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Carbon sequestration in riparian forests: A global synthesis and meta-analysis

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Abstract

Restoration of deforested and degraded landscapes is a globally recognized strategy to sequester carbon, improve ecological integrity, conserve biodiversity, and provide additional benefits to human health and well-being. Investment in riparian forest restoration has received relatively little attention, in part due to their relatively small spatial extent. Yet, riparian forest restoration may be a particularly valuable strategy because riparian forests have the potential for rapid carbon sequestration, are hotspots of biodiversity, and provide numerous valuable ecosystem services. To inform this strategy, we conducted a global synthesis and meta-analysis to identify general patterns of carbon stock accumulation in riparian forests. We compiled riparian biomass and soil carbon stock data from 117 publications, reports, and unpublished data sets. We then modeled the change in carbon stock as a function of vegetation age, considering effects of climate and whether or not the riparian forest had been actively planted. On average, our models predicted that the establishment of riparian forest will more than triple the baseline, unforested soil carbon stock, and that riparian forests hold on average 68–158 Mg C/ha in biomass at maturity, with the highest values in relatively warm and wet climates. We also found that actively planting riparian forest substantially jump-starts the biomass carbon accumulation, with initial growth rates more than double those of naturally regenerating riparian forest. Our results demonstrate that carbon sequestration should be considered a strong co-benefit of riparian restoration, and that increasing the pace and scale of riparian forest restoration may be a valuable investment providing both immediate carbon sequestration value and long-term ecosystem service returns.

KEYWORDS
carbon accounting, carbon credits, climate change, floodplain, multiple benefits, payment for ecosystem services, riparian restoration

1 | INTRODUCTION

Meeting the goal of the Paris climate accord to limit global warming to 2°C requires adopting multiple strategies for rapidly reducing and mitigating carbon emissions. These strategies include an array of negative emissions technologies that result in the net removal of greenhouse gases from the atmosphere, including the capture and storage of carbon in vegetation and soil through reforestation, afforestation, and changes in agricultural practices (Hansen et al., 2017; Smith et al., 2016). Restoration of degraded landscapes will also contribute to improving ecological integrity, which will provide many additional benefits to biodiversity and human well-being (IUCN, 2016). As a result, 56 countries have pledged to restore

168.4 million ha of deforested and degraded land through the Bonn Challenge, which will sequester an estimated 15.7 Gt CO_2 and generate $48.4 billion USD in economic activity (IUCN, 2018). The effectiveness of these efforts can be improved by information on how much and how quickly carbon can be stored in forest vegetation and soil.

Riparian forests, located along water channels, may be of particular importance to these efforts. Reference rates of carbon stock accumulation have been compiled for many forest types (e.g., IPCC, 2006), but these do not typically distinguish between riparian and upland forests. Despite their relatively small spatial footprint, riparian forests will usually have more favorable growing conditions (e.g., soil moisture), and they may accumulate carbon stocks at a greater rate than upland forests (Matzek, Stella, & Ropion, 2018; Naiman, Decamps, & McClain, 2010; Sutfin, Wohl, & Dwire, 2016), contributing more to rapid carbon sequestration in the short-term. Further, riparian ecosystems are widely recognized to provide numerous ecosystem services (Daigneault, Eppink, & Lee, 2017; Naiman et al., 2010; O'Brien et al., 2017), having the potential to mitigate the effects of climate change (Capon et al., 2013), and being biodiversity hotspots that provide critical habitat for fish and wildlife (Knopf, Johnson, Rich, Samson, & Szaro, 1988; Naiman et al., 2010). Because riparian ecosystems have been severely degraded worldwide (Nilsson & Berggren, 2000; Perry, Andersen, Reynolds, Nelson, & Shafroth, 2012; Zedler & Kercher, 2005), riparian forest restoration may be a valuable strategy for providing both rapid carbon sequestration value and long-term ecosystem services returns.

To inform this strategy, we conducted a global synthesis and meta-analysis of carbon stocks in riparian forests. Our first objective was to model general patterns of carbon stock accumulation in riparian forest biomass and soil as a function of vegetation age, including estimating the average annual rate of accumulation, time to equilibrium, and total carbon stock accumulation at equilibrium. These values would provide context for the magnitude and rate of carbon sequestration benefits from investment in riparian forest restoration. Our second objective was to test our hypothesis that these general patterns differ between riparian forests that were actively planted and those undergoing natural recruitment and succession. These results will help determine whether actively restored riparian forests can expect to sequester similar amounts of carbon in the biomass and soil.

2 | MATERIALS AND METHODS

2.1 | Data collection

We conducted a literature search in ISI Web of Science on November 29, 2016, using the search terms: "(riparian OR floodplain) AND (soil OR *forest* OR vegetation OR buffer* OR stand*) AND (biomass OR carbon) AND (pool* OR stock* OR stor* OR restor* OR sequest* OR accum* OR devel* OR productivity)." The search initially produced 1,806 publications, of which we focused on studies estimating the carbon stored in the soil or standing live and dead woody vegetation, or the total biomass of woody vegetation. We excluded studies estimating only rates of flux, including rates of sedimentation, erosion, soil respiration, litter-fall, or net ecosystem exchange, which were typically measured over relatively short time frames (<1 year). We also excluded studies focused solely on depressional or tidal wetlands, sampling units lacking woody vegetation, greenhouse experiments, or those that measured only the biomass or carbon content of individual plants. In addition to these published studies, we added data from four unpublished reports and data sets, for a total of 117 studies.

