes.

$$\sigma = \sqrt{\frac{\sigma_i^2+\ldots+\sigma_n^2}{n}} \qquad (1)$$

For mangroves, the relative extent of dwarf and tall mangroves has been assessed for a large region of northern New Zealand (Suyadi et al. 2020) and thus the relative abundance of these two structural classes could be accounted for, using weighted quadrature, as per Equation (2).

$$\sigma_w = \sqrt{\frac{w_i\times\sigma_i^2+\ldots+w_n\times\sigma_n^2}{w_i+\cdots+w_n}} \qquad (2)$$

where σ_w is the weighted standard deviation, n is the number of individual observations (i) and w_i are the weights by which the standard deviations are weighted which, in our case, are the relevant land cover areas.

We highlight that uncertainty associated with the land cover class spatial extents (e.g. mapping error and uncertainty) has not been quantified and we were unable to incorporate it into our analyses.

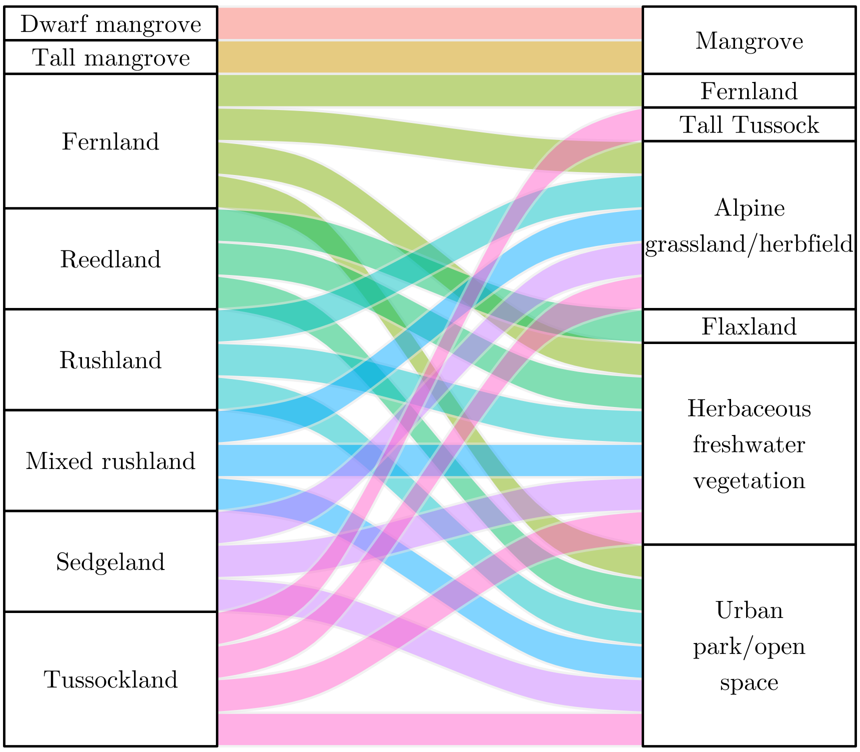

Figure 1. Relationship between structural classes (left side of diagram) and LCDB classes (right side of diagram). On the right side of the diagram, "Tall Tussock" is short for the LCDB class "Tall Tussock Grassland". There were no data available for the flaxland structural class, and therefore reedland was used as the closest proxy value.

Table 2. Sources of mean and measure of uncertainty for each structural class. Land cover classes were derived from an unweighted mean of component structural classes. Structural classes are listed first and contribute to land cover classes as per Figure 1. Land Cover Database classes are only listed where they are not an average of structural classes. The land cover classes not listed were aggregated from structural class using primarily unweighted means of the relevant structural classes (see main text). In multiple classes, the lack of data means uncertainty is assumed to be equivalent with levels of heterogeneity for the rushland and mixed rushland structural classes and scaled by the corresponding coefficient of variation (CV, where CV = SD/mean = τ/mean). In these cases, the source for the uncertainty reads 'Mean CV of rushland & mixed rushland, multiplied by the mean of this class'. LCDB class names are reproduced verbatim to facilitate searching within the LCDB; as such, capitalisation and lack of macrons are retained.

LCDB classes where no structural class	Structural classes	Class name	Mean source	Uncertainty source
	Fernland	Fernland	Meta-analysis	Mean CV of rushland & mixed rushland, multiplied by the mean of this class
	Reedland	Reedland	Meta-analysis	Mean CV of rushland & mixed rushland, multiplied by the mean of this class
	Rushland	Rushland	Meta-analysis	τ from meta-analysis
	Mixed rushland	Mixed rushland	Meta-analysis	τ from meta-analysis
	Tussockland	Tussockland	Literature: Burge et al. (2020) as modified by O'Connor et al. (1999) to account for dead biomass	Mean CV of rushland & mixed rushland, multiplied by the mean of this class
	Flaxland	Flaxland	Flaxlands were assigned the carbon density value for reedlands as this was the nearest match in terms of physiognomy	Mean CV of rushland & mixed rushland, multiplied by the mean of this class
	Dwarf mangrove	Dwarf mangrove	Meta-analysis	τ from meta-analysis
	Tall mangrove	Tall mangrove	Meta-analysis	τ from meta-analysis
High Producing Exotic Grassland		High Producing Exotic Grassland	Literature: Edirisinghe et al. (2012)	Mean CV of rushland & mixed rushland, multiplied by the mean of this class
Low Producing Grassland		Low Producing Grassland	Literature: Hoogendoorn et al. (2016)	Mean CV of rushland & mixed rushland, multiplied by the mean of this class
Short-rotation Cropland		Short-rotation Cropland	IPCC default for annual cropland (IPCC 2006) with a 0.3 root:shoot ratio according to estimates for alfalfa, barley, corn, and wheat (Acock and Allen 1985)	Nominal error from IPCC
Depleted Grassland		Depleted Grassland	Literature: estimate from Norbury et al. (2002)	Mean CV of rushland & mixed rushland, multiplied by the mean of this class
Herbaceous Saline Vegetation		Herbaceous Saline Vegetation	A mean from carbon density values for saltmarsh herbfields (*Sarcocornia* spp., *Sporobolus* spp.) grasslands (*Samolus* spp.) and rushlands (*Isolepis* spp., *Juncus* spp.) mostly from estuaries in southeast Australia (Kelleway et al. 2016; Santini et al. 2019; Owers et al. 2018) with one New Zealand source (*Isolepis* spp. in Bassett et al. 2010)	τ from meta-analysis

Table 2. Continued.

