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RESEARCH ARTICLE

Can carbon credits fund riparian forest restoration?

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Ecological restoration is increasingly called on to provide ecosystem services (ES) valuable to humans, as well as to benefit biodiversity and improve wildlife habitat. Where mechanisms to pay for ES exist, they may serve as incentives to embark on habitat restoration projects. We evaluated the potential of newly established carbon markets in the United States to incentivize afforestation along riparian corridors, by comparing the income earnable by carbon offset credits with the costs of planting, maintaining, and registering such a restoration project in California. We used a 20-year chronosequence of riparian forest sites along the Sacramento River as our model project. We found that carbon credits can repay more than 100% of costs after two decades of regrowth, if sufficient effort is put into sampling intensity in the first post-restoration decade. However, carbon credits alone are unlikely to entice landowners currently engaged in agricultural activities to switch from farming crops to farming habitat.

Key words: afforestation, allometric equations, carbon sequestration, ecosystem services, incentives, offsets, reforestation

Implications for Practice

- Practitioners performing restoration with woody species on lands in conservation ownership should consider designing projects to be compliant with protocols for earning carbon offset credits.
- If the protocol imposes a confidence deduction on earned credits, managers should expect restoration projects to earn few or no credits in the first decade, unless an intensive sampling regime is used to verify carbon stocks.
- It is unlikely that U.S. agricultural producers will shift from cultivating floodplain crops to doing riparian restoration purely in pursuit of carbon payments, but a combination of easement income with carbon credits can make afforestation more financially attractive.

Introduction

Habitat restoration is increasingly viewed as a way to protect or restore the flow of ecosystem services (ES) to humans (Aronson et al. 2007; Chazdon 2008; Benayas et al. 2009), in addition to its benefits to biodiversity. ES are the material and nonmaterial benefits humans derive from natural ecosystems, such as carbon sequestration, water purification, and recreational activities. Measuring, mapping, or modeling ES is becoming an essential step in deciding how to set conservation priorities and allocate resources for restoration (Sanchirico & Mumby 2009; Allan et al. 2013; Arkema et al. 2013). Meanwhile, ES goals are more and more frequently cited alongside traditional biodiversity goals as metrics of the success of conservation and restoration projects (Coen & Luckenbach 2000; Strange et al. 2002; Funk et al. 2014), although questions remain about whether restored habitats can provide the same quality and level of services as natural habitats (Benayas et al. 2009; Palmer & Filoso 2009).

It is often suggested that compensating landowners for provision of ES could incentivize habitat restoration (Clewell & Aronson 2006; Wu et al. 2011; Russell-Roy et al. 2014), or at least diversify the available funding options (Goldman et al. 2008). Formal mechanisms for compensation (payment for ecosystem services, or PES) are still in their infancy, however, and have been most prevalent in the developing world. A well-known example is South Africa's Working for Water program, which began by employing workers to remove alien weeds, and evolved into a PES program as the watershed benefits became more apparent (Turpie et al. 2008). The concept of making downstream water users pay for upstream watershed restoration has gained popularity in Latin America, with at least seven "water funds" becoming operational since 2000. The oldest of these, the Quito Water Fund, is revegetating watersheds at the rate of approximately 600 ha/year (Goldman-Benner et al. 2012). In the domain of carbon sequestration, several developing countries are shaping PES programs in response to REDD+, an initiative that rewards nations for reducing emissions from forest loss and degradation (Nepstad et al. 2013).

The industrialized world also makes use of PES mechanisms. One of the oldest is the U.S. Conservation Reserve Program (CRP), a land-retirement incentive that pays farmers

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to take marginal lands out of production and plant them with perennial cover. ES encouraged by the CRP include reduction of soil erosion, decreases in nutrient run-off to waterways, and habitat for game species. The CRP had more than 10.8 million hectares enrolled in 2013, paying out nearly US$1.7 billion in rental payments (FSA 2014), but CRP contracts are of relatively short duration (10–15 years). Permanent easements are, however, a feature of the newer Agricultural Conservation Easement Program (ACEP), instituted in 2014 and intended to both retire agricultural lands and subsidize the cost of restoration.

