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Development of a carbon calculator tool for riparian forest restoration

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Abstract

Aim(s): Methods for estimating carbon sequestration in riparian forest biomass are needed to qualify riparian restoration and conservation projects for funding through carbon credits. As part of a proposed accounting methodology for California's cap-and-trade system, we described five riparian vegetation associations and created a tool to predict their rates of carbon accumulation under different restoration and land-use scenarios.

Location: California, USA.

Methods: We assembled a database of 654 forest inventory plots of known age (3–108 years) from replanted and naturally recruiting California riparian forests. We then used detrended correspondence analysis and agglomerative hierarchical clustering to detect vegetation structure in the plots and delineate distinct forest types. We used allometric equations to estimate biomass from individual trees' diameters and calculate total biomass for each plot. Next, we fitted sigmoid growth functions to the plot biomass data to generate expected values for live-tree biomass over a 100-year period for each forest type. Finally, we created an algorithm to match user-inputted data for an intended restoration project with a look-up table that predicts carbon accumulation appropriate to the user's scenario.

Results: California riparian vegetation was divided into willow scrub, cottonwood–willow, mixed riparian, upland riparian and riparian woodland types. Total carbon sequestration in riparian vegetation and soil varied from 75.7 to 137.4 Mg C/ha at 30 years post-restoration and from 95.1 to 175.8 Mg C/ha 100 years after stand initiation. Vegetation types dominated by taller, more shade-tolerant trees developed more slowly than scrubby and pioneer stands, but came to a higher maximum biomass. Our models put riparian biomass accumulation in the range of the few extant literature values for mediterranean systems, but uncertainties are high, and soil carbon may be underestimated.

Conclusions: Our workflow and methods should be transferable to the development of carbon accounting tools for any other woody vegetation type. However, our difficulty in finding appropriate published data for the analysis suggests a critical need for field surveys appropriate to biomass estimation in woodland and forest

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Nomenclature: USDA Plants Database (http://plants.usda.gov/java/; accessed 1 Feb 2018)

communities not exploited for timber. The most important measurements are the age of the stand and a census of tree species, height and diameter.

KEYWORDS

biomass equations, cap-and-trade, chronosequence, coarse woody debris, DBH, downed dead, forest floor, growth curves, natural capital, passive restoration, payment for ecosystem services, riparian buffer

1 | INTRODUCTION

The potential for reforestation and habitat restoration to be embraced as a means of halting or mitigating climate change has grown rapidly with the adoption of payment for ecosystem services (PES) schemes worldwide. Current, recent or proposed mechanisms to pay for the ecosystem service of carbon sequestration in restored terrestrial vegetation include the UN's Reducing Emissions from Deforestation and Forest Degradation (REDD) programme operating in developing countries (http://www.un-redd.org/), the Emissions Reduction Fund in Australia (http://www.environment.gov.au/climate-change/emissions-reduction-fund) and the California Air Resources Board's greenhouse gas cap-and-trade system (https://www.arb.ca.gov/cc/capandtrade/capandtrade.htm). Payment for replanting or conserving forests is also a feature of voluntary carbon-crediting agencies, such as Verified Carbon Standard and the American Carbon Registry.

The PES schemes are controversial, particularly with respect to their effectiveness in achieving hoped for outcomes with respect to biodiversity conservation (Muradian et al., 2013; Redford & Adams, 2009). A common critique of PES systems focused on a single ecosystem service, such as carbon, is that they may bring about perverse conservation outcomes by relying on monocultures, non-native species or on artificial, engineered systems (Lindenmayer et al., 2012). Many have argued that PES systems, which are still in their infancy, can be effective for conservation if properly designed (Essl, Erb, Glatzel, & Pauchard, 2018; Reed et al., 2017; Wunder, 2013). Systematic reviews of PES effectiveness (Hejnowicz, Raffaelli, Rudd, & White, 2014) are emerging in the literature, contributing to development of a more robust theoretical approach (Börner et al., 2017) to the design and implementation of PES mechanisms.

One area of concern expressed about PES implementation centres on the ability of these programmes to accurately measure the size of the ecosystem service. In the case of carbon, regulatory entities may require post-project reporting of actual greenhouse gas offsets, or individuals may need pre-project estimates of carbon sequestration potential in order to decide whether to embark on a land-use change that could be compensated by carbon credits. Either way, the standard method is based on non-destructive forest inventory practices that census individual trees, measure their heights and trunk diameters, and use allometric equations to convert height and diameter data to volume or biomass (and thence to carbon). To model predicted carbon accumulation over long time scales requires empirical data from either repeat-sampled forest inventories, or from chronosequences of sites of similar origin sampled at different ages. Alternatively, software used to predict timber yields in forestry applications, such as the Forest Vegetation Simulator (Crookston & Dixon, 2005), can be adapted to estimate carbon accumulation. While carbon-crediting programmes typically require on-ground forest inventories to verify actual carbon sequestration before issuing credits, predictive models based on empirical data or on computer modeling are useful in helping stakeholders estimate the income potential, or greenhouse gas (GHG)-reducing potential, of specific changes to land use (Lubowski, Plantinga, & Stavins, 2006).

