RESEARCH ARTICLE

Development of Vegetation and Aquatic Habitat in Restored Riparian Sites of California's North Coast Rangelands

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Abstract

The preponderance of short-term objectives and lack of systematic monitoring of restoration projects limits opportunities to learn from past experience and improve future restoration efforts. We conducted a retrospective, cross-sectional survey of 89 riparian revegetation sites and 13 nonrestored sites. We evaluated 36 restoration metrics at each site and used project age (0–39 years) to quantify plant community and aquatic habitat trajectories with a maximum likelihood model selection approach to compare linear and polynomial relationships. We found significant correlations with project age for 16 of 21 riparian vegetation, and 11 of 15 aquatic habitat attributes. Our results indicated improvements in multiple ecosystem services and watershed functions such as diversity, sedimentation, carbon sequestration, and available habitat. Ten riparian vegetation metrics, including native tree and exotic shrub density, increased nonlinearly with project age, while litter and native shrub density increased linearly. Species richness and cover of annual plants declined over time. Improvements in aquatic habitat metrics, such as increasing pool depth and decreasing bankfull width-to-depth ratio, indicated potentially improved anadromous fish habitats at restored sites. We hypothesize that certain instream metrics did not improve because of spatial and/or temporal limitations of riparian vegetation to affect aquatic habitat. Restoration managers should be prepared to maintain or enhance understory diversity by controlling exotic shrubs or planting shade-tolerant native species as much as 10 years after revegetation.

Key words: site-specific riparian revegetation, trajectory analysis, restoration monitoring, regional assessment, post-project appraisal.

Introduction

Revegetation is a common tool to restore riparian areas for many reasons, often by excluding livestock and/or planting native trees. The number of river and stream restoration projects in the United States has steadily increased since the 1980s from 100 to over 4,000 projects per year (Bernhardt et al. 2005; Palmer et al. 2007). In California, over $2 billion was spent on river restoration since 1980 with riparian management the most common project type (Kondolf et al. 2007), but there has been limited systematic documentation of project effectiveness to provide quality habitat and watershed functions (Kondolf et al. 2007; Miller & Hobbs 2007; Palmer et al. 2007).

Evaluation of previously restored sites has provided valuable feedback for understanding riparian habitat response to various stream rehabilitation practices (Frissell & Nawa 1992; Opperman & Merenlender 2004; Tompkins & Kondolf 2007). Numerous studies quantified riparian vegetation recovery (Platts 1981; Kauffman et al. 1997; Opperman & Merenlender 2000) and indirect recovery of aquatic habitat has followed woody riparian vegetation establishment (Hupp & Osterkamp 1996; Opperman & Merenlender 2004; Corenblit et al. 2007). Restored project sites offer opportunities to learn about resulting community structure and ecosystem processes beyond static endpoints provided by reference sites (Parker 1997); however, long-term research over multiple decades has been limited to case studies unable to quantify regional variability or unintended consequences in a holistic evaluation.

Some have used the amount of time since project implementation in various forms of trajectory analysis to provide timelines for achieving specific objectives (Zedler & Callaway 1999; Golet et al. 2008). Watershed management carries the

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expectation that certain important societal objectives will be achieved over time as a result of vegetation interacting with physical processes (e.g., stochastic flood events transporting sediment and pollutants). Examples of these objectives include diversity (Hobbs 1993; Hupp & Osterkamp 1996), pollination (Kremen et al. 2004), sedimentation (Hupp & Osterkamp 1996; Corenblit et al. 2007), trophic dynamics (Baxter et al. 2005; Muotka & Syrjanen 2007), carbon storage (USDA 2000; Bush 2008), nutrient cycling (Kauffman et al. 2004; Sheibley et al. 2006; Bush 2008), water quality (Phillips 1989; Peterson et al. 2001; Houlahan & Findlay 2004), infiltration (Kauffman et al. 2004), flood retention (Hupp & Osterkamp 1996; Corenblit et al. 2007), available habitat (Dobkin et al. 1998; Opperman & Merenlender 2004), and habitat use (Dobkin et al. 1998; Golet et al. 2008). However, the trajectory analysis has not been applied to watershed management in a holistic approach using numerous attributes to assess the recovery of multiple ecosystem services (Kremen 2005).