LCDB classes where no structural class	**Structural classes**	**Class name**	**Mean source**	**Uncertainty source**
Manuka and/or Kanuka		Manuka and/or Kanuka	Carbon value was derived from LUCAS plots classified as "mānuka shrubland" (*n* = 5) and measured between 2009–2014 (Holdaway et al. 2017). Above-ground tally comprises shrub biomass, live trees, and coarse woody debris	SE from LUCAS plots (*n* = 5)
Gorse and/or Broom		Gorse and/or Broom	Carbon value derived from LUCAS plots (*n* = 5) classified as "gorse shrubland with cabbage trees" and measured between 2009–2014 (Holdaway et al. 2017)	SE from LUCAS plots (*n* = 5)
Mixed Exotic Shrubland		Mixed Exotic Shrubland	Carbon value derived from LUCAS plots (*n* =3) identified as successional shrublands and with a distinct component of the exotic shrubs *Buddleja davidii*, *Sambucus nigra*, or *Crataegus monogyna* (Holdaway et al. 2017)	SE from LUCAS plots (*n* = 3)
Matagouri or Grey Scrub		Matagouri or Grey Scrub	Carbon value derived from LUCAS plots (*n* = 27) classified as "grey scrub with kanuka" and measured between 2009–2014 (Holdaway et al. 2017)	SE from LUCAS (*n* = 27)
Sub Alpine Shrubland		Sub Alpine Shrubland	Carbon value derived from LUCAS plots (*n* = 3) classified as "Mountain neinei–inanga low forest and subalpine shrubland" and measured between 2009–2014 (Holdaway et al. 2017)	SE from LUCAS (*n* = 3)
Deciduous Hardwoods		Deciduous Hardwoods	LCDB v.5 class corresponds to "exotic deciduous woodlands, predominantly of willows or poplars". In the context of vegetated wetlands, we interpret these as areas invaded by *Salix cinerea*. No carbon estimates were found for *Salix* spp. in New Zealand (or Australia) as per Burrows et al. (2018). Further, *Salix* spp. are absent from LUCAS forest plots reclassified as wetlands. Our carbon estimate (Holdaway et al. 2017) is from a single LUCAS plot (DE64) where both *Salix cinerea* and *Salix fragilis* were present	Calculated by taking CV of mānuka/kānuka, gorse/broom, and mixed exotic shrubland, since *n* = 1 for the LUCAS plot, and then multiplying the mean for deciduous hardwoods by this number
Indigenous Forest		Indigenous Forest	Estimate from 2009–2014 LUCAS sample for beech, broadleaved, and podocarp forest alliances (Paul et al. 2021). Above-ground value comprises live tree biomass and coarse woody debris	SE from LUCAS plots (*n* = 758)

Table 2. Continued.

LCDB classes where no structural class	Structural classes	Class name	Mean source	Uncertainty source
Broadleaved Indigenous Hardwoods		Broadleaved Indigenous Hardwoods	LCDB v.5 class corresponds to "communities of mixed broadleaved short trees indicative of advanced succession toward indigenous forest". Carbon value derived from 2009–2014 LUCAS sample for "māhoe forest", "pepperwood–fuchsia–broadleaf forest" and "silver fern–māhoe forest" alliances in Paul et al. (2021)	SE from LUCAS plots (n = 147)
Exotic Forest		Exotic Forest	Based on carbon density values for pre-1990 and post-1989 planted forests (Paul et al. 2021) weighted by the corresponding pre-1990 and post-1989 planted forest areas (Ministry for the Environment 2021). Applies a 0.2 root:shoot ratio for radiata pine (Beets et al. 2012)	Area-weighted mean SE based on Paul et al. 2021
Forest - Harvested		Forest - Harvested	In a wetland context, we interpret areas of harvested forest as probably involving permanent deforestation, either due to harvest of (1) exotic species planted in unsuitable conditions, producing timber of low commercial value and unlikely to be replanted, or (2) clearing of invasive tree species such as willows. In accordance with international greenhouse gas accounting approaches and national Tier 1 inventory reporting, carbon emissions resulting from permanent deforestation are assumed to be immediate on harvest and residues of biomass carbon are assumed to be zero (Steven Wakelin, Scion, personal observation 1 July 2022)	Treated as zero
Estuarine Open Water		Estuarine Open Water	Treated as zero	Treated as zero
Lake or Pond		Lake or Pond	Treated as zero	Treated as zero
Sand or Gravel		Sand or Gravel	Treated as zero	Treated as zero
River		River	Treated as zero	Treated as zero
Gravel or Rock		Gravel or Rock	Treated as zero	Treated as zero
Built-up Area (settlement)		Built-up Area (settlement)	Treated as zero	Treated as zero
Surface Mine or Dump		Surface Mine or Dump	Treated as zero	Treated as zero
Orchard, Vineyard or Other Perennial Crop		Orchard, Vineyard or Other Perennial Crop	Treated as zero: this cover class has a very minor influence on total carbon stocks (0.004% of total wetland area) and was not accounted for	Treated as zero
Landslide		Landslide	Treated as zero	Treated as zero
Transport Infrastructure		Transport Infrastructure	Treated as zero	Treated as zero

Scaling up from land cover class densities to national stock estimates

Carbon stocks

We intersected the LUM class "Wetland - Vegetated non forest" with land cover classes of the LCDB to calculate the relevant area of each land cover class, allowing us to calculate area-weighted estimates of carbon stocks. We refer here to carbon density as values of carbon stock per unit area, consistent with use elsewhere in forests (e.g. Mascaro et al. 2011; Pan et al. 2013) and wetlands (e.g. Lovelock et al. 2017; Tran et al. 2017).