The standard methods for carbon inventories were developed by foresters explicitly for timber stands, and there is a wealth of data available from commercially exploitable tree species to parameterize their models. However, carbon accounting in woody vegetation types that are not useful for timber production, such as riparian forests and woodlands, is hampered by a lack of appropriate measurements (Gonzalez, Battles, Collins, Robards, & Saah, 2015). Overcoming this obstacle, to include riparian forest conservation and restoration in carbon-crediting programmes, is desirable for several reasons. First, despite their small extent, riparian forests in the western US have an unusually large impact on biodiversity, supporting approximately a third of terrestrial plant species and a majority of vertebrate species (Knopf, Johnson, Rich, Samson, & Szaro, 1988; Patten, 1998; Poff, Koestner, Neary, & Henderson, 2011). Second, riparian systems are acknowledged to provide a wide variety of ecosystem services other than carbon sequestration, including flood mitigation, nutrient retention and temperature regulation, while also playing a critical role in climate resilience (Capon et al., 2013). Third, floodplains are among the ecosystems most threatened by conversion to other land uses, such as agriculture and residential development (Krueper, 1993; Opperman, Moyle, Larsen, Florsheim, & Manfree, 2017). In the western US, riparian zones are a frequent target of restoration activities which, though typically aimed at improving salmonid habitat, floodplain connectivity or water quality (Kondolf et al., 2007), also provide a carbon sequestration co-benefit and could be funded by carbon credits (Matzek, Puleston, & Gunn, 2015).

As part of an effort to qualify stream corridor restoration for funding under the California (USA) cap-and-trade programme, we developed a tool for estimating carbon sequestration in riparian forests (https://secure.conservation.ca.gov/creec/). The calculator supposes that the user is planning to plant a riparian restoration project (active restoration), take measures to encourage natural regeneration that would not otherwise occur (passive restoration) or preserve an existing riparian site from cutting (avoided conversion). Our effort faced several challenges: a lack of published data comprising appropriate measurements for riparian species (e.g. DBH); the need to map users' planned community composition inputs onto a small set of distinct vegetation types; and a policy requirement that new carbon accounting protocols be methodologically consistent with previous ones. This paper relates how we addressed these constraints in developing the tool, and reports predicted carbon sequestration for five distinct riparian forest vegetation types.

2 | METHODS

Development of the carbon calculator tool proceeded in five phases: (a) assembling a database of appropriate known-age forest inventory plots; (b) delineating distinct vegetation types among the plots in the database; (c) estimating above-ground tree biomass from inventory data; (d) modelling carbon sequestration and summarizing the model outputs into look-up tables for each vegetation type; and (e) creating an algorithm to match user-inputted community composition data with look-up tables in the online tool (Figure 1).

2.1 | Assembling the database

Few published data reporting riparian forest inventories in stands of known age exist for California. In the U.S. Forest Inventory Analysis (FIA) database (https://www.fia.fs.fed.us/), which consists of repeat-measured permanent forest inventory plots from around the US, we found only a handful of plots in California representing the riparian vegetation of lowland rivers and streams. We therefore contacted researchers working in the field and requested unpublished data from chronosequences, eventually compiling a database of 654 known-age forest plots that inventoried woody stems >2.5 cm DBH. Some of these stands were of exact known age because they were part of a large-scale, decades-long effort to restore riparian forest on floodplains of the Middle Sacramento River (Golet et al., 2013) or smaller-scale restoration efforts on rangelands north of San Francisco Bay (Lennox et al., 2011). Others were plots of naturally recruiting vegetation (Stella, Riddle, Battles, Hayden, & Fremier, 2012) for which age was estimated from remote sensing data (Stella et al., 2011), generally constrained within a window of 10 years or less, using a floodplain-age mapping model (Greco, Fremier, Larsen, & Plant, 2007). It is important to note that these ages refer to the stand; individual trees measured in the inventories may be younger than the stand age. Our 654 plots, of which only 45 are included in previously published work, represent 59 separate sites in major watersheds of northern California (Supporting information Appendix S3).

In addition to compiling the database, we had to reconcile differences in data collection and measurement among different researchers. A subset of 45 plots only had measurements of trees 5 cm and above, and a different subset of 177 plots did not include measurements of standing dead trees; all of these plots were under 25 years old. For these, we modelled the missing data by finding the significant linear relationship between the age of the plot and the proportion of total woody canopy biomass that occurred in the "missing" category, using the <25-year-old plots for which we had complete data. We then used this linear equation to interpolate plot-level biomass values where we had incomplete data. When these "missing data" plots were subsequently classified as belonging to a particular