For each study, we recorded the location based on the reported or estimated coordinates of the study area's centroid, then assigned this location a unique ID, ensuring that multiple publications from the same study area received the same ID. Regional climate has a strong influence on carbon storage. Primary productivity, biomass, and soil carbon inputs are positively associated with warm, wet climates, and are limited in warm, dry climates, while warm temperatures are also associated with higher rates of decomposition and reduced soil carbon storage (Naiman et al., 2010; Sutfin et al., 2016). Thus, we expected carbon storage to be positively affected by annual precipitation and negatively affected by annual temperatures (Sutfin et al., 2016). We used the study area coordinates to extract the estimated 1970–2000 mean annual temperature and precipitation (Fick & Hijmans, 2017), and we used natural breaks in the distributions of these values at a mean annual temperature of 12.5°C and a mean annual precipitation of 1,400 mm to classify each study area as having relatively "cool" or "warm" and "wet" or "dry" climates, respectively. We also used the study location coordinates to assign each study area to one of four general Köppen–Geiger climate zones (Kottek, Grieser, Beck, Rudolf, & Rubel, 2006), which reflect a combination of temperature and precipitation: A (Tropical), B (Arid), C (Temperate), or D (Boreal).

There are several other factors in addition to climate that influence carbon storage in riparian ecosystems, including floodplain width, flow regime, frequency of inundation, and the presence of dams, diversions, and levees (reviewed in Sutfin et al., 2016). Carbon storage per unit area is expected to be higher in wide floodplains with complex channel forms, strong lateral hydrologic connectivity, and saturated soils, where accumulation of sediment and growth of a complex vegetation structure is promoted and decomposition is slow (Sutfin et al., 2016). Most studies did not provide these additional details, and hence, we could not incorporate these factors into our analyses to provide carbon stock estimates specific to each case. We assumed that the riparian carbon data within each climate classification included a range of geomorphic conditions and hydrologic connectivity, such that our estimates are representative of general patterns of biomass and soil carbon accumulation. Thus, individual riparian systems can be considered likely to be above or below these averages according to these additional factors.

For each study, we extracted any estimates of carbon stored per unit area in the soil or standing woody vegetation. Depending on the study's design, these included individual estimates for one or more distinct sampling units or plots, or mean estimates for groups

of sampling units, such as multiple plots within distinct successional stages. Where possible, we extracted individual estimates for each sampling unit rather than using mean values. For biomass carbon, we recorded any estimates of dry biomass or carbon stored per unit area (Mg/ha) in live and dead standing woody riparian vegetation and roots. We excluded any estimates of herbaceous vegetation, leaf litter, and lying dead biomass because most studies did not report these. Where separate estimates were reported for live and dead standing woody vegetation and/or roots, we added them together. For studies that reported only the dry biomass per unit area, we estimated the carbon fraction as 0.47 (IPCC, 2006).

For soil carbon, we recorded any estimates of soil carbon concentration ($C_\%$; %), bulk density (BD; g/cm^3), and carbon stock (C_{stock}; Mg/ha), as well as the depth of soil samples (d; cm). For studies that reported soil organic matter concentration or stocks, we estimated the carbon fraction as 0.58 (Poeplau et al., 2011). For studies that reported only three of the four metrics, we inferred the third:

$$C_{stock} = C_\% * BD * d \tag{1}$$

For studies that reported these metrics separately for multiple soil samples at successive depths in the same sampling unit, we calculated the total carbon stock and the weighted average bulk density and carbon concentration for each successive depth. However, to avoid including too many values from the same sampling unit, and because we anticipated differences in soil C accumulation rates with depth (Poeplau et al., 2011), we retained at most two samples from each sampling unit: the deepest sample up to 20 cm deep, representing surface soil, and the deepest sample >20 cm.

For each sampling unit, we also recorded whether or not it had been actively planted with riparian vegetation, and any information about the age of the vegetation, such as the number of years since planting or the estimated age of the stand of trees. Where only an age range was reported, we recorded the midpoint, and where only a minimum age was reported (e.g., >50), we recorded the minimum age. For sampling units described as "mature," "undisturbed," "remnant," or "reference" forest, we assumed a minimum age of 100 years unless reported otherwise. For biomass carbon stocks, we assumed a baseline of 0 Mg C/ha stored in woody vegetation at age = 0, but for soil carbon stocks we needed information on baseline soil conditions. Thus, we repeated the process of extracting soil carbon data for any paired sampling units that the authors considered to be representative of the baseline soil condition for an unforested sampling unit (age = 0). The baseline was usually an adjacent, disturbed sampling unit, such as cropland or pasture. In some cases, where no baseline was reported, we used the youngest, early successional sampling units ≤5 years old as representative of the baseline soil condition. We then calculated the age difference from the baseline.

We used WEBPLOTDIGITIZER (Rohatgi, 2015) to manually extract data from figures as necessary. We standardized all units of measurement for carbon stock (Mg/ha), soil carbon concentrations (%), and soil bulk density (g/cm^3).