For each mapping year we weighted the carbon density of each land cover class by the areal extent of that class. However, we understand that New Zealand's carbon accounting system currently can only handle one grand weighted mean density of carbon for wetlands (i.e. cannot re-assess the carbon density each mapping year). We therefore explored the effect of assuming (as the carbon accounting system does) a constant relative proportion of land cover classes, as compared to recalculating the relative proportion of each land cover class each year. We did this by calculating area-weighted means for each year of 2008, 2012, and 2016. We compared these numbers to the effect of applying the 2012 land cover composition to area-weight for each of the mapping years of 2008 and 2016. The latter option will only lead to carbon change where the total area mapped as LUM class "Wetland - Vegetated non forest" has changed, whereas the former will have a second source of carbon change where the areal extent of individual LCDB land cover classes within the LUM has changed.

Uncertainty around mean carbon stocks

We scaled up (or propagated) the uncertainties from land cover classes to national-scale using weighted quadrature, as per Equation (2), with the weight for above-ground carbon of each land cover class being its area. Except for mangroves, we were unable to estimate uncertainties for below-ground carbon stocks due to insufficient studies reporting these stocks in our literature review. As a result, we default to reporting a nominal error range of ± 75% for the mean (IPCC 2006) both for below-ground and for total biomass carbon. The range is interpreted as equivalent to two times the standard deviation above and below the mean (IPCC 2006). Although we estimated uncertainties for below-ground carbon stocks for mangroves, they are too distinct from other wetland vegetation types for their values to be applied more widely.

Results

Meta-analysis of carbon densities from compiled data from literature review

We identified 42 original biomass estimates for wetland vegetation from 14 New Zealand studies mostly conducted from the 2000s onwards (Woodroffe 1985; Agnew et al. 1993; Thompson et al. 1999; Pegman & Ogden 2005, 2006; Clarkson et al. 2009; Bassett et al. 2010; Pearce et al. 2010; Hodges 2013; Keyte Beattie 2014; Bulmer et al. 2016b; Tran et al. 2017; Burge et al. 2020; Suyadi et al. 2020), with three overseas studies (Kelleway et al. 2016; Owers et al. 2018; Santini et al. 2019). While most papers provided site-specific estimates, two papers presented data from multiple study sites to provide multi-site means. All the source studies were conducted in natural, and mostly unmodified wetlands. Biomass was usually estimated from harvests within a set area, except for mangroves, which relied on plot measurements and locally-derived biomass allometries. Compiled biomass estimates spanned six wetland types (swamp, saltmarsh [mangrove], shallow water, fen, pakihi, bog) and six structural classes (plus subdivisions in rushlands and mangroves). We found no studies that assessed biomass for flaxlands, mānuka wetlands, exotic willows, or wetlands with other woody vegetation, aside from mangroves. See Table 2 for the alternative data sources that were used.

Among structural classes, total above-ground carbon varied from a mean of 4.8 Mg C ha^{-1} for sedgelands to 23.8 Mg C ha^{-1} for tall mangroves (Fig. 2). For wetland types, total above-ground carbon varied from a group mean of 3.6 Mg C ha^{-1} for pakihi to 10.8 Mg C ha^{-1} for swamps and to 19.5 C ha^{-1} for mangroves (Appendix S3).

Estimates of below-ground biomass were uncommon (n = ten estimates from four studies) but the limited data available suggest that below-ground carbon density was less than total above-ground carbon density for three herbaceous structural classes (below-ground divided by above-ground ranged from 0.70 for reedland to 0.86 for rushland), while for mangroves below-ground density varied: below-ground divided by above-ground was 0.99 for tall mangroves and 2.09 for dwarf mangroves (Fig. 3). Below-ground biomass by wetland type is provided in Appendix S4.

Scaling up carbon densities and uncertainties

An intersect of land cover classes from LCDB v.5 with the LUM class "Wetland - Vegetated non forest" (from LUM v.8) for the map year 2012 yielded 32 land cover classes (21 vegetated classes) overlapping with LUM class "Wetland - Vegetated non forest". The two most extensive land cover classes were "Herbaceous Freshwater Vegetation" and "Manuka and/or Kanuka" with 112 201 and 26 601 ha, respectively, followed by 19 cover classes between 12 130 ha and 512 ha, and then a group of 11 cover classes with cover of 81 ha or less (Appendix S2). Eight structural classes were represented among the 26 estimates from source studies that provided locations in which we were confident (Appendix S1). The matches between these structural classes and four land cover classes mapped by LCDB v.5 were ecologically plausible: 13 sites with non-woody structural vegetation were mapped by LCDB v.5 as either "Herbaceous Freshwater Vegetation" or "Lake or Pond", one mixed rushland/tussockland was mapped as "Tall Tussock Grassland", and 12 mangrove sites were mapped as "Mangrove" by LCDB v.5. Changes in land use and land cover between sampling dates from source studies and mapping dates are unlikely because most studies (except for mangroves) were conducted in unmodified wetlands and some on public conservation land.

The area-weighted grand mean total carbon density for biomass in LUM class "Wetland - Vegetated non forest" (and spanning all wetland categories as mapped in 2012) is 19.90 (10.74–29.05 for 95% CI) Mg C ha^{-1} for above-ground biomass, 7.30 (1.82–12.77 nominal error range) Mg C ha^{-1} for below-ground biomass (i.e. excluding soils), and 27.195 (6.80–47.59 nominal error range) Mg C ha^{-1} for total above and below-ground biomass combined. Total carbon density was similar in 2008 (27.199; 6.80–47.60 nominal error range) and lower in 2016 (27.080; 6.68–47.48 nominal error range) (Table 3).