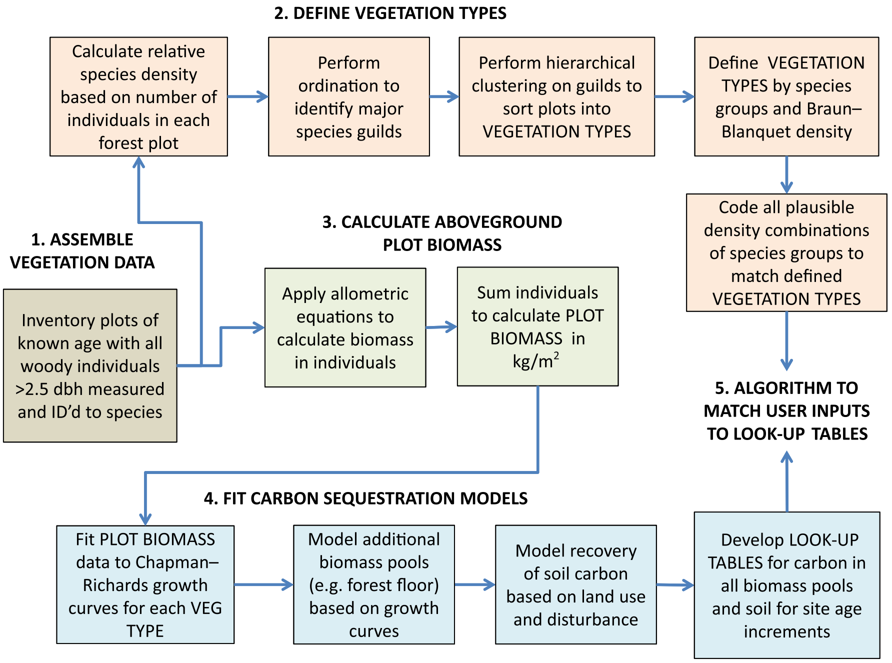

FIGURE 1 Phases of creating a tool to predict the carbon sequestration potential of planned riparian restoration and conservation activities

vegetation type (see below), they tended to be evenly distributed across vegetation types except for the "mixed riparian" type, which had 24% of its plots missing sapling data and 34% of its plots missing standing dead data. Additionally, some data sets collected stem diameters for woody shrubs such as *Baccharis pilularis*, whereas others collected elliptical canopy volumes, which are more appropriate for allometric equations that convert in-field measurements to biomass. To reconcile these metrics, shrub stem diameters were converted to canopy volume by ranking both diameters and canopy volumes and assuming that for every percentile range, the distribution of diameters matched the distribution of volumes. After adjusting for these differences in data collection, the final database contained more than 25,000 live and dead individual trees and woody shrubs.

2.2 | Delineating vegetation types

We reviewed the vegetation associations in the *Manual of California Vegetation* (Sawyer, Keeler-Wolf, & Evens, 2009), a comprehensive botanical treatment of regional plant communities that recognizes more than 20 alliances with riparian species as dominants. However, these were too fine-grained and numerous for our needs. To make a matching algorithm feasible in the calculator, we needed a relatively small number of ecologically meaningful vegetation types, each containing enough plots from our database to produce statistically robust growth curves, and each definable on the basis of functional groups or guilds rather than as combinations of 20+ individual species (Merritt, Scott, LeRoy, Auble, & Lytle, 2010). We began with an initial ordination of the 654 plots by relative species density, using detrended correspondence analysis (DCA) in the R package *vegan* (R Foundation for Statistical Computing, Vienna, AT; Supporting information Appendix S4). Axes in the DCA suggested that species tended to group together by habit, genus and successional status: scrubby willows, pioneer trees, upland shrubs and shade-tolerant trees. Similar groupings were used by Stella and Kui (2015) to predict the distribution of coarse woody debris production by floodplain age. We considered this division to be ideal because it combines phylogeny, ecology and biomass, and because other California woody riparian species could readily be assigned to one of the four groups. Species in our data set belonging to the shrubby willow group included *Salix exigua*, *S. laevigata*, *S. lasiolepis* and *S. lucida*. Pioneer trees included *Populus fremontii*, *Salix gooddingii* and *Alnus rhombifolia* as well as non-native invaders such as *Ailanthus altissima*. Upland shrubs included *Baccharis* spp., *Centranthus occidentalis* and *Sambucus* spp. Shade-tolerant trees included *Acer negundo*, *Fraxinus latifolia*, *Platanus racemosa*, *Juglans californica* and *Quercus lobata*.

After converting species relative density to relative density in each of the four groupings, we used agglomerative hierarchical clustering in the R package *vegan* to detect structure in the data and delineate vegetation types. We used the unweighted pair group method with arithmetic mean, UPGMA (Michener & Sokal, 1957), based on Bray–Curtis dissimilarity (Bray & Curtis, 1957) and truncated the resulting dendrograms to produce five clusters, according to a suggested optimal cluster number (R package *NbClust)*. These five vegetation clusters (Table 1) were examined and the vegetation associations described as follows: "willow scrub" (dominated by shrub-form willows, with up to 25% pioneer trees); "cottonwood–willow forest" (dominated by pioneer trees, with up to 25% shrub-form willows); "upland riparian forest" (>50% shade-tolerant trees, not more than 15% shrubs, willows or pioneer trees); "riparian woodland" (open, low-density vegetation dominated by shrubs and late-successional trees, especially oaks); and "mixed riparian forest" (dense, diverse forests not dominated by any species group). We also combined all naturally recruited (i.e. unplanted) forest plots into a "natural regeneration" type for a separate use in our online tool (see below); these plots are therefore represented twice in the clustering.