The other prospects for long-term afforestation funded by PES mechanisms in the United States are carbon offset programs associated with regulatory greenhouse gas cap-and-trade agreements, which now exist in the state of California and in the Northeastern/Mid-Atlantic region. These programs allow forest projects located in the 50 U.S. states, an area of 9.8 million km^2, to earn carbon credits for reforestation and then sell them to greenhouse gas emitters bound by the regulatory framework. To date, however, no afforestation or reforestation projects have been registered within these compliance markets, though several projects have been completed in the broader voluntary carbon offset marketplace. Buyers in the voluntary marketplace include corporate entities not included under regulatory caps, who are often more willing to invest in projects with environmental and social benefits beyond carbon sequestration (Peters-Stanley et al. 2012). As an example, the Walt Disney Company has invested more than US$20 million in forest carbon projects since 2009, including a large restoration project in the Angeles National Forest, an important provider of water to Los Angeles County. In the Southeastern United States, the private firm Green Trees arranges for farmers to combine CRP payments with carbon credits awarded through voluntary markets. Because the United States is excluded from international emissions trading as a nonsignatory of the Kyoto Protocol, the principal means for American landowners to earn forest carbon credits will be through regional compliance cap-and-trade markets or the voluntary marketplace, which have virtually identical carbon accounting protocols. Consequently, these markets have a potentially enormous impact on the future of forest habitat restoration in the United States.

As with any policy, though, the devil is in the details; whether income from carbon credits meaningfully reduces restoration costs to public agencies, or provides adequate incentive to induce private landowners to embark on afforestation efforts, will depend on how the program is structured. Compared with projects in developing countries, the U.S. PES mechanisms may be less successful at encouraging restoration because of higher costs in the developed world. For instance, in Australia, the cost to small landowners of embarking on reforestation projects was reckoned at more than double the value of carbon credits, even with the most generous economic assumptions (Hunt 2008).

To assess the feasibility of carbon credits as a restoration funding mechanism in the United States, we calculated the carbon credits earnable by a >20-year-old riparian forest restoration in California, accounting for both the credits earned from carbon storage and the costs of planting, maintaining, and verifying the eligibility of the forest project. We used a chronosequence approach, choosing sites that differed in age but were similar in soil type, climate, planting density, and species composition, to illustrate the trajectory and variability of carbon credit income.

Methods

Study Sites

All sites were planted in mixed riparian forest, composed largely of Fremont cottonwood (*Populus fremontii*), box elder (*Acer negundo*), Oregon ash (*Fraxinus latifolia*), and willows (*Salix lasiolepis*, *Salix exigua*, and *Salix gooddingii*), along with two understory shrubs, elderberry (*Sambucus mexicana*) and coyotebrush (*Baccharis pilularis*). All sites were formerly in agriculture, typically walnut orchards. Plantings were weeded and irrigated for the first 3 years and then left unmanaged (for details, see Alpert et al. 1999). Restored lands were generally acquired from willing sellers and owned by state and federal agencies; at present, land in U.S. federal ownership is exempted from earning carbon credits.

Sites were distributed over 100 km, from 40°N 05′02″, 122°W 05′45″ near Red Bluff, California, to 39°N 11′38″, 122°W 0′40″ near Colusa, California. All are on fine sandy loams of the Columbia series, classified as Oxyaquic Xerofluvents. The climate is Mediterranean, with a mean annual precipitation of 676 mm and mean annual temperature of 16.2°C. In June and July 2012, we sampled two to four sites in each of four restoration age classes that corresponded to approximately 5, 10, 15, and 20 years since planting (Table 1), plus a 3-year-old site where trees were just beginning to reach measurable heights. Within each site, three subplots of 750 m^2 were randomly located.

Inventory of Carbon Stocks

To inventory carbon stocks and account for carbon credits, we used protocols adopted by the State of California Air Resources Board (CARB 2013) and the Northeastern and Mid-Atlantic Regional Greenhouse Gas Initiative (RGGI 2013), which are identical with respect to determining credits in reforestation projects, and are hereafter referred to as "the Protocol." Sampling methods for measuring carbon stocks followed guidelines for California projects (Brown et al. 2004), described below.

Trees and Shrubs. In each subplot, all trees ≥5 cm diameter at breast height (dbh, 1.37 m) were measured for height and diameter. Allometric equations specified by the Protocol were used to convert dbh and height to aboveground oven-dry biomass. Belowground biomass density was calculated as a function of aboveground biomass density. For shrubs, elliptical volume measurements were used to calculate oven-dry biomass from species-specific allometrics (Smukler et al. 2010). See Appendix S1, Supporting Information, for equations used to estimate biomass.