We conducted a retrospective, cross-sectional survey (i.e., chronosequence) of site-specific riparian revegetation projects in three northern California coastal counties. Riparian vegetation and aquatic habitat response to stream rehabilitation was quantified in a trajectory analysis using regression relationships with project age for 36 restoration metrics at 102 sites to provide a holistic regional evaluation of long-term success over multiple decades. We used these trajectories to infer changes in ecosystem services and watershed functions (Black 1997) provided by riparian restoration.

Methods

Project Identification

Riparian revegetation sites were located in the mixed oak woodland and annual grassland of California's north coast. The region has a Mediterranean climate with cool wet winters and hot dry summers. However, this coastal region of California is cooler with more moderate rainfall than most hardwood rangelands. During the study period, mean annual precipitation in the study area was 1,019 mm (range $= 679 - 1,629$ mm) and mean annual temperatures were $13.7°C$ (range $= 12.0 - 15.1°C$). Streams and rivers in the region are dominated by varying degrees of channel incision (Darby & Simon 1999) and are located in watersheds with an average area of 23.5 km^2 (range $= 0.2 - 133.1$ km^2), elevation of 145.3 m asl (range $= 3.7 - 656.4$ m asl), and 21.9% forested (range $= 0 - 100$%).

We surveyed 102 sites in Marin, Mendocino, and Sonoma Counties (Fig. 1). Sites were selected in collaboration with consultants, agencies, and landowners, whose permission was solicited for access to conduct surveys. Project cooperators identified both "successful" and "unsuccessful" projects to be included in the study. Site selection focused on projects with documented implementation dates in alluvial stream reaches of willow and mixed oak woodland vegetation with few trees present prior to project installation (e.g., Fig. 2a). Surveyed project sites were primarily on second- and third-order streams with a range in project age from 4 to 39 years since restoration.

Revegetation design at surveyed projects ($n = 89$) was site-specific and focused on establishing *Salix* species to "jump start" recovery of riparian forests to control erosion and sustain multiple watershed functions (Kauffman et al.
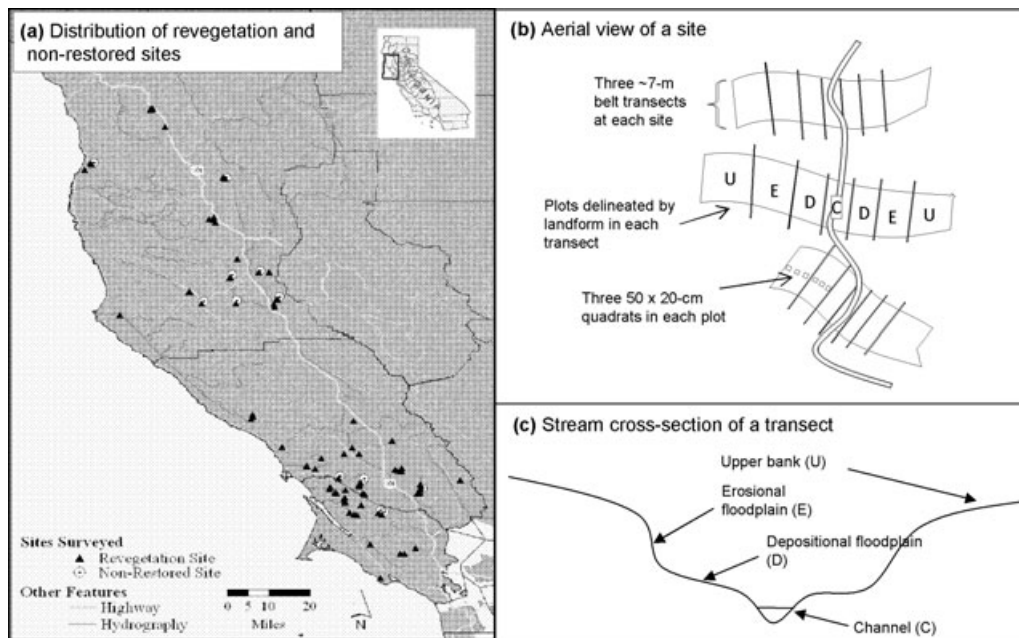

Figure 1. The three county study areas north of San Francisco Bay including locations of restored and nonrestored survey sites (image courtesy of Sonoma County GIS Central) a). Aerial view of an idealized survey site depicting belt transects, plots delineated by landform, and herbaceous quadrats b). Stream channel cross-section showing landforms along a transect c).