2.3 | Estimating biomass

Species-specific allometric equations are not available for most of the species in our data set. We therefore estimated above-ground biomass from DBH using generic allometric equations that apply to sets of tree species grouped by taxonomy and wood density (Chojnacky, Heath, & Jenkins, 2013). For some non-native species in our database, such as *Tamarix* sp. and *Ficus carica*, we assigned

TABLE 1 Vegetation groups as determined by hierarchical clustering

Vegetation type	Description	*n*
Riparian woodland	Co-dominated by shrubs and large trees, especially oaks; low density	74
Cottonwood–willow forest	Dominated by pioneer trees; up to 25% shrub-form willows	103
Upland riparian forest	Dominated by shade-tolerant trees; ≤15% shrubs, willows or pioneer trees	209
Natural regeneration	Group composed of all naturally recruiting plots	385
Mixed riparian forest	Diverse; not dominated by any species class	99
Willow scrub	Dominated by shrub-form willows; up to 25% pioneer trees	169

Notes. The number of plots in each group is given by *n*. The 385 "natural regeneration" plots are shown as a separate category and are also included in the other vegetation types; the total number of plots in the database is 654.

available equations based on wood density alone because we could not match by genus. Forest carbon protocols previously approved by the California agency responsible for cap-and-trade, the California Air Resources Board (CARB), rely on a different set of allometric equations (Woodall, Heath, Domke, & Nichols, 2011) that require both DBH and height as inputs. However, our data set contained height data for only 8% of trees. The Chojnacky equations tend to estimate greater above-ground biomass than the Woodall equations (Chojnacky et al., 2013), so we adjusted our plot-level biomass estimates downward by the amount of the assumed overestimate (by 25.5% for plots dominated by ash/willow/cottonwood and 19.5% for plots dominated by oaks and other hardwoods). We did this not because the Woodall equations are known to be more accurate but because CARB requires carbon estimates be methodologically consistent and conservative in the face of uncertainty (CARB 2017). After calculating above-ground biomass, we used the two-parameter equation of Cairns, Brown, Helmer, and Baumgardner (1997) to predict below-ground biomass in each forest plot from age and above-ground biomass. This equation takes the form

$$\mathrm{BGB} = e^{[k+a(\ln A)+b(\ln B)]}$$

where BGB is the below-ground biomass of a plot, *k*, *a* and *b* are regression coefficients, *A* is the above-ground biomass of the plot and *B* is the age of the plot. This equation is used because most forest inventories, ours included, take measurements on live trees and cannot directly estimate biomass below the bole. Coefficient values used with all carbon component equations are reported in Supporting information Appendix S1. Following typical forest carbon accounting assumptions, biomass was considered to be 50% carbon.

2.4 | Modelling carbon sequestration

To model carbon accumulation in above-ground biomass over time, we used nonlinear regression in the R package *nlstools* to fit generalized Chapman–Richards growth functions (Liu & Li, 2003; Pienaar & Turnbull, 1973) to the plots in each vegetation group. Equations took the form

$$\mathrm{Biomass} = a(1-e^{-b(\mathrm{age})})^{c}$$

where *a* is the maximum stand-level tree biomass, *b* is a growth rate that determines how fast the stand reaches its asymptote and *c* determines the shape of the sigmoid curve and the location of its inflection point. In accordance with theory suggesting that stand growth operates at either quarter- or third-power scaling (Enquist & Niklas, 2001), *c* was initially constrained to equal either 3 or 4. However, we found that the exponent mattered little to the curve fit and generated nearly identical parameter estimates, so we defaulted to the value of 3, which is equivalent to the well-known Von Bertalanffy equation. Curves were iteratively fit using nonlinear least squares, until they converged on an optimal solution; appropriate starting values were suggested by graphical previews of curve fits. Due to non-normality in the regression residuals, we estimated 95% confidence intervals for parameter estimates (*a* and *b*) with a non-parametric bootstrapping procedure (*nlsBoot*). To estimate the propagated error from the non-linear fit around a single point estimate, we repeated the non-linear curve fitting in a Bayesian framework in R package *brms*, using the parameter estimates as priors. We then generated Bayesian 95% confidence intervals from the posterior draws for point estimates of biomass at 30 and 100 years of age.

We estimated the sigmoid growth functions twice, first on the combined biomass (live plus standing dead trees) to generate expected values in the look-up tables, and then on the live-tree biomass alone. This latter analysis was necessary because the equations used to estimate additional forest carbon pools for which we lacked empirical data – for example, downed dead wood and understorey – require knowing the amount of live-tree carbon (Smith, Heath, & Hoover, 2013). The methods we employed to estimate these additional forest pools are those used for reporting US greenhouse gas sources and sinks to the National Greenhouse Gas Inventory (Heath, Smith, Skog, Nowak, & Woodall, 2011).

Downed dead wood carbon was considered to be a simple ratio of live-tree carbon (ratio = 0.062 for elm, ash, cottonwood, willow and oak-dominated plots; ratio = 0.095 for alder-dominated plots), as in FORCARB2, the U.S. forest carbon budget model (Heath, Nichols, Smith, & Mills, 2010).

Understorey carbon (herbs, vines and saplings) was modelled to increase briefly at the beginning of stand replacement but then decline over time as understorey plants are shaded out by the closure of the canopy. The relationship is

$$U = \mathrm{livetreeC} \times e^{c_1-(c_2 \times \ln(\mathrm{livetreeC}))}$$

where *U* is understorey carbon, livetreeC is the carbon in live trees, and c_1 and c_2 are coefficients appropriate to western U.S. hardwood forests. Both equation and coefficients come from Smith et al. (2013), building on the work of Birdsey (1996) and Heath et al. (2010).