Table 1. Pools of live and dead biomass carbon in each site and age class, in Mg/ha.

Site	*Year Planted*	*Age Class*	*Trees*	*Shrubs*	*Herbaceous*	*Standing Dead*	*Lying Dead*	*Forest Floor*	*Soil*	*Inventory*	*Included Pools*
Wilson Landing	2009	<5	1.73 ± 0.67	19.82 ± 17.76	2.27 ± 0.28	0	0	1.01 ± 0.43	25.36 ± 0.69	23.82 ± 18.02	49.18 ± 17.87
Capay	2007	5-yr	14.51 ± 4.00	16.16 ± 13.07	0.94 ± 0.35	0	0.48 ± 0.48	3.49 ± 0.95	24.11 ± 3.02	31.61 ± 17.21	55.72 ± 20.11
Deadman's Reach	2007	5-yr	5.82 ± 1.35	3.05 ± 0.50	2.88 ± 0.10	0	0.29 ± 0.29	3.09 ± 0.57	27.41 ± 0.95	11.75 ± 0.77	39.16 ± 0.34
Drumheller	2007	5-yr	7.96 ± 3.03	20.43 ± 13.80	3.50 ± 0.79	0	0	1.99 ± 0.31	22.23 ± 0.64	31.90 ± 12.78	54.14 ± 13.20
Pine Creek 5	2006	5-yr	7.49 ± 1.17	5.96 ± 2.43	1.36 ± 0.18	0	0.16 ± 0.16	2.54 ± 0.30	29.85 ± 3.54	14.81 ± 3.36	44.67 ± 4.95
5-yr mean			8.94 ± 1.51	11.40 ± 4.62	2.17 ± 0.37	0	0.23 ± 0.14	2.78 ± 0.30	25.90 ± 0.85	22.52 ± 5.41	48.42 ± 5.63
LaBarranca	2003	10-yr	27.26 ± 13.31	24.20 ± 11.69	0.55 ± 0.15	0.24 ± 0.24	0.99 ± 0.91	2.24 ± 0.65	29.77 ± 3.56	52.25 ± 19.40	82.01 ± 21.84
Ohm	2003	10-yr	34.74 ± 4.87	1.37 ± 0.62	0.50 ± 0.08	3.47 ± 3.47	0.33 ± 0.19	2.94 ± 0.57	30.28 ± 2.57	40.08 ± 2.98	70.35 ± 1.50
Moulton Weir	2002	10-yr	19.07 ± 5.19	2.68 ± 1.38	1.20 ± 0.67	0	0	3.00 ± 0.37	28.26 ± 2.76	22.96 ± 4.47	51.22 ± 3.50
Sul Norte	2002	10-yr	34.89 ± 16.49	2.92 ± 1.26	1.26 ± 0.43	0	0.40 ± 0.32	2.40 ± 0.92	36.59 ± 2.94	39.07 ± 18.02	75.66 ± 16.96
10-yr mean			28.99 ± 5.15	7.79 ± 3.82	0.88 ± 0.20	0.93 ± 0.87	0.43 ± 0.24	2.64 ± 0.30	31.22 ± 1.01	38.59 ± 6.47	69.81 ± 6.89
Flynn 498	1998	15-yr	59.04 ± 8.88	0	1.07 ± 0.14	0.13 ± 0.13	0	3.30 ± 1.57	28.86 ± 1.90	60.24 ± 8.92	89.10 ± 10.01
PCU2	1998	15-yr	32.64 ± 13.57	28.06 ± 9.18	0.70 ± 0.25	0	0.97 ± 0.91	2.52 ± 0.39	29.43 ± 3.05	61.39 ± 10.41	90.82 ± 13.13
Flynn 397	1997	15-yr	57.41 ± 14.06	12.34 ± 1.56	2.43 ± 0.91	0.07 ± 0.07	0.60 ± 0.60	3.85 ± 0.19	25.84 ± 1.49	72.26 ± ± 13.38	98.10 ± 12.03
Rio Vista	1997	15-yr	41.73 ± 7.77	7.21 ± 3.22	0.32 ± 0.11	0	0.45 ± 0.12	2.01 ± 0.06	31.96 ± 1.55	49.27 ± 9.12	81.22 ± 9.15
15-yr mean			47.70 ± 5.89	11.90 ± 3.75	1.13 ± 0.32	0.05 ± 0.04	0.50 ± 0.26	2.92 ± 0.41	29.02 ± 0.75	60.79 ± 5.15	89.81 ± 5.10
Princeton East	1993	20-yr	67.01 ± 13.01	0	0.33 ± 0.15	1.67 ± 0.68	1.48 ± 0.73	5.98 ± 0.80	36.16 ± 1.75	69.01 ± 12.51	105.17 ± 13.46
River Unit	1991	20-yr	74.59 ± 12.22	0	0.67 ± 0.15	1.25 ± 0.80	5.76 ± 3.90	4.37 ± 1.22	24.54 ± 2.94	76.51 ± 11.80	101.05 ± 12.41
20-yr mean			70.80 ± 8.16	0	0.50 ± 0.12	1.46 ± 0.48	3.62 ± 2.02	5.17 ± 0.75	31.20 ± 1.59	72.76 ± 7.87	103.11 ± 8.24