Figure 2. Photographic time-series of an example project site on a tributary to Walker Creek in Marin County, documenting vegetation response at 0 a), 2 b), 8 c), and 12 years d) since restoration occurred (images courtesy of Marin Resource Conservation District).

1997). The methods utilized were often implemented as combinations of practices including tree or shrub planting with dormant willow posts or container plants (Johnson 2003), biotechnical bank stabilization (Johnson 2003; Flosi et al. 2004), and passive restoration (Kauffman et al. 1997) using large herbivore management (e.g., removal, reduced stocking rate, or exclusionary fencing for livestock and/or deer). Nonrestored sites were surveyed ($n = 13$) where local experts indicated that a particular stream reach had vegetation similar in structure to the project site before revegetation occurred.

Site Characterization

We characterized riparian forest and aquatic habitats at riparian restoration project sites using 36 ecological attributes collected at 5 nested spatial scales: (1) site ($n = 102$, Fig. 1a), (2) belt transect ($n = 3$ per site, Fig. 1b), (3) landform class ($n = 4$ per transect, Fig. 1c), (4) plot ($n = 2+$ per landform), and (5) quadrat ($n = 3$ per plot, Fig. 1b). Landform classes were delineated by channel morphology and depositional or erosional features adapted from Harris (1987, 1999). Specifically, we used the lowest observed bankfull location and flood-prone elevation ($2 \times$ bankfull depth) described by Rosgen (1996) to delineate plots in the active floodplain. The final plot sampled on each bank extended from the top of the bank to the fence or field edge, and included alluvial valley, terrace, or upland hillside geomorphic features. This landform-based approach to collecting vegetation data allowed for comparable results to be analyzed from various types of stream channels.

At the site scale, data collected included *small woody debris* (diameter < 12 in), *large woody debris* (diameter > 12 in), and *aggregate woody debris* (debris jam clumps of 4 or more pieces) counted within the bankfull channel (Flosi et al. 2004). Pool characteristics assessed were *mean pool depth*, *maximum pool depth*, *pool frequency* and percent *pool habitat type* (Flosi et al. 2004). We collected stream substrate data at each site and calculated percent *fine sediment* and *embeddedness* (Flosi et al. 2004). The linear distance of riparian shade over the thalweg was recorded at intervals with a hip chain as *linear channel canopy*.

We placed three cross-sections and transects perpendicular to the channel stratified within each site at fast-water riffle locations. Stream width and depth were measured and documented as *bankfull width-to-depth ratio* (Rosgen 1996). *Streambank stability* was assessed for both banks at each cross-section according to Platts et al. (1987) and *bank angle* was measured using a clinometer. *Canopy density* was measured with a spherical densiometer following California Department of Fish and Game protocols (Flosi et al. 2004) and solar radiation was measured with a solar pathfinder by using the month of August to standardize values before calculating *intercepted*

solar radiation (Platts et al. 1987). Both measurements were taken from the thalweg at each cross-section.

Data gathered within each plot included *woody vegetation density* (trees > 1 m) and *canopy cover*. Species identification followed Hickman (1993). Herbaceous vegetation cover was estimated using a modified Daubenmire Frame (20 × 50 cm) to stratify quadrats equidistant in each plot perpendicular to the stream channel (BLM 1996). The metric *ground cover* included the sum of litter, vegetation, and stone cover (BLM 1996). Relative cover was calculated for six herbaceous functional groups. Documenting survival was not possible because of the lack of consistent record keeping on specific numbers of plant species installed during the restoration project and difficulty finding individual plantings in the field at the oldest restored sites.

Data Analysis

We focused our analysis on detecting relationships between project age and riparian forest and aquatic habitat metrics. Plot and stream cross-section data were summarized into one mean value of each metric by site for analysis to avoid pseudoreplication (Hurlbert 1984). We then tested each metric for curvilinear or linear fits. Models were constructed with the generalized least squares function in S-Plus version 8 (Insightful Corp., Seattle, WA). Polynomial, linear, and null

(intercept only) models were compared with likelihood ratio tests. If the models were significantly different ($P < 0.05$), we chose the model with the lowest Akaike Information Criteria (AIC; Akaike 1974), otherwise the model with fewer parameters was selected. If a linear model was better than the polynomial model, we compared the linear model to a model with no slope parameter using the same approach. Once best fits were determined, the same parameters were estimated with least squares regression to extract multiple R^2 values as an assessment of goodness-of-fit.