2.2 | Biomass C stock growth models

To estimate the accumulation of carbon in riparian woody vegetation over time, we fit four alternative nonlinear growth models to the carbon stock estimates by age, including a two-parameter Von Bertalanffy model (Equation 2; Van Deusen & Heath, 2018), 3-parameter Chapman–Richards model (Equation 3; Raymond & McKenzie, 2013), and 3-parameter logistic (Equation 4) and Gompertz (Equation 5) models (Paine et al., 2012; Pinheiro & Bates, 2000):

$$C_t = K\left(1 - e^{rt}\right)^3 \tag{2}$$

$$C_t = K\left(1 - e^{rt}\right)^a \tag{3}$$

$$C_t = K\left(1 + e^{r(a-t)}\right)^{-1} \tag{4}$$

$$C_t = Ke^{ar^t} \tag{5}$$

where C_t is the carbon stock for a given vegetation age (t), K is the asymptote toward which the average carbon stock at equilibrium is approaching, and a and r are coefficients that determine the position and shape of the curve between 0 and the asymptote. For each of the growth models, we fit 13 alternative parameterizations with and without fixed effects on all model parameters (K, r, and a) of (1) whether or not planted (planted), (2–6) climate classifications, including either combinations of temperature and precipitation classifications or the Köppen–Geiger (KG) climate zones (temp; precip; temp + precip; temp * precip; KG), (7–11) planted combined with climate classifications (planted + precip; planted + temp; planted + temp + precip; planted + temp * precip; KG + planted), and (12) a null model with no fixed effects. We treated sampling units as our unit of replication, but to accommodate clustering of sampling units within study areas and variation among study areas due to differences in methodology, local climate, and species composition, we included random effects of study area ID on K in all models. To minimize heteroscedasticity, we log-transformed the C_t biomass carbon stock estimates (Paine et al., 2012).

2.3 | Soil C stock growth models

We used a similar approach to model the growth of riparian soil carbon stocks from paired baseline values, while accounting for changes in soil bulk density. Soil samples of the same depth (d) and volume but different bulk densities (BD) result in different soil masses sampled, and thus differences in estimated soil carbon stock even if the soil carbon concentration is the same (Equation 1). Thus, changes in carbon stock are more accurately estimated by comparing samples on an equal mass rather than equal-volume basis (VandenBygaart & Angers, 2006). We corrected for differences in soil bulk density between the riparian sampling unit (BD_t) and the paired baseline sample (BD_0) of the same depth by estimating the carbon stock on an equal mass basis ($C_{t, em}$) as:

$$C_{t,em} = C_t * \left(\frac{BD_0}{BD_t}\right) \tag{6}$$

where C_t is the reported, uncorrected carbon stock for the riparian sampling unit (Poeplau et al., 2011). We then calculated the change in carbon stock (ΔC_t) from baseline as a percentage of the baseline carbon stock (C_0):

$$\Delta C_t = \frac{(C_{t,em} - C_0)}{C_0} \tag{7}$$

where $\Delta C_t = 1$ represents a 100% increase in (or a doubling of) soil carbon stocks (Laganière, Angers, & Paré, 2010; Poeplau et al., 2011). This approach improves comparability of sampling units with different baseline carbon stocks and allows estimating relative growth rates of soil carbon stocks.

We fit the same nonlinear growth models to the change in soil carbon stock estimates, but due to the relative sparseness of the available soil data, we fit a smaller number of alternative parameterizations for each one: (1) whether or not planted (planted); (2–4) climate classifications (temp; precip; temp + precip); (5) an effect of whether the soil sample was shallow (≤20 cm) or deep (>20 cm; depth; and (6) a null model with no fixed effects. We again included in all models a random effect of study area ID on K to accommodate repeated samples in the same study area.

2.4 | Model fitting and inference

We fit all models in R using the package *nlme* (Pinheiro, Bates, DebRoy, & Sarkar, 2017). Within each set of models, we compared model fit using AICc scores. We evaluated the relative importance (RI) of each fixed effect by summing the Akaike weights of all models containing that parameter (Burnham & Anderson, 2002). We included all models with an evidence ratio (ER) >0.05 as competing models in the confidence set, and we used a model averaging approach to predict biomass and soil carbon stocks by age over the first 200 years. We then used a bootstrapping approach to estimate the 95% confidence intervals while accounting for multiple competing models. For each of 1,000 bootstrap iterations, we resampled the vegetation or soil data by study area ID as well as the residuals within each study area ID with replacement, refit the models, and calculated the model-averaged predicted values over a range of vegetation ages (Burnham & Anderson, 2002). Across all 1,000 iterations, we then calculated the 2.5 and 97.5 percentiles at each vegetation age. We also model-averaged estimates of K, the asymptote toward which the average carbon stock at equilibrium is approaching. As a measure of the uncertainty of K, we used the 95% bootstrapped confidence intervals at age = 200, which was well beyond the inflection point at which the growth curves began to flatten out and approach equilibrium. As measures of the rate at which carbon stocks accumulated in the biomass and soil, we calculated the average annual increase over years 1–10 and 10–50, as well as the total number of years required to accumulate 90% of K.

3 | RESULTS

Our literature review resulted in a total of 1,844 individual observations compiled from 117 publications, reports, or unpublished data sets collected from 103 unique study areas located around the world (Figure 1; Dybala, Matzek, Gardali, & Seavy, 2018). Over half of these study areas (51%) were located in North America. The majority of these studies (72%) presented data only for natural (i.e., unplanted) riparian forest, and nearly half (49%) only provided soil data. The earliest study included in the data set dates to 1987, with over half (52%) produced since 2011.

3.1 | Biomass carbon

We collected 491 observations from 53 study areas that reported biomass or carbon stock per unit area of riparian vegetation. The median carbon stock value reported was 63 Mg C/ha, with the highest estimates (318–487 Mg C/ha) coming from mature riparian forest over 150 years old located in the relatively cool and wet temperate rainforest of the Olympic Peninsula, Washington, USA (Balian & Naiman, 2005), and old-growth riparian forest of unknown age in the relatively warm and wet Malaysian Borneo (Singh, Malhi, & Bhagwat, 2015). We subset these data to 371 observations of known-age forests from 33 study areas. These included data from planted and unplanted forests in study areas with all combinations of relatively warm or cool, and wet or dry climates (Table 1). However, we did not locate any biomass data from planted forests in study areas with a relatively wet climate. Vegetation ages ranged up to 410 years old, but we did not locate data from any planted forests more than 50 years old.