Values represent mean ± standard error of three subplots per site. "Inventory" is the sum of standing live and dead pools; "Included pools" adds soil carbon.

Herbaceous Understory. Within each subplot, three 0.5625-m^2 sampling plots were randomly located along a 50-m transect line, and all herbaceous vegetation was clipped to ground level and collected for drying at 55°C and weighing.

Standing Dead. Biomass was calculated as for live trees, according to the allometric equation for "unknown species," then reduced by a percentage based on the state of decay of the tree (Brown et al. 2004).

Soils. In each subplot, four soil samples were taken by coring to a depth of 30 cm at random locations along a 50-m transect line. One core was used for bulk density estimation, and the other three were subsampled, oven-dried, and ground for soil organic C (SOC) assays on an automated CN analyzer (NC2100, Brown University). To mitigate the confounding effect of soil bulk density on the carbon stock estimates, we calculated SOC stocks on an equal-mass rather than equal-volume basis, by figuring the mass of soil to 15 cm depth in the most densely compacted soil and comparing equal masses of soil from cores in all other plots.

Inventory Estimates. The inventory estimate is the sum of aboveground and belowground biomass for all standing live and dead pools, plus soil carbon (required when sites undergo extensive soil preparation at planting).

Baseline Assumptions

To earn credits, changes in carbon pools due to reforestation must exceed a baseline representing the changes in carbon pools that would occur without the reforestation. For our business-as-usual scenario, we assumed land had been abandoned after agricultural use and cleared. Natural regeneration of riparian forest on cleared sites is negligible over the time frame of this study, so we used a baseline of zero in counting biomass pools. For soil organic carbon, the baseline was represented by measurements from 15 soil cores of 30-cm depth taken from a site immediately after being prepared for restoration plantings.

Estimates of Credits Earned

To estimate carbon credits, we used Equation (1), modified from the Protocol to eliminate unneeded terms associated with timber harvests and salvage. We started the analysis at year 3 with measurements from our single 3-year-old site, then used age class averages to represent years 5, 10, 15, and 20. For the intervening years, we projected forward the growth of individual trees from each site using the Inland Cascades variant of the U.S. Forest Service Forest Vegetation Simulator (Keyser 2008). Biomass pools were assumed to be 50% carbon, and carbon stocks were converted to tons of CO_2 equivalent (CO_2e) when calculating credits.

$$\mathrm{QR}_y = \left(\Delta\mathrm{AC}_y - \Delta\mathrm{BC}_y + \mathrm{SE}_y\right) + N_{y-1} \qquad (1)$$

where QR_y = qualified emission reductions in year y, $\Delta\mathrm{AC}_y = \mathrm{AC}_y \times (1 - \mathrm{CD}_y) - \mathrm{AC}_{y-1} \times (1 - \mathrm{CD}_y)$, $\Delta\mathrm{BC}_y = \mathrm{BC}_y - \mathrm{BC}_{y-1}$, AC_y = actual carbon stocks in year y, BC_y = baseline carbon stocks in year y, CD_y = confidence deduction in year y, SE_y = secondary effect emissions in year y, and N_{y-1} = negative carryover from previous year.

Confidence deductions: If the 90% confidence limits lie within 5% of the inventory estimate, no confidence deduction is applied. However, if the confidence limits are 20% or more of the inventory estimate, the confidence deduction is 100%, with a prorated deduction for limits lying between 5 and 20%. Confidence deductions reduce credits in the year in which actual measurements were made and all prior modeled years since the last issued credit.