National carbon stocks in vegetated non-forest wetlands

The area of the LUM mapped as "Wetland - Vegetated non forest" was 219 307 ha in 2008, 217 234 ha in 2012, and 216

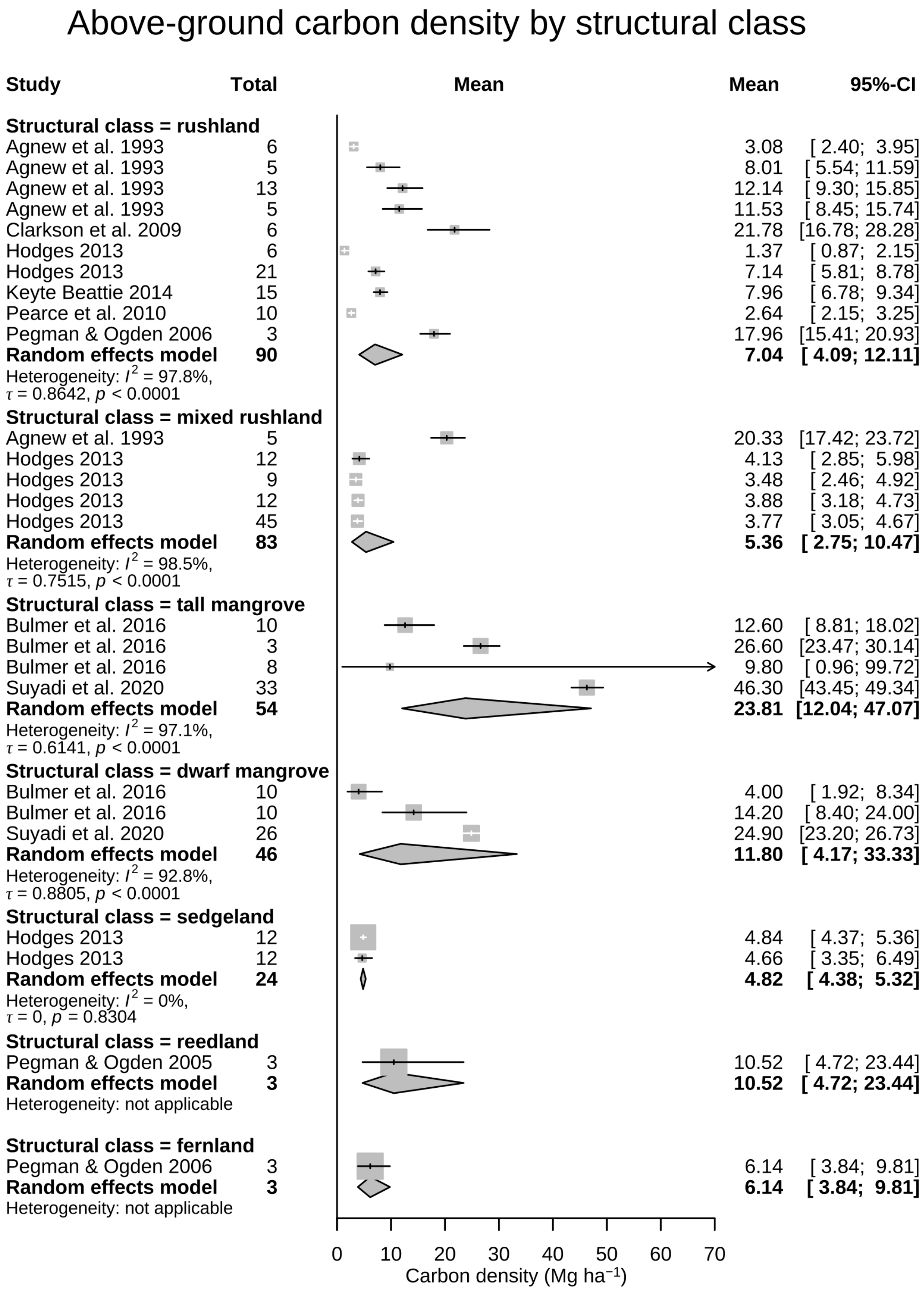

Figure 2. Summary of above-ground carbon density by vegetation structural class. Cross-study group means and heterogeneity estimates (grey diamond) were derived from a random effects meta-analysis.

717 ha in 2016, meaning a loss of 2073 ha from 2008 to 2012, and a loss of 517 ha from 2012 to 2016. Averaged over the entire eight-year reporting period, this equates to an annual loss of 324 ha per year.

Total vegetation carbon stocks in 2012 were estimated at approximately 6 million (1.5–10.5 million nominal error range) Mg C (see Table 3 for all three years and less coarse rounding) and these partition into land cover classes, and above and below ground components, as set out in Appendix S2. “Herbaceous Freshwater Vegetation” and “Herbaceous Saline Vegetation” have a much larger carbon store than expected based on carbon density alone because of their great extent, and the small extent of mapped mangroves make them a smaller than expected carbon store (Fig. 4). Figure 4 shows the carbon stocks of mangroves when areas of mangroves that are mapped by LCDB as land cover, but are not mapped at all by the LUM as land use, are added to the LUM class “Wetland - Vegetated non forest land use”. This increases the area of mangroves in 2012 mapped within the LUM class “Wetland - Vegetated non forest” from 621 ha to 25 655 ha.