Fitting growth models to the known-age biomass carbon stock data, we found the strongest support for the Chapman–Richards growth model with fixed effects of temp * precip + planted (Akaike weight = 0.56; Table 2). We calculated high relative importance values (RI) for the fixed effects planted (RI = 1.000), precip (0.996), and temp (0.690), but we were unable to detect strong differences between study areas by KG (RI < 0.001). Focusing first on the unplanted forests with naturally regenerated riparian vegetation, the model-averaged value for K was considerably higher in study areas with a relatively wet climate than a relatively dry climate (Figure 2). However, the effect of temperature interacted with precipitation such that the predicted biomass carbon stocks at maturity were highest in study areas with a warm and wet climate, and lowest in study areas with a warm and dry climate, where growth is more likely to be water-limited (Table 3).

In comparison with these naturally regenerating riparian forests, the model-averaged value of K for planted forests in study areas with a relatively dry climate was lower than their naturally regenerating counterparts, but with overlapping confidence intervals (Table 3). However, there were considerable differences in the rate of accumulation between planted and unplanted forests. Average annual growth rates over the first 10 years were more than double that of their naturally regenerating counterparts, reaching maturity
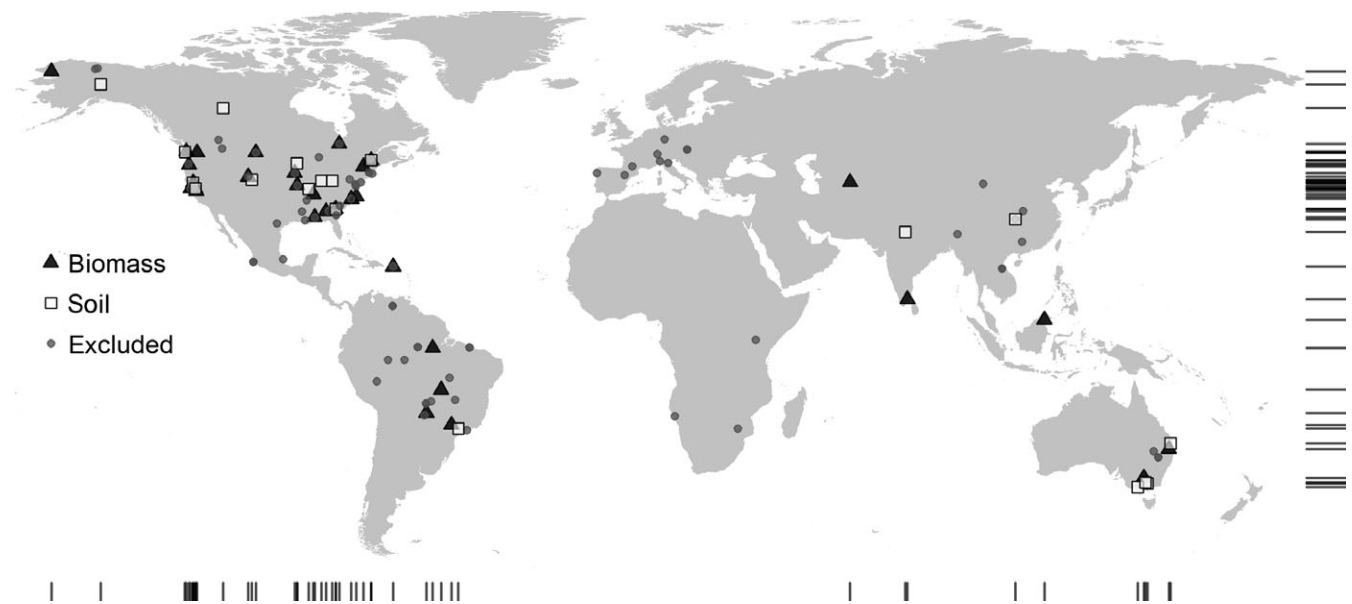
FIGURE 1 Coordinates of each study identified in the literature review, showing whether each contributed data on biomass (black triangles) or soil (white squares) carbon stocks in known-age riparian vegetation to the final analyses, or whether it did not provide sufficient information to be included in the final analyses (gray points). The rug plots in the margins indicate the relative density of the studies by latitude and longitude

TABLE 1 Summary of the biomass and soil carbon stock observations in riparian vegetation of known age included in the final analysis, grouped by mean annual temperature and precipitation classifications, and whether or not riparian vegetation was planted

Climate	Biomass C stock			Soil C stock		
	Unplanted	Planted	Total	Unplanted	Planted	Total
Warm & Wet	59	0	59	0	0	0
Cool & Wet	57	0	57	7	3	10
Warm & Dry	89	83	172	35	64	99
Cool & Dry	65	18	83	28	14	42
Grand Total	270	101	371	70	81	151

TABLE 2 Partial model selection results for growth in biomass and soil C stocks by vegetation age. Fixed effects include whether or not the riparian vegetation was planted (planted), and classifications for mean annual temperature (temp) and mean annual precipitation (precip) in the study area. Also shown are the number of parameters in each model (n), Akaike weight (w), and the evidence ratio (ER) relative to the top model. Only models with ER >0.05 are shown

Growth model	Fixed effects	n	ΔAICc[a]	w	ER
A. Biomass C stocks					
Chapman–Richards	temp * precip + planted	17	0.00	0.56	1.00
Chapman–Richards	precip + planted	11	1.24	0.30	0.54
Chapman–Richards	temp + precip + planted	14	3.96	0.08	0.14
Gompertz	temp * precip + planted	17	5.00	0.05	0.08
B. Soil C stocks					
Chapman–Richards		5	0.00	0.70	1.00
Gompertz		5	3.21	0.14	0.20

[a]Calculated from minimum AICc score of 730.03 for biomass C stocks and 373.92 for soil C stocks.

sooner and slowing over years 10–50 (Figure 2). Thus, while the estimated time to reach 90% of *K* was 65–90 years for unplanted forests in a wet climate, and 38–49 years for unplanted forests in a dry climate, it took only 14–15 years for planted forests in a dry climate (Table 3).