Secondary effects: These include direct GHG emissions from site preparation activities as well as the potential for "leakage," the shifting of GHG-emitting agricultural activities to newly cleared lands as a result of the project's existence. Secondary effects are calculated as follows:

$$\mathrm{SE}_y = \left(\mathrm{AS}_y + \mathrm{MC}\right) \qquad (2)$$

where $\mathrm{MC} = (-1 \times \mathrm{EF} \times \mathrm{PA})$, $\mathrm{AS}_y = (-1 \times L \times \Delta\mathrm{AC}_y - \Delta\mathrm{BC}_y)$, MC = mobile combustions from site preparation, EF = per-acre emissions factor, PS = project acreage, L = leakage risk percentage.

Leakage risk and mobile combustion emissions factors are default values for particular scenarios. We assume that the leakage associated with shifting cropland to new acreage is 0% because this agricultural land was abandoned prior to restoration, due to poor economic prospects. Mobile combustion emissions were reckoned at 0.174 metric tons of CO_2e per hectare because site preparation may have included stump removal. Mobile combustions associated with maintenance (e.g. weeding and planting) are excluded from consideration in the Protocol.

Credits must be adjusted for the risk of reversal, that is the risk that the carbon storage will not be permanent due to factors such as wildfire. We assume that the projects are covered by qualified conservation easements. The default risks associated with such projects amount to 12.4% of earned credits being sacrificed to a risk buffer pool, as a form of insurance against lost carbon storage.

Net Income Estimates

We used the California auction reserve price of a ton of CO_2e in April 2014 (US$11.34) to figure the income possible from riparian restoration. Installation costs (e.g. planting and maintenance) were derived from actual successful bids for comparable projects in the Sacramento River area (M. Cook 2014, River Partners, personal communication). The other costs—e.g. legal fees, verification, and transaction costs—were estimated from actual person-hours of forest sampling and from the use of a *pro forma* tool (Saah et al. 2012) developed for this purpose. Net income on a hypothetical 40-ha parcel was calculated as the difference between the project's nominal costs and nominal revenues from carbon payments.

Statistical Analyses

We used one-way ANOVA with age class as a random effect and sites nested within age class to characterize differences in carbon accumulation along the chronosequence; post hoc tests were performed with Tukey's honestly significant difference (HSD). Simple linear regression was used to approximate the rate of carbon increase over the chronosequence with actual ages used in place of age classes.

Results

Carbon storage is summarized by site and age class in Table 1. Generally, total live biomass increased over time, driven by increases in canopy tree biomass, and offset by decreases in understory (shrub and herbaceous) biomass that occurred as forests grew taller (Fig. 1). Meanwhile, storage in standing dead, lying dead, and forest floor biomass pools also increased as forests aged and experienced more mortality (Fig. 2). Soil carbon also increased over time, but significant gains in carbon storage compared with the baseline condition did not occur until after 15 years of forest growth (Fig. 3).

Carbon accumulated at a rate of approximately 3.25 Mg ha^{-1} yr^{-1} in trees (Fig. 4a) and 3.67 Mg ha^{-1} yr^{-1} in all included pools, which encompasses standing live, standing dead, and soil carbon pools (Fig. 4b). Total soil carbon in the 20-year age class amounted to 31.20 ± 1.59 Mg/ha (Table 1), or an accumulation of 8.48 Mg/ha above baseline stocks (Fig. 3). Standing live and dead biomass in the 20-year age class was 72.76 ± 7.87 Mg/ha, while forest floor and lying dead biomass, which are pools not considered in the calculation of carbon credits, were measured at 5.17 ± 0.75 and 3.62 ± 2.02 Mg/ha, respectively (Table 1).

For all age classes, the 90% confidence interval of the inventory estimate (standing live and standing dead) was large enough that it resulted in a 100% confidence deduction, meaning no carbon credits would be issuable. Based on the means and standard deviations for each age class, we estimated that the number of 750 m^2 plots required for avoiding the total loss of credits to the confidence deduction is approximately 47, 23, 6, and 5, for the 5-, 10-, 15-, and 20-year age classes, respectively.