Results

Riparian Vegetation

Sixteen of 21 riparian vegetation metrics were significantly related to project age, including 12 positive and 4 negative trajectories (Table 1). The considerable increase over time in total woody vegetation (Fig. 3), native tree, and exotic shrub/vine densities were best characterized by polynomial relationships with project age, but only total woody vegetation had a relatively good fit. Exotic tree density did not demonstrate a significant trajectory while the best fit for native shrub/vine density was linear, but the fit was poor (Table 1).

Total canopy cover, native tree canopy cover, ground cover, and exposed root cover increased curvilinearly as a

Table 1. Riparian vegetation parameter estimates for best fits determined by likelihood ratio tests ($P < 0.05$) comparing polynomial, linear, and null models using generalized least squares.

Restoration Metric	Best Fit	Parameter Estimates			R^2
		y-intercept	*x*	x^2	
Density (individuals ha^{-1})					
Total woody vegetation	polynomial	459.8	329.9	−7.6	0.39
Native tree	polynomial	145.5	60.6	−1.5	0.16
Native shrub/vine	linear	204.8	25.1	—	0.08
Exotic tree	n.s.	4.6	—	—	—
Exotic shrub/vine	polynomial	32.3	92.2	−1.9	0.13
Absolute cover (%)	—	—	—	—	—
Total canopy	polynomial	11.6	4.9	−0.09	0.56
Native tree canopy	polynomial	10.7	4.7	−0.09	0.54
Ground cover	polynomial	81.9	0.4	−0.01	0.04
Exposed root	polynomial	−0.3	0.5	−0.01	0.26
Total vegetation	linear	43.2	−0.3	—	0.05
Litter	linear	19.9	0.4	—	0.21
Relative cover (%)	—	—	—	—	—
Native perennial grass	n.s.	4.5	—	—	—
Native perennial forb	n.s.	2.5	—	—	—
Exotic perennial grass	n.s.	2.9	—	—	—
Exotic perennial forb	n.s.	1.8	—	—	—
Annual grass	polynomial	15.3	−0.8	0.01	0.28
Annual forb	polynomial	10.3	−0.6	0.008	0.30
Species richness (spp. plot^{-1})	—	—	—	—	—
Tree	polynomial	0.6	0.16	−0.004	0.27
Shrub/vine	polynomial	0.4	0.1	−0.002	0.24
Perennial herbaceous	polynomial	1.9	0.1	−0.004	0.14
Annual herbaceous	linear	4.4	−0.1	—	0.21

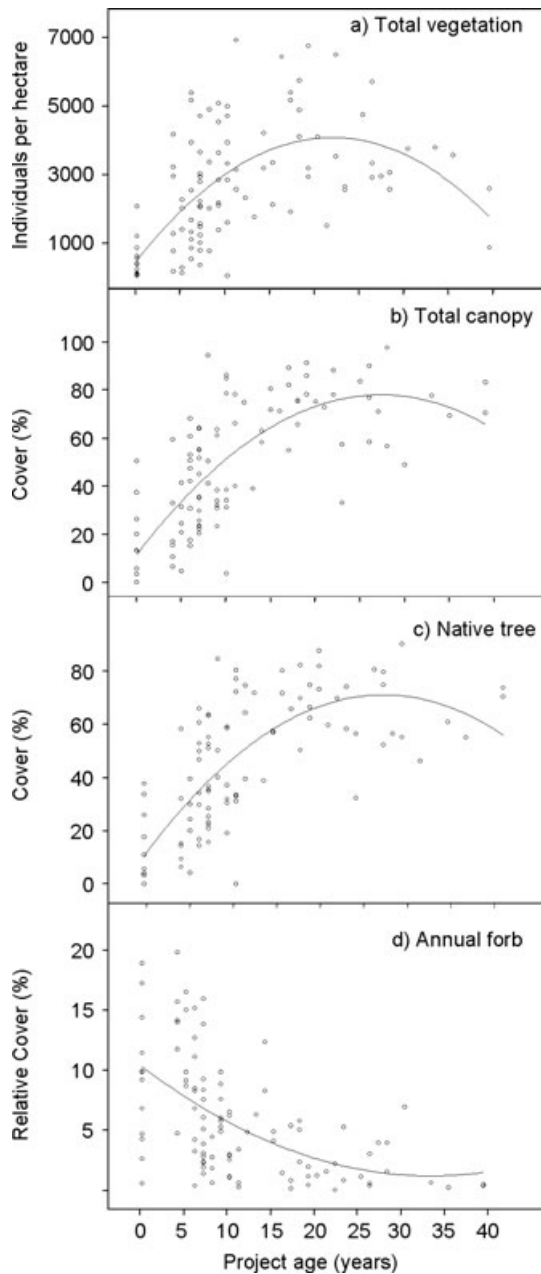Correlation coefficient (R^2) determined with ordinary least squares regression.