3.2 | Soil carbon

We collected 598 observations from 76 study areas from which riparian soil carbon stocks were reported or could be inferred from soil carbon concentrations and bulk density estimates. These included at most two samples from each sampling unit, one ≤20 cm

and one >20 cm deep. For these raw data, the median carbon stock was 35 Mg C/ha, with the highest soil C stock estimates (658–913 Mg C/ha) coming from reference forests at least 75 years old along the relatively warm and dry Savannah River, South Carolina, USA (Wigginton, Lockaby, & Trettin, 2000), an unplanted forested wetland of unknown age in Veracruz, Mexico (Campos et al., 2011),
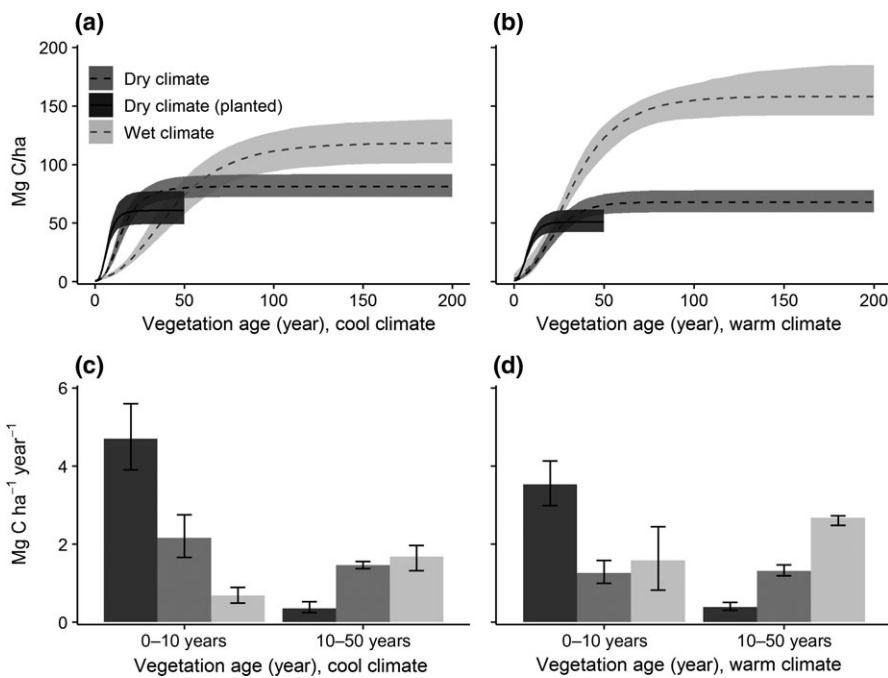
FIGURE 2 Model-averaged predicted growth in biomass carbon stocks by climate classification and whether or not planted, shown with 95% bootstrap confidence intervals. (a and b) Predicted values by vegetation age in study areas with (a) relatively cool climate and (b) relatively warm climate. Predicted values for planted forests are only shown for the first 50 years, because no biomass data were found for older planted forests. (c and d) Average annual rates of biomass accumulation over years 0–10 or 10–50 in study areas with (c) relatively cool climate and (d) relatively warm climate

TABLE 3 Model-averaged estimates of K, the asymptote toward which the average carbon stock at equilibrium is approaching, grouped by mean annual temperature and precipitation classifications, and whether or not riparian vegetation was planted. Biomass C stock values are reported in total Mg C/ha for standing live and dead woody vegetation, and soil C stock values are reported as the % increase over baseline. Also shown are the 95% bootstrap confidence intervals for the predicted value at age 200, and the ages by which 90% of K is expected to be accumulated

	Unplanted		Planted	
	K	Age at 90%	K	Age at 90%
A. Biomass C stock (Mg C/ha)				
Warm & Wet	158 (142–185)	65	–	
Cool & Wet	118 (101–139)	90	–	
Warm & Dry	68 (59–78)	49	51 (42–62)	15
Cool & Dry	81 (72–92)	38	61 (49–77)	14
B. Soil C stock (% increase)				
Warm & Wet	–		–	
All others	215 (187–244)	115	215 (187–244)	115

and a floodplain of unknown age in Brasilia, Brazil (Skorupa, Fay, Zinn, & Scheuber, 2013), both study areas with a relatively warm and wet climate. We subset these data to the 151 observations from forests of known age that were also reported with baseline soil carbon stocks and bulk density estimates. These included data from planted and unplanted forests in study areas from all but the warm

and wet combination of climate classifications (Table 1). Vegetation ages ranged up to 225, but we did not locate any vegetation data from planted forests in study areas with a relatively wet climate, or any planted forests more than 27 years old.