Forest floor carbon was estimated as in Smith and Heath (2002), as a function of stand age, using the equation:

$$\mathrm{FF} = \frac{f_1 \times (\mathrm{age})}{f_2 + (\mathrm{age})}$$

where FF is forest floor carbon, f_1 and f_2 are coefficients appropriate to Western hardwood forests. An alternate form of this equation includes a decay component to represent the biomass input from slash left after clearcutting; we neglected this term because restored or preserved riparian forests would not typically follow a clear cut.

We had too few data for soil carbon in the known-age plots to constrain its accumulation over time in the different vegetation types. Instead, we made the assumption that planted restoration projects would typically occur on degraded or depleted soils, and that revegetation would gradually replenish the soil carbon pool to an expected mean value. The amount of soil carbon at any time point is given by the equation

$$\mathrm{soilC} = \mathrm{meansoilC} \times (p + (1-p) \times (1-e^{-(\frac{\mathrm{age}}{50})^2}))$$

where values for mean soil carbon are taken from Smith et al. (2013) and values of p, the initial proportion of remaining soil carbon, were set for different pre-restoration land uses based on a review of broad-scale meta-analyses assessing the impact of agricultural activities on soil carbon stocks (Guo & Gifford, 2002; Laganiere, Angers, & Pare, 2010; Li, Niu, & Luo, 2012). Look-up tables include only the recovered soil carbon, not the total amount present in the soil, at each time step.

Our data set had several important gaps, most importantly that it included no known-age inventory plots from high mountain elevations. High-elevation riparian forests may be expected to experience cooler, wetter, shorter growing seasons than their valley and foothill counterparts. Fortunately, high-elevation sites were well-represented in the FIA database, and an online estimator, COLE 3.0 (Van Deusen & Heath, 2010), generates look-up tables from FIA plots using nearly identical methods to ours. We designated "high-elevation" riparian forests as a seventh vegetation type (data not shown) and queried COLE 3.0 for a look-up table based on FIA plots located in the appropriate California sub-regions, Ecological Sections M261A-M261G (Cleland et al., 2007), and containing riparian indicator species or species groups. In the carbon calculator tool, this look-up table appears for any project located above 1,000 m a.s.l. in the Klamath, Cascades or Sierra Nevada mountain ranges, with appropriate soil carbon/land-use variants. We also lacked data for the drier, hotter riparian zones of southern California, so CREEC defaults to the lowest biomass forest type (willow scrub) for this region.

2.5 | Creating a matching algorithm

After calculating or modelling all the carbon pools, we programmed the resulting look-up tables into an online carbon calculator, the Carbon in Riparian Ecosystems Estimator for California (CREEC). The tool connects the user's plans for a GHG-reducing riparian forest intervention with the appropriate look-up table summarizing the carbon sequestration potential of the activity. The three types of intervention are *planting* (active restoration), *natural regeneration* (passive restoration) and *avoided conversion*. For planting and avoided conversion projects, the user's known or intended community composition is converted into Braun–Blanquet density classes for each of the four functional groups (e.g. pioneer trees). We mapped all possible combinations of relative densities in the functional groups onto one of the five vegetation types generated from our database, to connect the planned plantings or existing forests with an appropriate look-up table. Natural regeneration projects, which do not have a planting plan, are matched with look-up tables pertaining to the entire set of naturally regenerated plots (excluding restored plots). Variants of the look-up tables with identical values for carbon in plant biomass but different rates of soil carbon recovery are returned by CREEC according to user inputs for the pre-restoration land use (e.g. grazing, crops).

3 | RESULTS

Complete carbon look-up tables for the five identified vegetation types and the set of naturally regenerated sites can be found in Supporting information Appendix S2. Growth curve parameter estimates and 95% confidence intervals for canopy (live plus standing dead tree) biomass, as well as point estimates at 30 and 100 years of age, are summarized in Table 2.

Predictions for canopy carbon (i.e. 50% of canopy biomass) in riparian vegetation 30 years after stand initiation ranged from 42.8 Mg C/ha (95% CI: 38–48.9) in willow scrub to 101.2 Mg C/ha (95% CI: 87.9–113.5) in cottonwood–willow forest. The 100-year predicted canopy biomass carbon in riparian forests ranged from 43.6 Mg C/ha (95% CI: 38.3–50.0) in willow scrub to 118.5 Mg C/ha (95% CI: 93.1–145.2) in riparian woodland. Mean values for canopy carbon across the five forest types were 63.9 ± 1.0 Mg C/ha at 30 years and 81.5 ± 1.4 Mg C/ha at 100 years of age. The natural regeneration set of plots had 72.4 Mg C/ha (95% CI: 62.0–80.9) at 30 years and 83.2 Mg C/ha (95% CI: 74.2–92.5) at 100 years.