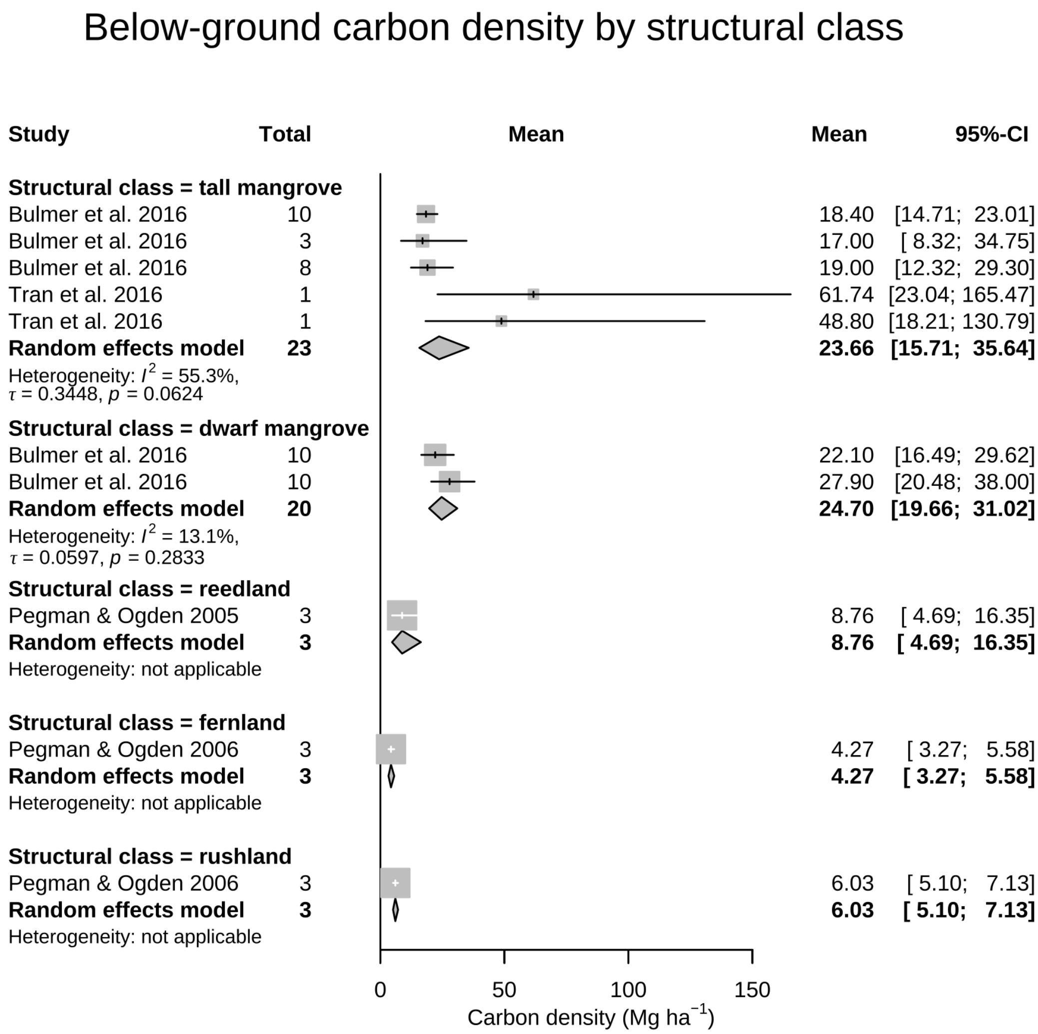

Figure 3. Summary of total below-ground carbon density by structural class. Cross-study group means and heterogeneity estimates (grey diamond) were derived from a random effects meta-analysis.

Table 3. Grand mean carbon density and total (above- and below-ground) carbon stocks in wetland vegetation within areas mapped as LUM class "Wetland - Vegetated non forest". Carbon stocks account for change in the extent of LUM class "Wetland - Vegetated non forest" and were calculated (1) assuming that the 2012 LCDB v.5 land cover composition represented 2008 and 2016 (fixed weights), or (2) accounting for any compositional shifts in land cover composition (updated weights). Nominal error ranges are presented in parenthesis (see Methods section on error propagation and estimated uncertainties). Note total stocks are calculated with more decimal places than shown here.

LUM map year	LUM area (ha)	Area-weighted total mean carbon density (Mg ha^{-1})	Total carbon stocks based on updated weights (Mg)	Total carbon stocks based on fixed weights (Mg)
2008	219 307.04	27.199 (6.80–47.60)	5 964 997 (1 491 249–10 438 744)	5 964 077 (1 491 019–10 437 135)
2012	217 233.91	27.195 (6.80–47.59)	5 907 698 (1 476 924–10 338 471)	5 907 698 (1 476 924–10 338 471)
2016	216 717.20	27.080 (6.68–47.48)	5 868 710 (1 467 178–10 270 243)	5 893 646 (1 473 411–10 313 880)