Fitting growth models to the known-age change in soil carbon stock data, we again found the strongest support for the Chapman–Richards growth model, but with no additional fixed effects (Akaike weight = 0.70; Table 2). We were unable to detect effects of precip, temp, planted, or depth, all of which were relatively unimportant (RI < 0.05). Thus, the model-averaged predicted values and annual accumulation rates for all combinations of these groups were identical (Figure 3). The model-averaged value for K was an increase of more than 200%, or more than triple the baseline soil carbon stock, and we estimated it would take 115 years to achieve 90% of this increase (Table 3). Thus, for a sampling unit with a baseline carbon stock of 20.66 Mg C/ha, equivalent to the median of the baseline samples included in this analysis, our results predicted a K of 68.1 Mg C/ha (95% CI at age 200: 56.9–79.3), with an average annual growth rate over the first 10 years of 0.78 Mg C/ha (95% CI: 0.52–1.03).

4 | DISCUSSION

Riparian ecosystems around the world have been severely degraded by anthropogenic activity, including altered flows from dams, levees, and water diversions, and conversion of riparian forests to urban and agricultural development (Nilsson & Berggren, 2000; Perry et al., 2012; Zedler & Kercher, 2005). These activities have resulted in the loss of ecological integrity and numerous ecosystem services. Recognition of these losses has inspired large-scale riparian restoration visions such as Brazil's Riparian Forest Restoration Project, which aims to reforest 1 million hectares of riparian rainforest in the state of São
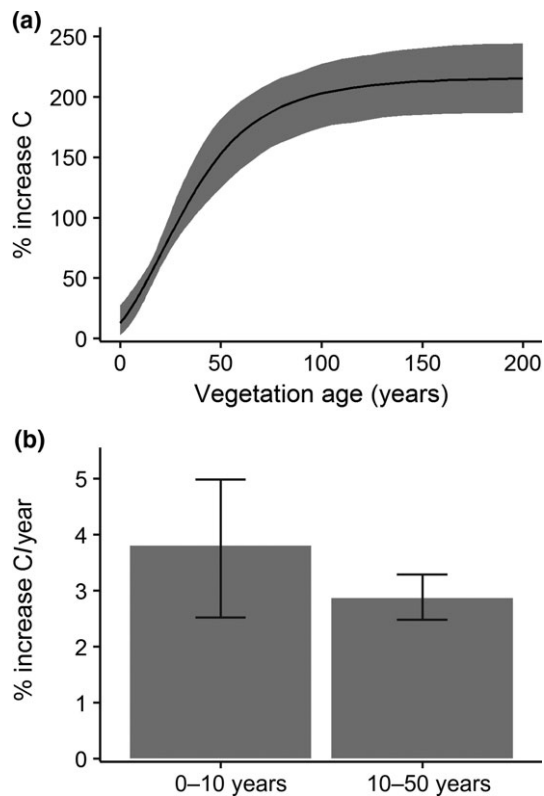
FIGURE 3 Model-averaged predicted increase in soil carbon stocks over baseline values, shown with 95% bootstrap confidence intervals. (a) Predicted increases by vegetation age. (b) Average annual increases over years 0–10 or 10–50

Paulo (Wuethrich, 2007), and California's Central Valley Joint Venture, which set a long-term goal of restoring more than 186,000 ha of temperate riparian forest (Dybala et al., 2017). There has been a widespread interest in quantifying carbon storage as a useful and monetizable co-benefit of riparian restoration, which could contribute to funding that would help increase the pace and scale of riparian forest restoration (Daigneault et al., 2017; Matzek, Puleston, & Gunn, 2015), and in turn, help reach global forest landscape restoration goals (IUCN, 2018). To support these efforts, our results provide the first global synthesis of riparian carbon data and meta-analysis of the growth rate of riparian carbon stocks, identifying general patterns of carbon sequestration specific to riparian forests.

Substantial amounts of carbon accumulate in the soil under riparian forest. Our modeling projected increases in soil carbon stocks by more than 200% over the baseline, unforested soil carbon stock (Figure 3), an increase that is substantially larger than the 0%–83% increases previously estimated for reforestation of cropland (Guo & Gifford, 2002; Laganière et al., 2010; Paul, Polglase, Nyakuengama, & Khanna, 2002; Poeplau et al., 2011; Stevens & van Wesemael, 2008). However, these previous estimates were not specific to riparian forests, and some of these studies estimated change in soil carbon over relatively short time spans (<30 years) and/or calculated averages without respect to vegetation age. Our results suggest that using these short-term studies as estimates of long-term change in

soil carbon would underestimate the total change in soil carbon, which may take over a century to stabilize. We estimated that the rate of change is relatively slow and would take 115 years to achieve 90% of this increase (Table 3), which is comparable to previous estimates of >100–120 years it would take to reach equilibrium (Poeplau et al., 2011; Stevens & van Wesemael, 2008).

Our modeling also projected remnant and naturally regenerating riparian forests to hold an average of 68–157 Mg C/ha in the biomass carbon at maturity, depending on climate (Table 3). These biomass carbon stock estimates are largely in agreement with previous summaries of riparian forest biomass (Naiman et al., 2010; Sutfin et al., 2016; Udawatta & Jose, 2011) and rival the highest estimates for any forest biome (Figure 4). Floodplains are estimated to cover only 0.5%–1% of the global land surface, up to an estimated 2 million km^2 (Sutfin et al., 2016). However, if all of these floodplains were forested, they would have the potential to store a total of 13.6–31.4 billion Mg C in woody biomass alone, or 2.9%–6.7% of the estimated 466 billion Mg C currently stored in vegetation worldwide (IPCC, 2000). Further, our results indicate that planting riparian forest significantly jump-starts the carbon accumulation process, with rapid growth of biomass carbon stocks in the first 10 years that may be able to immediately contribute to urgently needed negative emissions for climate change mitigation (Hansen et al., 2017).