To estimate the minimum sampling effort needed to avoid the confidence deduction, we modeled the effect on the variance of changing plot size and number while keeping sampled area fixed. We tried hypothetical plot sizes ranging from 225 m^2 (10 plots per site) to 2,250 m^2 (1 plot per site) and did 10,000 model runs to calculate the mean inventory estimate and standard error for each age class. Credits and costs were then recalculated under two hypothetical scenarios: "least effort," in which we assumed that the total area sampled at a site within an age class had occurred in a single subplot, and "optimal sampling," in which it was assumed that sampling had occurred using a number of plots that minimized the total sampling area

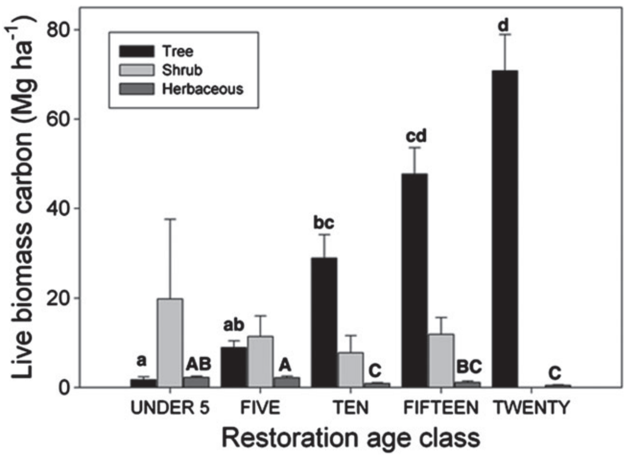

Figure 1. Changes in live biomass pools with increasing restoration age, in Mg/ha. Different letters indicate significant differences between age classes within a pool type.

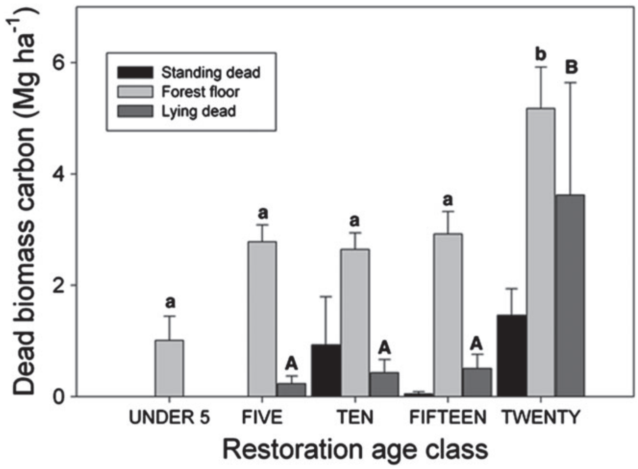

Figure 2. Changes in dead biomass pools with increasing restoration age, in Mg/ha. Different letters indicate significant differences between age classes within a pool type.

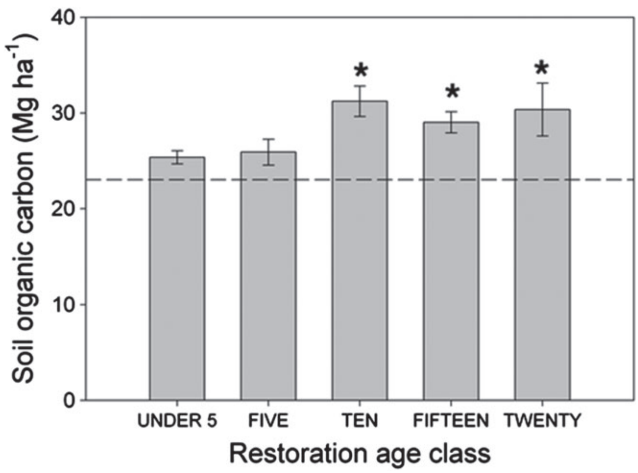

Figure 3. Soil carbon gains over restoration chronosequence, in Mg/h. The dotted line represents the baseline condition of comparable soils prior to planting. Asterisks indicate significant differences from the baseline.