Figure 3. Vegetation attributes as a function of project age ($n = 102$) for total woody density a), total canopy cover b), native tree cover c), and annual forbs relative cover d).

function of project age, while litter cover increased in a linear positive manner and total vegetation cover decreased linearly. Native and exotic perennial grass and forb results were highly variable and no significant relationships with project age were found. Relative cover of annual forbs (Fig. 3) and grasses had negative curvilinear trajectories. Species richness metrics had positive curvilinear relationships to project age for the tree, shrub/vine, and perennial herbaceous functional groups. Annual species richness decreased linearly as project age

increased. Of all these significant relationships, the best fits were total canopy cover and native tree canopy cover (Fig. 3).

Aquatic Habitat

Significant relationships with project age were observed for 11 of 15 aquatic habitat metrics, including eight positive and three negative trajectories (Table 2). Stream channel morphology results had significant trajectories for five of the six attributes. The width-to-depth ratio of the bankfull channel had a negative linear relationship with project age. Streambank stability had a positive curvilinear relationship with project age and no relationship was found for bank slope angle. The three woody debris frequency metrics increased over time (Fig. 4). Small and large wood frequencies were best described by curvilinear relationships with project age, while aggregate debris jams of wood were best described by a linear relationship.

Water column attributes had significant trajectories for six of the nine investigated. Stream shade metrics, including intercepted solar radiation, canopy density, and linear channel canopy all increased curvilinearly over time (Fig. 4). Fine sediment and embeddedness showed no significant trajectory. Pool habitat metrics that had curvilinear relationships with project age were maximum and mean pool depth as well as pool habitat type. Pool frequency was not significantly related to project age (Table 2).

Discussion

Riparian Vegetation

While many significant polynomial and linear relationships with project age were detected, most were relatively weak as indicated by the R^2 values. However, we expected high variability given the complex biophysical settings inherent to riparian ecosystems specifically and Mediterranean climate in general. The fact that we detected trajectories at all indicates their broad application and importance to understand fundamental changes following restoration.

Site-specific revegetation strategies accomplished the main objectives of increasing woody species abundance and diversity. Native tree establishment was the focus of revegetation efforts, so the large increases in tree density and cover were expected (Fig. 2). Overall, an indirect plant community response was predicted to follow a successional shift over time from exotic annual herbaceous species to woody vegetation composed of overstory trees with a mosaic of native shrubs and herbaceous perennials (Parker 1997; Dobkin et al. 1998). We detected this basic sequence, although native perennial grasses and forbs did not show any long-term directional trend, and shrubs colonized faster than has been observed at more xeric inland riparian areas (Dobkin et al. 1998). Tree density peaked 15–25 years after restoration. Canopy cover increase was relatively rapid indicating improved terrestrial habitat for birds (Dobkin et al. 1998; White et al. 2005; Golet et al. 2008), amphibians (USFWS 2002; Bulger et al. 2003), and various wildlife species (Golet et al. 2008). In addition,

Table 2. Aquatic habitat parameter estimates for best fits as determined by likelihood ratio tests (P < 0.05) comparing polynomial, linear, and null models using generalized least squares.