Our results provide evidence that riparian forest restoration can provide a strong carbon storage benefit, in both the biomass and soil, but that the magnitude of this benefit varies with climate and restoration design. Recent research has pointed out a number of cases in which natural regeneration and recovery following disturbance can be more successful than active restoration, such as planting (Crouzeilles et al., 2017; Jones et al., 2018). Here, we found that, on average, planted riparian forest in study areas with a relatively dry climate would have a somewhat lower K for biomass carbon stock than their naturally regenerating counterparts (Figure 2), but we note that these estimates represent predicted averages. The successful establishment and growth of planted riparian forests may vary widely due to initial conditions and financial constraints (Chazdon, 2008), and particularly in areas with a relatively dry climate, whether there is hydrologic connectivity or irrigation (Friedman, Scott, & Lewis, 1995; Stromberg, 2001; Swenson, Whitener, & Eaton, 2003). While the naturally regenerating riparian forests in our data set were, by definition, in suitable locations that were capable of recruiting riparian forest vegetation, the planted riparian forests in our data set are more likely to include locations facing more challenging conditions (Reid, Fagan, & Zahawi, 2018). Thus, our data set is likely to include some planted forests with higher mortality rates, lower species diversity, and reduced ability to recruit new woody vegetation than naturally regenerating forests. These differences could explain the reduced average K of biomass carbon stocks in planted forests (Table 3). We expect that at locations with characteristics comparable to those of the naturally revegetating forests, planted forests should be able to match the biomass carbon stocks of naturally regenerating forests at maturity, while still exceeding the initial growth rates of naturally regenerating forests. Thus, where
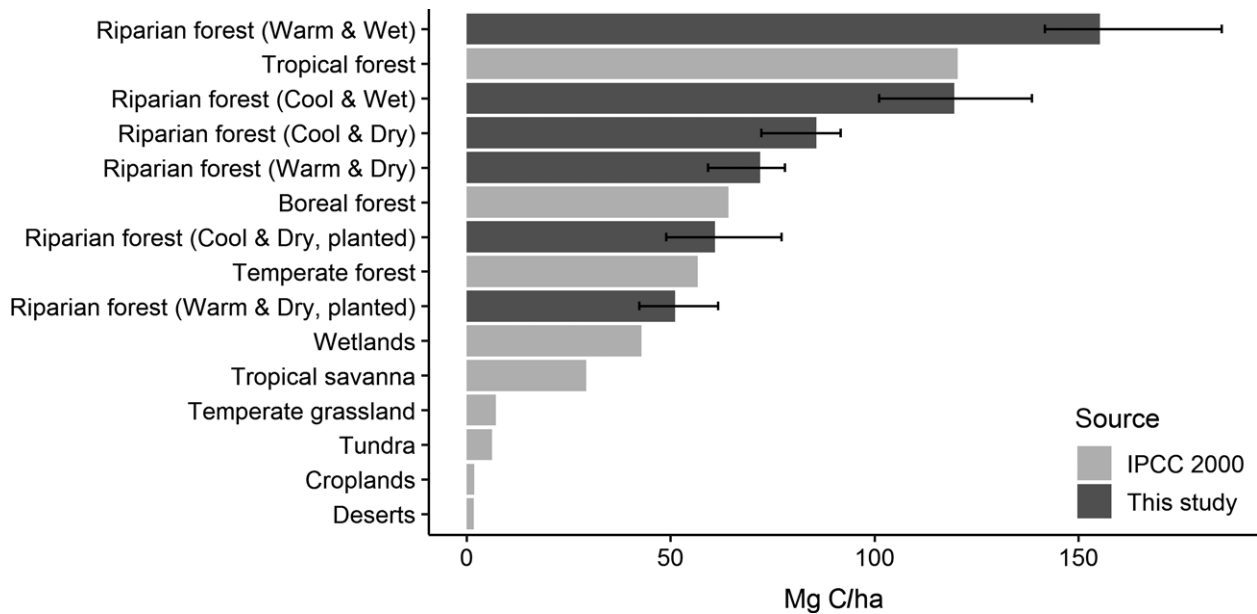
FIGURE 4 Estimated average biomass carbon stocks (Mg C/ha) by biome, adapted from IPCC (2000), shown with estimates of K from this study (Table 3) for comparison

rapid increases in biomass carbon stock are preferable or natural regeneration is unlikely to occur without intervention, planting of riparian forest is an effective strategy.

We also identified differences in riparian biomass carbon among climate classifications. Biomass carbon stocks and their growth rates were on average highest in study areas with a relatively warm and wet climate, where primary productivity is expected to be highest, and lowest in study areas with a relatively warm and dry climate, where growth may be limited by water availability (Figure 2). However, it is in these warm and dry climates that primary productivity and biomass are especially likely to be higher in riparian areas than adjacent upland areas (Sutfin et al., 2016). Thus, while the expected carbon storage benefits of any riparian restoration project should take these differences among climate classifications into consideration, riparian restoration is an effective strategy for storing carbon even in warm and dry climates.