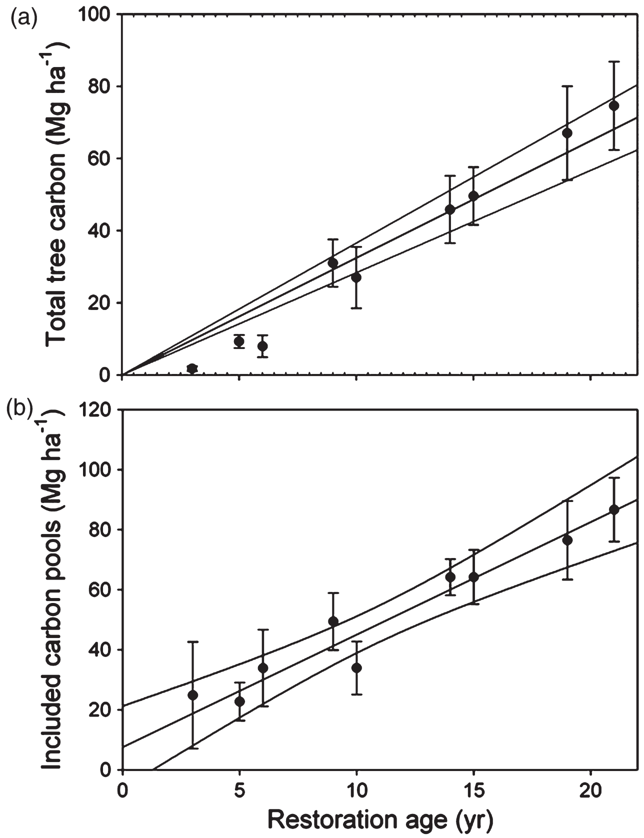

Figure 4. (a) Linear regression, with 95% confidence intervals, of tree carbon stocks in Mg/ha versus time, with line forced through origin, $y = 3.2455x$. (b) Linear regression, with 95% confidence intervals, of all included pools in Mg/ha versus time, $y = 3.6688x + 10.3616$. Included pools equal standing live and standing dead biomass, plus soil carbon.

necessary to avoid the confidence deduction entirely in each age class.

In the "least effort" scenario, credits were only earned after year 10, with confidence deductions of 7.7% in years 11–15 and 3.5% in years 16–20. After adjustments for secondary emissions and contributions to the risk buffer pool, 1 ha of restored forest earned credits equivalent to 127.5 tons of CO_2e, valued at US$1,446. Costs associated with this scenario were US$1,823 ha^{-1} for initial planning, installation, and maintenance of the forest plantings, and US$204 for registration and verification of the credits over 20 years. The percentage of restoration costs returned by carbon credits was 71.3% in this scenario. To pay back 100% of costs, the price of carbon would need to rise about 40%, to US$15.90.

In the "optimal sampling" scenario, the total number of credits earned was 259.5 tons of CO_2e/ha after 20 years of restoration, for a monetary value of US$2,943. Costs associated with this scenario were higher due to the extra sampling effort and were estimated at US$2,695 ha^{-1}, of which US$872 was associated with registration and verification and the remainder with planting and maintenance. Restoration therefore recoups 109% of costs when no confidence deduction is imposed.

Discussion

Riparian forest habitat in California has suffered the loss of more than 95% of its former area due to floodplain development and agriculture (Katibah 1984). Since the early 1990s, more than 2,500 ha along the Middle Sacramento River have been planted with riparian vegetation, at a cost of more than US$30 million (Golet et al. 2013). By our estimates, carbon credits have the potential to offset most or all of the cost of riparian forest plantings by the end of their second decade, providing a substantial benefit to public agencies that undertake restoration on lands protected by conservation easement or public ownership. We consider this estimate conservative because our chronosequence approach probably yields higher variability in forest biomass than would be true if we had measured a single forest over time, and this variability either reduces income (through the confidence deduction) or increases costs (due to the need for additional sampling).

We compared our tree carbon estimates to U.S. Forest Service Forest Inventory and Analysis (FIA) plots for elm-, ash-, cottonwood-, and willow-dominated riparian forests in California (USDA 2014). The FIA plots showed an average annual increase of 3.93 Mg of above- and below-ground tree carbon per year in plots measured at 10-year intervals, similar to our result of 3.25 Mg/ha annual increase. Standing tree biomass was estimated to contain 78.21 ± 30.47 Mg/ha of carbon, also close to our estimate of 70.80 ± 8.16 Mg/ha at 20 years of age. We are therefore confident in our estimate that riparian forests in the Western United States accumulate carbon at a rate fast enough to be economically advantageous to public agencies seeking to offset restoration costs.