		Parameter Estimates			
Restoration Metric	*Best Fit*	*y-intercept*	*x*	x^2	R^2
Stream channel morphology					
Bankfull width:depth ratio	linear	35.5	−0.6	—	0.10
Bank stability (%)	polynomial	67.6	2.5	−0.06	0.26
Bank slope (degrees)	n.s.	15.2	—	—	—
Small woody debris (count $100m^{-1}$)	polynomial	−0.4	0.5	−0.007	0.48
Large woody debris (count $100m^{-1}$)	polynomial	−0.2	0.1	−0.002	0.32
Aggregate woody debris (count $100m^{-1}$)	linear	0.003	0.07	—	0.34
Water column					
Intercepted solar radiation (%)	polynomial	19.1	4.8	−0.08	0.52
Canopy density (%)	polynomial	12.3	5.0	−0.08	0.49
Linear channel canopy (%)	polynomial	−0.5	5.8	−0.1	0.49
Fine sediment (%)	n.s.	15.3	—	—	—
Embeddedness (%)	n.s.	41.2	—	—	—
Pool habitat (%)	polynomial	28.7	1.8	−0.04	0.10
Pool frequency (count $100m^{-1}$)	n.s.	3.3	—	—	—
Maximum pool depth (m)	polynomial	0.6	0.04	−0.0009	0.19
Mean pool depth (m)	polynomial	0.4	0.03	−0.0007	0.18

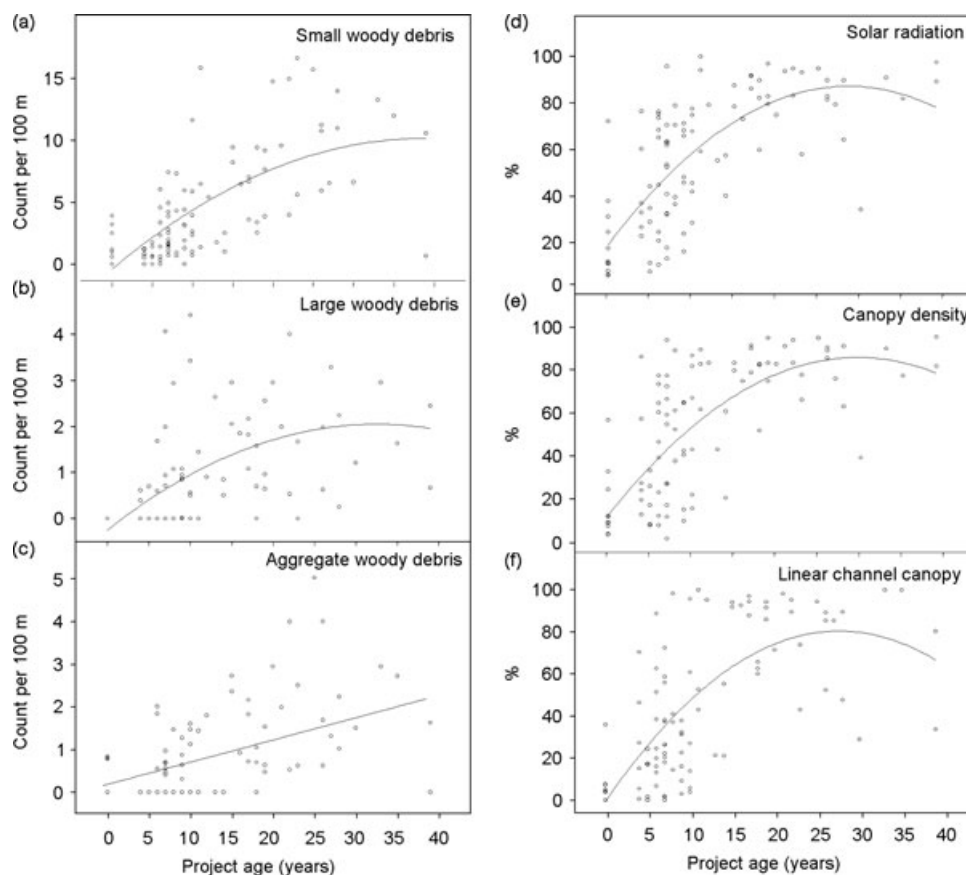Correlation coefficient (R^2) determined with ordinary least squares regression.