We expected soil carbon stock growth to be influenced by climate, whether or not riparian vegetation was planted, and the depth of soil samples, but we were unable to detect strong differences. Previous work has identified differences in the effects of land use and land cover change on soil carbon stock by climate (Laganière et al., 2010) and soil depth (Paul, et al., 2002), and short-term decreases in soil carbon stocks have been attributed to soil disturbances caused by planting (Paul et al., 2002; Poeplau et al., 2011). A small number of the observations in our data set did show a reduction in soil carbon stocks compared to baseline estimates ($n = 25$), primarily in plots with relatively young vegetation (median: 19.5 years), but these included both planted and unplanted sampling units, and the reductions were small (median: −16%). Although the growth models we used structurally could not predict an initial negative growth rate, they also did not predict an initial delay in soil carbon stock growth, which would be expected if initial declines

were common. Instead, our models predicted an immediate rapid increase in soil carbon stocks (Figure 3), which suggests a strong positive change, on average. We were unable to detect any strong differences in soil carbon stock growth rates by climate classification or whether or not planted, which may be attributable to the relatively small number of total observations in our data set that provided all of the necessary information to be included in our modeling (Table 1).

Although we were able to identify general patterns of carbon accumulation in riparian biomass and soil from this global data set, our inferences are limited by the available data. We had to exclude numerous studies that did not provide the necessary information for inclusion in our modeling, limiting the number of observations per combination of climate classification and whether or not planted (Table 1). As a result of excluding these studies, our modeling was primarily based on studies from North and South America; there were no useable data from Africa or Europe and relatively few from Asia and Oceania (Figure 1). Further, biomass and soil carbon stocks are expected to be larger in riparian forests in wide, complex floodplains with frequent inundation (Sutfin et al., 2016), but we were also unable to account for these factors because this information was rarely reported for each study area. We assumed that within each climate classification, our data sets were representative of the range of riparian geomorphic and hydrologic conditions, but it is possible that our estimates are currently biased high or low due to these data gaps and geographical bias. For example, several estimates from riparian forests in Europe that could not be incorporated in this meta-analysis have reported large soil carbon stocks and substantially higher biomass carbon stocks than the averages presented here (Cabezas & Comín, 2010; Cierjacks et al., 2010; Grabe, Kleber, Hartmann, & Jahn, 2003). We encourage investment in future research quantifying riparian carbon stocks, particularly in under-sampled

regions, and we encourage researchers to clearly report riparian vegetation age, information about hydrologic connectivity, channel form, and floodplain width, and for soil carbon stocks, baseline estimates and soil sample masses. As the available biomass and soil carbon data for riparian forests grow, we will be able to further refine these estimates and better document the longer-term carbon storage value of planted forests, the influence of additional environmental factors, and the restoration design features contributing to more successful riparian forest restorations (e.g., Dybala, Steger, et al., 2018), particularly under changing climatic conditions.

While our results demonstrate that riparian forests are a net carbon sink, it is also important to consider the net climate change mitigation benefit of riparian forest restoration. For example, we were unable to distinguish between changes in soil carbon stock due to local carbon sequestration or the accumulation of carbon in sediments deposited by floods or filtered from run-off. The larger the proportion due to deposition, generally the smaller the net mitigation benefit of the riparian restoration, unless it will be slower to decompose in the riparian forest than it would have elsewhere (Berhe, Harte, Harden, & Torn, 2007). In addition, increases in the frequency of floodplain inundation associated with riparian forest restoration can result in more frequent anoxic soil conditions that temporarily increase methane (CH_4) and nitrous-oxide (N_2O) emissions. The magnitude of these increases depend on factors such as climate and depth to water table (Vidon, 2017), and the frequency and duration of inundation (Jacinthe, Bills, Tedesco, & Barr, 2012), and it varies widely on small spatial scales with topography, soil texture, and nutrient concentrations (Jacinthe, Vidon, Fisher, Liu, & Baker, 2015; Vidon, Marchese, Welsh, & McMillan, 2015). These "hot spots" and "hot moments" (McClain et al., 2003) of greenhouse gas emissions reduce the net mitigation benefit of riparian forest restoration. At the same time, the elimination of any greenhouse gas emissions associated with the previous land use or land cover type would increase the net mitigation benefit. Region-specific estimates of annual greenhouse gas emissions by land cover type could be used to estimate the net mitigation benefit (e.g., Jenkins, Murray, Kramer, & Faulkner, 2010) and inform payment for ecosystem services programs.

Riparian forest restoration is already widely recognized for providing multiple benefits, including (1) creating transition zones between water channels and adjacent land uses that can improve water quality and store flood water (Daigneault et al., 2017; Naiman et al., 2010; O'Brien et al., 2017); (2) enhancing fish and wildlife habitat (Dybala, Engilis, Trochet, Engilis, & Truan, 2018; Golet et al., 2008; Jansen & Robertson, 2001); and (3) providing recreational opportunities such as wildlife watching, fishing, and hunting that can help support local economies (Carver, 2013; Carver & Caudill, 2013; Golet et al., 2006). Our results demonstrate that riparian forests have a strong potential to contribute to carbon sequestration, which should be considered an additional co-benefit of riparian restoration. While the value of the carbon benefit alone may not be sufficient to induce landowners to embark on afforestation (Nelson & Matzek, 2016), the combined value of just a few of these multiple benefits can be significantly greater than implementation costs (Daigneault

et al., 2017; Holmes, Bergstrom, Huszar, Kask, & Orr, 2004), helping to increase the pace and scale of riparian forest restoration. By providing estimates of average carbon stocks in riparian forest biomass and soil, and demonstrating the rapid rate at which riparian forests can sequester carbon, our results suggest that investing in riparian forest restoration can be a valuable global strategy for contributing to urgent climate change mitigation goals as well as long-term biodiversity conservation and ecosystem services.

AUTHORS' CONTRIBUTIONS

KD and NS conceived the idea; KD, NS, and VM compiled the literature and data; KD compiled the database, analyzed the data, and led the writing of the manuscript. All authors contributed to designing the scope and content of the manuscript, revising the drafts, and gave final approval for publication.

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