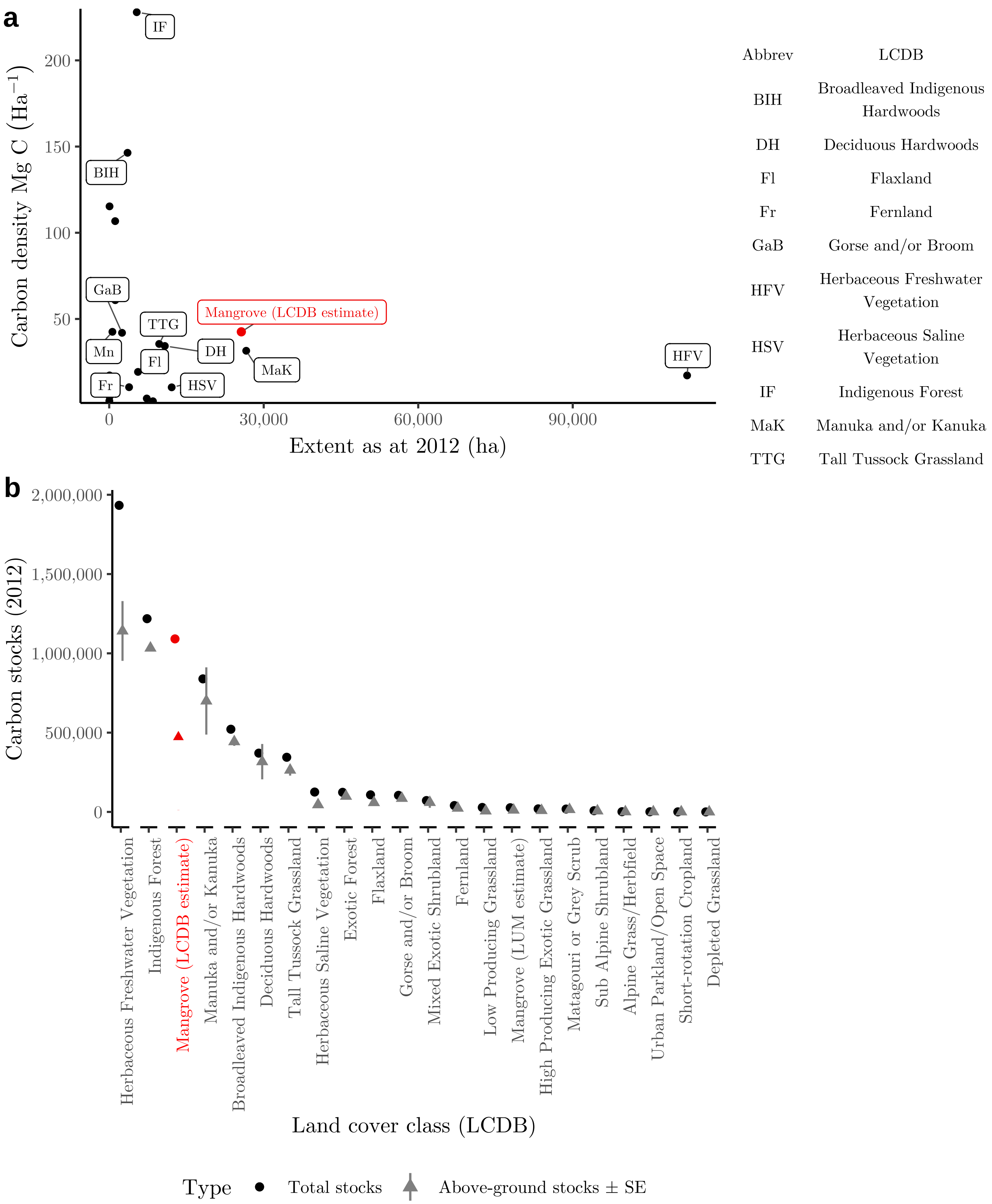

Figure 4. (a) Carbon density against extent for all land cover (LCDB) classes. Only classes where the total area was greater than 2000 ha and the carbon density was greater than 10 Mg C ha^{-1} are shown with labels. Abbreviations for labels are shown to the right of the figure. Mangroves mapped by LCDB that fall within the LUM class “Wetland - Vegetated non forest area” (total 621 ha) are not shown, as the extent is < 2000 ha. The area (25 655 ha) mapped by the LCDB but not included in any LUM class due to a restricted seaward LUM extent, along with the current area included in our analysis, is shown by “Mangrove (LCDB estimate)” in red. (b) Carbon stocks (Mg C) for total (above- and below-ground biomass) and above-ground biomass, as at 2012. All land cover classes from the LCDB that fell within the LUM class “Wetland - Vegetated non forest area” are shown. Mangroves that are included in the 621 ha currently accounted for in carbon mapping and mapped by the LUM are shown in black as “Mangrove (LUM estimate)”, and in red as “Mangrove (LCDB estimate)”, as per (a).

Discussion

We collated a range of information to calculate a total carbon stock of approximately 6 million (1.5–10.3 million nominal error range) Mg C (see Table 3 for all three years and less coarse rounding), made up of a density of 27.080 (6.68–47.48) Mg C ha^{-1}, and 216 717 ha in 2016, the most recent mapping year available. We compare our estimates with international wetland ecosystems, and then discuss limitations of our approach, opportunities for future research. Importantly, if the relative proportion of structural classes was known, our approach could be simplified by avoiding the use of land cover data. We consider this, alongside data collection to address under-sampled communities where uncertainty is high, as priorities to improve carbon stock estimation in New Zealand wetlands.

Carbon density variation within New Zealand and internationally

Within New Zealand

Areal extent is the major driver of carbon stocks attributed to each land cover class. Herbaceous freshwater vegetation is the largest LCDB land cover class within the "Wetland - Vegetated non forest class" mapped under the LUM and includes multiple vegetation structural classes with a range of carbon densities. Because we lacked data to spatially weight structural classes within LCDB classes, herbaceous freshwater vegetation had the same estimated density as the land cover class urban parkland (which had a relatively low carbon stock, due to its low areal extent). The densest carbon store in New Zealand was indigenous forest, which is not surprising given that, internationally, forests tend to have exceptionally high above-ground biomass. As extent varied more than density, it has a disproportionate impact where mapping errors or inconsistencies occur. We identified several key areas where improvements would reduce uncertainty and increase confidence in New Zealand reporting.

International comparisons

We collated international estimates of carbon density for herbaceous bogs, woody bogs, fens, and marshes/swamps to place our carbon density values within the global context (Table 4). This was considered to be the appropriate resolution for comparison, with regard to data availability. Overall, we find our estimates to be relatively close to international estimates. New Zealand mangroves were accounted for in a global analysis of mangrove biomass, and are on the lower end, as expected, because mangrove biomass declines with increasing latitude, and New Zealand mangroves are some of the highest latitude mangrove ecosystems (Hutchison et al. 2014). New Zealand swamp forests have a low carbon density compared to international estimates, although data for New Zealand are sparse and the New Zealand example is not based on plot data. We suggest further work is required to make robust international comparisons for this wetland type. New Zealand bogs and fens fall within the international range.