Total carbon in the other vegetation components combined (downed dead, forest floor, understorey) ranged from 23.1 to 26.4 Mg C/ha at 30 years and from 37.7 to 42.0 Mg C/ha at 100 years. Downed dead carbon represented ~3%–5% of the total non-soil carbon at age 30 and at age 100 years, depending on forest type;

TABLE 2 Parameter estimates and confidence intervals for the Chapman–Richards growth function, defined as live-tree biomass = $a(1-e^{(-b^{*}\text{age})})^{3}$, with age in years and live-tree biomass in Mg C/ha

Vegetation type	*a*	95% CI	*b*	95% CI	*est30*	95% CI	*est100*	95% CI
Riparian woodland	245.1	190.7, 345.3	0.05	0.03, 0.08	99.6	38.3, 179.3	237	77.5, 100.0
Cottonwood–willow forest	215.5	178.6, 245.1	0.13	0.09, 0.20	202.3	175.8, 226.9	215.5	176.0, 257.6
Upland riparian forest	180	158.0, 205.7	0.1	0.08, 0.13	156.5	133.3, 176.4	180	157.7, 205.1
Natural regeneration	166.4	150.4, 186.2	0.1	0.08, 0.14	144.7	123.9, 161.8	166.4	186.2, 290.4
Mixed riparian forest	95.2	78.4, 122.7	0.28	0.17, 0.56	95.1	80.9, 110.1	95.2	81.0, 110.6
Willow scrub	87.2	69.2, 116.2	0.17	0.11, 0.33	85.6	76.0, 97.7	87.2	148.4, 185.1

Notes. The parameter *a* represents the maximum biomass accumulation and *b* is a growth rate. The table also shows Bayesian upper and lower confidence bounds for live-tree biomass at a stand age of 30 years (*est30*) and 100 years (*est100*).

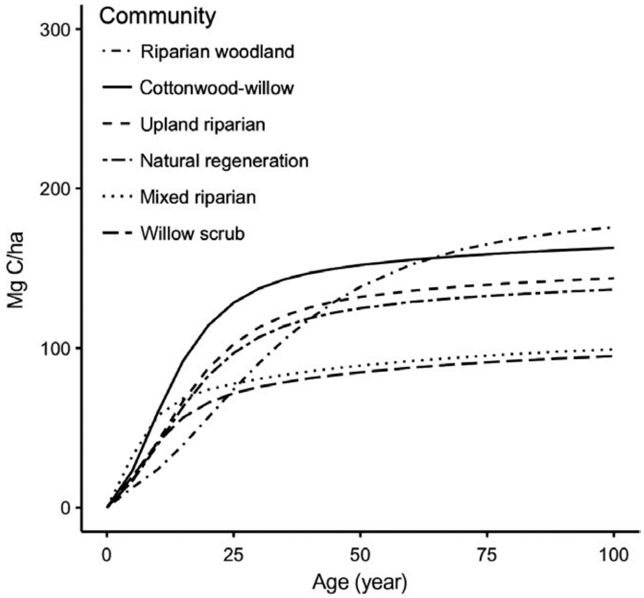

FIGURE 2 Comparison of total carbon sequestration in biomass and soil for different California riparian vegetation associations after the initiation of restoration

the corresponding figures for forest floor are 13%–23% at 30 years and 19%–38% at 100 years. Understorey represented ~3%–7% and 2%–5% of the total non-soil carbon at 30 and 100 years, respectively. Soil carbon accumulation, which was modelled identically for all forest types, accounted for a maximum of 9.8 Mg C/ha additional carbon at age 30 and 13.7 Mg C/ha at age 100 years. At a maximum, the additional soil carbon added during forest development constituted ~7%–13% of the total carbon from soil and non-soil components at age 30 and ~8%–14% at age 100 years.

With all pools combined, total carbon sequestration in riparian vegetation and soil varied from 75.7 to 137.4 Mg C/ha at 30 years post-restoration and from 95.1 to 175.8 Mg C/ha 100 years after stand initiation. Fast-growing vegetation types such as willow scrub and mixed riparian had the fastest initial growth rates and the smallest ultimate biomass; communities dominated by taller, more shade-tolerant trees developed more slowly but reached a higher maximum biomass (Figure 2).

4 | DISCUSSION

Riparian forests in California sequester a substantial amount of carbon, even compared to forests exploited for timber. For comparison's sake, we queried COLE 3.0 for the carbon density of other major timber species in California. Afforestation of California mixed conifer forests yields 37.0 Mg C/ha in all biomass and soil pools at age 30 and 166.8 Mg C/ha at age 100 years, while California Douglas fir (*Pseudotsuga menziesii*) forests yield 85.3 and 283.1 Mg C/ha, respectively. Afforested ponderosa pine (*Pinus ponderosa*) stands have a carbon density of 53.4 Mg C/ha at age 30 and 133.1 Mg C/ha at age 100 years, while the comparable figures for white fir (*Abies concolor*) are 51.0 and 166.0 Mg C/ha. Generally, riparian forests appear to sequester less carbon in the long term than commercially harvested coniferous forests, but can sequester as much or more in the short term (Figure 3). Values from COLE are produced by similar methodology to those in CREEC, with the notable exception that allometric conversion of DBH to biomass is performed in CREEC with the Chojnacky equations and then adjusted to align better with the component-ratio method used in COLE.