However, payments for carbon sequestration are much less likely to induce private landowners to switch to habitat restoration from agricultural activities. Growers in the Sacramento River floodplain typically grow almonds or walnuts, two of the state's most lucrative crops. For almonds, the cost to establish an orchard on land already owned by the farmer is about US$2,047/ha before the first harvest, similar to the US$1,823/ha associated with habitat restoration plantings. By year 20, though, the accumulated net income of almonds, having repaid the amortized costs of planting by year 13, comes out to US$2,273/ha, and this total rises to US$3,580/ha by year 25, when the orchard rotation ends and is replanted (Freeman et al. 2008). For walnuts, the establishment cost is US$3,391/ha before the first harvest, but at age 20 walnut plantations have long since broken even on planting costs, and earned US$5,251/ha in net income, with a sum of US$15,097/ha expected by the end of the rotation at age 35 (Krueger et al. 2012). Considering that earned carbon credits are decreased by 24% in the Protocol for leakage risk when land is removed from agricultural use, resulting in net income from carbon plantings of US$248/ha after 20 years, it is clear that farming carbon compares very unfavorably to farming crops in this system.

We considered whether three other variables could make the arithmetic more favorable for shifting agricultural production into habitat. First is the potential for the conservation easement, performed to protect the perpetuity of the carbon credits, to itself be a source of income. Easement values are difficult to estimate because each easement is an individual agreement between landowner and purchaser. However, a study of 14 land trusts in California suggested that the majority of agricultural easements are purchased for 40–60% of the land's assessed value (Lassner 1998). Assuming that an easement is worth 50% of assessed value and that land suitable for walnut cultivation in the Sacramento River valley is assessed at approximately US$2,850/ha (Krueger et al. 2012), one-time income from a conservation easement on a 40-ha parcel converted to habitat would be US$57,000 or US$1,425/ha. For private landowners, easement income would therefore constitute the vast majority of the income potential from restoration. Combining an easement purchase with a 75% subsidy of restoration costs, possible under the new ACEP program, would make restoration more economically competitive with farming.

Second, we asked if habitat restoration would be more financially attractive if agriculture was forced to reduce or mitigate its GHG emissions under the cap-and-trade system. Currently in California, caps on emissions apply only to energy industries (e.g. power plants) and a few other high-emission sectors (e.g. cement production) that are responsible for more than 25,000 tons of CO_2e annually. Based on a recent life cycle assessment of almond production (E. Marvinney, UC Davis, personal communication), each acre of cultivated almonds is responsible for 2.95 tons/ha of CO_2e emissions over the lifespan of an orchard, which would require the purchase of US$34/ha in offset credits for each 25-year rotation. So this penalty does not loom large as a potential future incentive for restoration on active agricultural land.

Third, we considered whether conservation measures embraced by public agencies but less likely to be of concern to the private landowner, such as the planting of elderberry shrubs to provide habitat for the threatened Valley elderberry longhorn beetle, reduced net carbon credit income. Although our sites are similar in productivity to bottomland hardwood forests of the sort being profitably planted for carbon credits in the southeastern United States (Schoch et al. 2009), our planting costs are as much as 50% higher (Frey et al. 2010). One contributing factor is irrigation, a necessity in the arid West, which increases costs by 15%. As for biodiversity, we found that shrubs contributed C storage in proportion to their planting density, suggesting no sacrifice of carbon credits to the imperiled beetle. However, pure cottonwood stands can accumulate carbon at three times the rate we observe here (DeBell et al. 1996); and revegetating with a highly diverse mixture of species, including understory herbs, likely increases planting costs.

Perhaps the optimal way to make carbon credits work to fund restoration activities on private lands is to relieve the small landowner of the costs associated with measurement, reporting, and verification, by developing a less formal, but still credible, offsets marketplace. For example, the Clear Water Carbon fund, established in the northeastern United States in 2011, uses a voluntary carbon offset marketplace to fund riparian forest restoration projects but stays cost-effective by stopping short of formally registering the offset credits (Wilkerson & Gunn 2013). Such a program relies on the reputation of the organizations involved in restoration to assure offset buyers that the funds support meaningful projects. As we demonstrated, the requirement that forest carbon be measured directly (rather than indirectly, e.g. via remote sensing), along with the high bar for statistical confidence, can increase costs prohibitively at small scales.

In sum, our analysis shows that, under the major U.S. forest carbon PES mechanism, carbon prices are sufficient to repay the costs to public agencies of restoration on land in conservation ownership. Investing resources in intensive sampling in the early years of forest regrowth is critical to avoiding the confidence deduction and earning the full value of carbon credits. However, significant additional incentives, or a rise in the price of carbon, will be required to coax landowners out of viable agricultural activities along floodplains.

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Supporting Information

The following information may be found in the online version of this article:

Appendix S1. Allometric equations used for above- and below-ground tree and shrub biomass.

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