New Zealand forested bogs, such as those characterised by *Manoao colensoi*, *Lepidothamnus intermedius*, *Phyllocladus alpinus*, and *Halocarpus biformis* have no known biomass samples, compounding the lack of estimates for forested swamps. Therefore, for all forested wetlands, we applied the value published for all indigenous forests, which are primarily not wetlands (Table 2). New Zealand forested wetlands, which can grade to shrublands and are often adjacent to shorter-stature bog vegetation, such as *Empodisma* spp., have been described by Wardle (1979), Norton (1989), Brownstein et al. (2013), Vandergoes et al. (1997), Dawson (1988), and Mark and Smith (1975). Given the high potential for storage of above-ground (and below-ground, particularly where a peat substrate exists) carbon in these ecosystems, data collection is critical to understand the carbon benefits that are provided by forested wetlands.

Carbon stocks

Mangrove carbon stocks: missing extent

We derived two estimates of mangrove biomass, one where extent is restricted to the area mapped by the LUM class "Wetland - Vegetated non forest" (621 ha), and the other that includes areas outside of the LUM extent where land cover is

Table 4. Ranges, or means, in above-ground carbon densities for comparable wetland types found overseas and in New Zealand. The value from Shen et al. (2021) includes the 95 percent CI. We note that the Hutchison et al. (2014) work cited below for mangroves considers New Zealand biomass, but finds a strong latitudinal gradient in biomass, hence the overall mean (all latitudes) being higher than that which we use for New Zealand (southern latitude).

Origin	Wetland type	Above-ground carbon Mg ha^{-1}	Source
International	Bogs	9.58 (0.48–34.1)	Moore et al. (2002)
International	Poor fens	4.57 (0.08–16.4)	Moore et al. (2002)
International	Herbaceous marshes	2.25 ± 0.23	Shen et al. (2021)
International	Mangroves	77.62	Hutchison et al. (2014)
International	Swamp forests (peat; tropical)	132–199	Verwer & van der Meer (2023)
International	Swamp forests (riverine floodplains; USA)	133.1	Streeter et al. (2023)
New Zealand	Fens (non-forested)	4.34	This review
New Zealand	Bogs (non-forested)	8.79	This review
New Zealand	Swamps (non-forested)	10.84	This review
New Zealand	Mangroves	19.50	This review
New Zealand	Swamp forest (*Dacrycarpus dacrydioides*; hypothetical planting at 80 years planted at 1000 stems ha^{-1})	47.43	Paul (2021)
New Zealand	Forested bog	Unknown	None

mapped as mangroves (25 655 ha). This happens because the LCDB map of land cover extends farther offshore than does the LUM. Our estimate including offshore extent was more than 40 times greater than the alternative. This is important, because it is the LUM that is the basis for New Zealand's international carbon reporting, meaning that current formal reporting substantially underestimates wetland carbon in New Zealand. It is also noteworthy that the LUM includes mangroves within the LUM "Wetland - Vegetated non forest" class by definition, whereas ecologically mangroves are a forest type. Mangroves can exceed the 6 m height threshold that defines trees in New Zealand (McGlone et al. 2010), as heights of 7 m and 8 m have been recorded in the Paihia and Haruru Falls areas of northern New Zealand (Osunkoya & Creese 1997). This explains why their carbon density is an anomaly in a class defined as "non-forest". Moreover, more areas mapped as mangroves by LCDB are spatially congruent with areas mapped by the LUM as "Natural Forest" (1253 ha) than are congruent with areas mapped as "Wetland - Vegetated non forest" (621 ha). We suggest that a reassessment of the most appropriate LUM class be undertaken, and additionally thereafter, all 25 034 ha of mangroves currently not mapped in the LUM be included in such a class. This will be beneficial to account for a large missing carbon stock for New Zealand, but also because mangrove distribution and extent is changing rapidly. Mangroves have increased in the Auckland region from an estimated 2313 ha in 1940 to 10 483 ha in 2014, an increase of 3.2% per year (Suyadi et al. 2019).

Effect of on-going wetland loss

On-going loss of wetlands has been documented in New Zealand, particularly on private land (Robertson et al. 2019; Denyer & Peters 2020). To the extent that wetland is being converted to high producing grassland (as per Denyer & Peters 2020), this represents a likely decrease in vegetation biomass. However, these losses are small compared to the effects of drainage on soil carbon, particularly in peat wetlands, such as those in the Southland and Waikato regions (Ausseil et al. 2015; Campbell et al. 2015). In the three mapping years we examined, land cover classes were lost unevenly, meaning that the total wetland area changed (a loss of 25.9 km^2; see Results), as did the mean density of carbon in the remaining wetlands (a decline; refer Table 3). This has two implications. Firstly, uneven rates of loss among land cover classes mean that a static weighted value fixed in one year is less accurate than a re-weighted value for reporting purposes. Secondly, mean carbon density across land transitioning into vegetated wetlands and mapped as LUM class "Wetland - Vegetated non forest" may not be the same as the overall weighted mean, and so estimated gains may be biased.

Representativeness of carbon density estimates and implications for scaling-up

Biomass estimates were not available for several land cover classes, such as flaxlands (5565 ha in LUM year 2016), herbaceous saline vegetation (12 069 ha in LUM year 2016), and some shrubby or forested wetlands. This is important given the estimated extent of these structural classes and their potential carbon densities. In this assessment we used carbon density surrogates from non-wetland contexts. If the distribution of biomass differs between terrestrial and wetland sites within the same vegetation structural class, the use of surrogates introduces uncertainty into our estimates. This raises the question of how best to estimate biomass for vegetation structural classes on wetlands for which biomass has not been measured. Stem size and density decrease with increasing wetness, even in wetland tree species such as *Dacrycarpus dacrydioides* (Wardle 1974).