In seeking to compare CREEC's values to other biomass estimates for California riparian vegetation, we found little in the literature. The US Natural Resources Conservation Service (NRCS) publishes a tool, COMET-Planner, that estimates the carbon sequestration potential of replacing cropland or pastureland with riparian buffer strips. Typical COMET-Planner values for California counties predict annual sequestration of 0.67–1.45 Mg C/ha when restoring

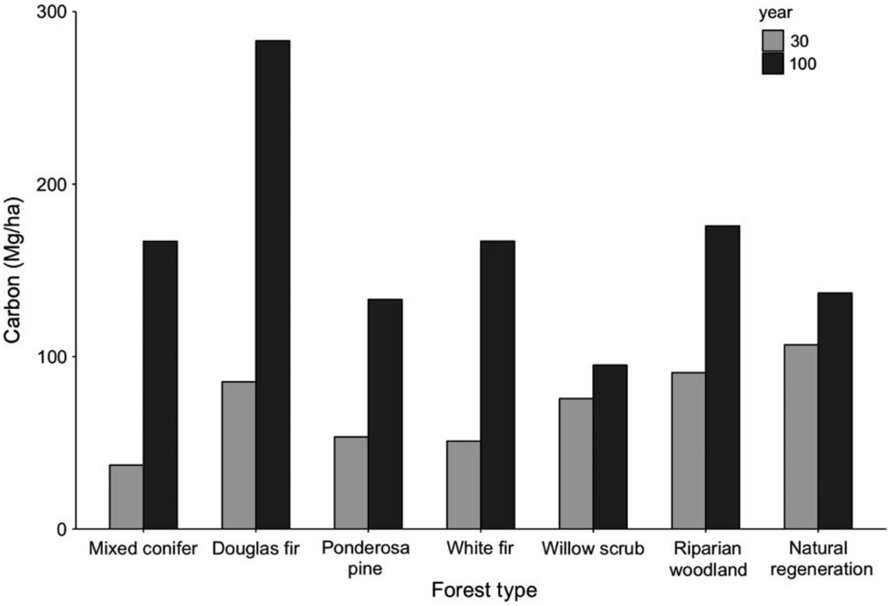

FIGURE 3 Comparison of selected riparian forest types (willow scrub, riparian woodland, natural regeneration) with selected California coniferous forest types, estimated at 30 and 100 years after stand initiation. Values are for total carbon, including biomass and soil. Riparian forest values are from this study, and coniferous forest values are from COLE 3.0 (Van Deusen & Heath, 2010)

degraded lands, which would yield 20.2–44.3 Mg C/ha at 30 years and 67.4–147.5 Mg C/ha at 100 years. These values are in a similar range to CREEC's estimates, but make an assumption of linearity that is inappropriate for the long-term carbon sequestration potential of these sites.

A 20-year chronosequence study by Matzek et al. (2015) that similarly expressed growth as a linear (not logistic) function predicted that tree carbon would increase at a rate of 3.25 Mg C/ha annually, resulting in an estimated 97.5 and 325 Mg C/ha in 30-year-old mixed and 100-year-old mixed riparian forests, respectively. (All plots from that study are included in our database.) Linear extrapolation beyond the time frame of the sampling likely results in increasingly unrealistic overestimates of biomass at later ages. We know of no other relevant peer-reviewed work from California riparian areas.

Estimates for riparian tree biomass from similar forests outside California also vary widely. A global meta-analysis on the topic divided 371 observations of regenerating riparian forests into relatively dry and relatively wet study regions and distinguished restored plantings from naturally regenerating vegetation (Dybala et al., unpubl data; data set archived at https://doi.org/10.5281/zenodo.1252510). This study fitted sigmoid growth models to the data and determined the asymptotes for maximum long-term carbon accumulation in vegetative biomass at 55 Mg C/ha (95% CI: 37–81) for dry area plantings; 73 Mg C/ha (95% CI: 55–99) for naturally regenerating stands in dry areas; and 150 Mg C/ha (95% CI: 93–252) for natural regeneration in relatively wet areas. A Mediterranean comparison comes from a study of riparian vegetation on the Paglia River in Italy, which recorded an average of ~45 Mg C/ha and a maximum of ~170 Mg C/ha in woody above-ground biomass (Cartisano et al., 2013). Comparable figures from semi-arid regions of the Australian sub-tropics include a mean value of 144.1 Mg C/ha in mature river red gum (*Eucalyptus camaldulensis*) forests (Smith & Reid, 2013), and a mean of 151.7 Mg C/ha in riparian zones of the Condamine River catchment (Maraseni & Mitchell, 2016).

Some carbon pools in CREEC are modelled, not measured. From the plots in Matzek et al. (2015), we have a small number of comparison points with empirical measurements of carbon accumulation in forest floor and in soil. Forest floor carbon accumulated at a rate of 0.282 Mg C·ha^{-1}year^{-1} in that study (95% CI: 0.23–0.877), which would result in 8.46 Mg C/ha at 30 years and 28.2 Mg C/ha at 100 years. The modelled value for forest floor carbon in our data set was 16.3 Mg C/ha at 30 years and 30.9 Mg C/ha at 100 years, i.e. comparable in range. These values may reflect a high contribution of fine woody debris to the forest floor layer from fast-growing, early-senescing species.

As for soil, in the Matzek et al. (2015) study, the baseline (i.e. depleted) soil carbon value was measured at 22.72 Mg C/ha and carbon accumulated at approximately 0.50 Mg C·ha^{-1}year^{-1} over the two decades represented in the chronosequence. Extrapolating out to 100 years would give a value 72.72 Mg C/ha in soil carbon. The global meta-analysis of Dybala and colleagues (unpubl data) has an even higher linear accumulation rate of approximately 0.78 Mg·C ha^{-1}year^{-1}, for the first ten years, above a baseline of 20.66 Mg C/ha. Linear rates of accumulation are often assumed for soil carbon, although some authors have suggested a saturating function is more appropriate (Stewart, Paustian, Conant, Plante, & Six, 2007). Our modelled outputs in CREEC assume that soil carbon must top out at the mean value set for Western hardwood forests. With that assumption, the initial depleted soil carbon value ranged from 42 to 53.2 Mg C/ha and recovered after 100 years to a value of only 56 Mg C/ha, substantially lower than the measured values cited above. These comparisons suggest that the soil carbon accumulation rates dictated by the method we adopted from the US greenhouse gas inventory reporting are underestimates compared to empirical observations of riparian soil carbon, which may show faster rates of linear accumulation and higher baseline carbon stocks.

A deficiency of our data set was its bias toward younger plots, especially for restored forests. About half of the plots were under 20 years of age, and 79% were younger than 50 years. The oldest restoration plot was 45 years old. This is an unavoidable symptom of the widespread habitat destruction of California's riparian forests (~95% in California's central valley; Katibah, 1984), the dynamism of floodplains and the relative recentness of efforts to restore them. The global meta-analysis of Dybala et al. (unpubl data) has no riparian inventory older than 50 years. A consequence of this shortcoming for our analysis is that growth functions may rise more swiftly to a lower maximum biomass than if we had data for older forests. Conversely, they may fail to account for a decline in overall biomass in later years due to senescence of pioneer trees. We modelled 100 years of accumulation because that is the standard for look-up tables in CARB protocols and because all forests reached their asymptote by that age, but the estimates are probably most accurate for the first 30 years of growth, where we have the highest data density.

It may be helpful to others working in this field to mention that we attempted to parameterize the Inland California and Southern Cascades variant of the Forest Vegetation Simulator (FVS) to create stand growth tables for different vegetation alliances, but abandoned it as unworkable for riparian forest. We expected to create species-specific growth curves for riparian species from age–DBH relationships in our forest plot database and use them to parameterize FVS. However, attributes of riparian forests, and structural features of the software, made it difficult to adapt FVS for this use. First, we found it was problematic to constrain age–DBH relationships for riparian species when trees occurred as both single-stemmed and multi-stemmed individuals. The difficulty was exacerbated by the fact that we had plot ages but not tree ages (i.e. from cores), so individual trees may have been younger, by an unknown amount, than their plot ages suggested (Irons, 2016). Second, woody shrubs are important canopy components in some riparian vegetation types, but FVS could not account for more than one shrub species in the understorey and could only model shrubs in the overstorey if they were considered as single-stemmed trees with exceptionally short maximum heights. Third, the FVS module most appropriate for habitat restoration, a bare-ground replanting scenario, had default values for initial tree growth that were highly inaccurate for riparian forests and could not be changed by the user. (None of these issues posed problems for the plot-based growth curve modelling we describe above.) In the end, we decided that using a model intended for timber yield in coniferous forests required too many unacceptable compromises when attempting to predict carbon stocks in willows and cottonwoods in mediterranean California.

Our lack of success with FVS highlights the fact that the data, methods and models required by GHG crediting programmes for predicting carbon stocks in forests reflect the knowledge base and methodologies of commercial forestry. Not only are the relevant data rare for forests not exploited for timber, but even the basic methods (e.g. using DBH to convert to biomass) may be of questionable value for species that do not grow single-trunked, straight and tall. Pulido, McCreary, Cañellas, McClaran, and Plieninger (2013) lamented that diameter–age relationships were poorly constrained in oaks, making it difficult to estimate stand age and predict changes in biomass accumulation over time. A recent statewide inventory of California wildlands concluded that the ecosystem in which most changes to carbon stocks were occurring – shrublands – was also the least well quantified (Gonzalez et al., 2015).

This paper serves as an estimate of carbon sequestration rates in riparian forest vegetation for California (and perhaps other mediterranean or semi-arid systems), and also as a roadmap for developing biomass-estimating tools in other carbon-crediting programmes, for other forest types. In considering how this work might continue, we conclude that there is a great need for improved allometric equations for non-timber species, as well as for vegetation surveys along chronosequences of restoration or succession, where the change in canopy cover can be related to shifts in biomass over time. Ground truthing remotely sensed stands with targeted biomass estimates would fulfill the same purpose. Another approach might be to assess the value of the state-and-transition models developed for rangelands (Briske, Fuhlendorf, & Smeins, 2005) when estimating carbon density in sparsely wooded savannas and woodlands. Instead of requiring many stand measurements at different ages, carbon density could be measured in forest canopies typical of various managed states, and carbon crediting could be applied to initiatives that move ecosystems from one state to another. In any case, we predict that vegetation surveys will be critical to the rapidly developing methodologies of carbon accounting and will need to be adapted for a variety of vegetation types.

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SUPPORTING INFORMATION

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

Appendix S1. Local coefficients for carbon component equations.
Appendix S2. Carbon look-up tables.
Appendix S3. Regional map.
Appendix S4. Detrended correspondence analysis plot.

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