Whilst individual source studies largely followed objective sampling designs and used standard procedures for defining a sampling universe and estimating biomass, collectively, our combined sources were not designed, and are insufficiently robust, for an unbiased estimation of wetland vegetation biomass or carbon at national scale. New Zealand non-forest vegetated wetland types vary widely in their vegetation composition, function, and structure (Johnson & Gerbeaux 2004; Clarkson et al. 2015). Beyond those land cover classes mapped by LCDB v.5, the spatial extent of different wetland vegetation structural classes is unknown, and thus the carbon densities presented here do not account for the relative extent of different vegetation structural classes. For example, we assumed that the relative composition of LCDB land cover class herbaceous freshwater vegetation, was the same as that found in the LCDB land cover class urban park/open space (Fig. 1). Mangroves in New Zealand are an exception as they have been extensively sampled, including both biomass estimates and extent of dwarf and tall mangroves (Suyadi et al. 2019). This allowed us to account for the relative abundance of dwarf and tall mangroves within the area mapped as mangroves by LCDB v.5. The benefit of this rich information source is evident in the low sampling uncertainty for mangroves (CV 4.1%). A recent study that post-dates our literature review strengthens estimates for coastal wetlands (mangroves, saltmarshes, and seagrasses) (Bulmer et al. 2024).

The IPCC recommends that carbon is delineated separately for inland and coastal wetlands (Hiraishi et al. 2014). New Zealand-specific biomass and carbon data were unavailable to split estimates of carbon stocks within each land cover class into those that apply to coastal versus inland areas. New primary data is required to address this information gap. As such, we had to assume that each land cover class maintains constant carbon stocks across inland and coastal areas.

Estimated uncertainties are themselves uncertain (Harrer et al. 2021) and as many of the uncertainties in above-ground carbon presented here are approximated from surrogates or based on small sample sizes, they should be interpreted as indicative and applied with caution. Additionally, a lack of information meant we had to rely on IPCC default values for uncertainty for below-ground stocks, and consequently, for total carbon stocks. The use of default values leads to very high uncertainty in stock estimates; we consider this could be reduced if uncertainties could be estimated for below-ground and total stocks, and replication for above-ground carbon stocks were increased.

Improving estimates of vegetation carbon stocks and densities

Our estimates of carbon stocks had high uncertainty because the data available from the literature review were fragmentary, particularly for woody vegetation, below-ground pools, and coastal ecosystems. Here, we provide recommendations to improve precision, and reduce potential bias and uncertainty.

We recommend making new empirical estimates of wetland vegetation biomass in tandem with an unbiased survey of vegetated wetlands, building on the successful approach developed for carbon accounting in natural forests (Holdaway et al. 2017; Paul et al. 2021). Empirical sampling could be comprehensive, targeted, or some combination of the two. Comprehensive sampling would involve a two-phase approach.

Firstly, this would involve destructive above- and below-ground biomass harvests coupled with field measurements to develop the allometric relationships required to link vegetation dimensions to vegetation biomass. Harvests and measurements would need to span the full range of vegetation structural classes present in wetlands (e.g. scattered shrublands, diverse herbaceous forms). Secondly, these measurements would then be applied to an unbiased network of permanent vegetation plots that sample vegetation within the LUM class "Wetland - Vegetated non forest". This approach would reveal the true vegetation cover within the LUM (without relying on the land cover composition as mapped by LCDB within the LUM), and field measurements would provide empirical biomass estimates for a range of vegetation types. Remeasurement would provide empirical data on change in vegetation cover across the LUM class. Targeted sampling would involve destructively harvesting above- and below-ground biomass in the most common vegetation structural classes, and those LCDB classes and vegetation structural classes where the information from the literature review was weakest. This approach would assume that current mapping information is adequate and would focus only on improving the biomass values applied to the existing mapped area. Key foci would be woody wetlands, coastal and inland ecosystems within the same LCDB class, and the LCDB herbaceous freshwater vegetation class, as this is the largest LCDB class by area. However, continuing to stratify by LCDB will introduce an extra set of classes for which statistical replication will be required; it may be more efficient to simply adopt the comprehensive approach. A more parsimonious approach would be to undertake a circumscribed, but spatially representative, sample of structural classes (no biomass estimation required) to understand the spatial distribution of structural classes within the area of interest (LUM class "Wetland - Vegetated non forest"). This would avoid the need to use land cover class data and its associated unquantified uncertainties. Several structural classes would remain data deficient and would require supplementary biomass and carbon sampling.

We further suggest that carbon reporting should account for changes in land cover within the area mapped by the LUM. In the absence of an empirical, permanent plot-based sampling approach, we recommend a review of carbon accounting practices for wetlands such that changes in the extent of LCDB classes within the LUM can be accounted for at each measurement period. This would more appropriately account for changing vegetation structure and composition across wetlands, and more closely track the nature of these changes. Simultaneous sampling of the LCDB and the LUM (i.e. the same mapping years) would facilitate stronger inferences of land cover as mapped under the LCDB to the LUM. In undertaking this work, we have confirmed that although the metadata for the LUM and LCDB suggest that the 2008 nominal steps differed by 11 months, the imagery used to derive those layers is the same. However, the LUM year 2016 had the closest analogue with LCDB year 2018. Consideration could also be given to incorporating LUM mapping error into all carbon calculations (i.e. all ecosystems, not only wetlands).

Current carbon estimates are based on the LUM, which has a limited spatial extent beyond the shoreline and underestimates the distribution of mangrove vegetation (discussed above). We recommend extending the LUM to incorporate near-shore areas in which mangroves occur currently or could occur in the future. This would improve the accuracy of the estimate of wetland vegetation carbon and allow for changes in wetland vegetation carbon due to projected changes in mangrove extent to be taken into account.

Finally, while noting that wetland soil carbon is outside the scope of this paper, we would suggest where possible to combine vegetation and soil carbon sampling. This would allow, subject to replication, a link between soil carbon and the vegetation communities which promote it.

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Supplementary material

Additional supporting information may be found in the online version of this article:

Appendix S1. Map of study locations

Appendix S2. LCDB class areas

Appendix S3. Meta-analysis results by wetland type: above-ground

Appendix S4. Meta-analysis results by wetland type: below-ground

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