Figure 4. Aquatic habitat attributes as a function of project age ($n = 102$) for small woody debris a), large woody debris b), aggregate woody debris jams c), intercepted solar radiation d), canopy density e), and linear channel canopy f).

riparian vegetation changes at restored sites indicated improvements in ecosystem services such as carbon storage via greater tree abundance (USDA 2000). Other ecosystem services that may be improved under these trajectories include diversity (Hupp & Osterkamp 1996; Hobbs 1993), pollination (Kremen et al. 2004), sedimentation (Hupp & Osterkamp 1996; Corenblit et al. 2007), nutrient cycling (Peterson et al. 2001; Kauffman et al. 2004; Sheibley et al. 2006), and trophic dynamics (Baxter et al. 2005; Muotka & Syrjanen 2007).

The increase in exotic shrub density over time was unintended and undesirable. This phenomenon has been noted in past work (Borgmann & Rodewald 2005; Badano et al. 2007). Exotic tree abundance did not correlate with project age, but these taxa were occasionally present at restored sites from previous plantings. In contrast, the most common exotic shrub, Himalayan blackberry (*Rubus discolor*), dominated many older restored sites (greater than 20 years old) by establishing homogeneous patches, which is similar to observations by Lambrecht-McDowell and Radosevich (2005). The rapid trajectory of exotic shrub abundance reduces options for management in the riparian corridor. Consideration of exotic vegetation should focus on the trade-offs that exotic species present for achieving management goals over multiple decades (Parker 1997). For example, White et al. (2005) found juvenile Swainson's Thrush (*Catharus ustulatus*) used Himalayan blackberry for cover and food, so removing this vegetation from recently restored sites may affect wildlife populations negatively. However, delaying active control of exotic shrubs past the initial 20 years of restoration may eliminate chances for adaptive management and cost effective solutions, as explained by Zavaleta (2000).

It was not surprising that perennial herbaceous species did not respond to restoration since the focus of revegetation was woody species. Holl and Crone (2004) made similar observations. Annual vegetation was clearly reduced over time, but resurgence of native perennial grasses and forbs is not likely without significant propagule supply (Bartolome et al. 2004) from flood inundation (Hupp & Osterkamp 1996) and less competition from exotic (Holl & Crone 2004) or shrub species (Brown & Archer 1999).

Aquatic Habitat

A primary purpose for establishing native trees, in particular *Salix* species, was to stabilize streambanks (Johnson 2003) because forested vegetation contains the greatest fine root density for erosion resistance (Wynn et al. 2004) and tree density increases channel roughness increasing sedimentation and retention of flood water (Hupp & Osterkamp 1996; Corenblit et al. 2007). Therefore, the changes we found in stream channel morphology and streambank stability were expected and should result in improved water quality with less chronic sediment delivery to streams from restored sites (NCRWQCB 1998; Corenblit et al. 2007). Decreasing the bankfull channel width-to-depth ratio was also an expected response from revegetation because stream channels tend to deepen and narrow as sedimentation on floodplains increases following tree

establishment (Hupp & Osterkamp 1996; Opperman & Merenlender 2004; Corenblit et al. 2007). This process was enhanced by live wood interacting with woody debris forming persistent instream structure, as explained by Opperman and Merenlender (2007). The accumulation of large wood and debris jams provides greater complexity of instream habitat such as deeper pools (Beechie & Sibley 1997) and cover (Cederholm et al. 1997)

Improved pool habitat and depth indicate greater abundance and diversity of aquatic fauna may be able to use habitat at restored sites as complexity within the water column increased over time. Pools provide cover that protect prey from predators, create slower flow niches during winter storms, and contribute to temperature stratification for thermal refugia in summer (Ebersole et al. 2001). The large increase of stream shade attributes over time was an expected outcome and indicates water temperature may be reduced following riparian revegetation (Brown 1969; Opperman & Merenlender 2004). Aquatic habitat metrics that did not improve over time offer further insight into biogeomorphic processes in the riparian zone (Corenblit et al. 2007). Fine sediment and embeddedness of stream channel substrate did not change indicating that these metrics may be linked to watershed processes operating at spatial scales larger than those of the typical revegetation project site (Houlahan & Findlay 2004; Opperman et al. 2005). Moreover, the temporal range of our survey may not have been sufficient to encompass change in these parameters.

While long-term monitoring of individual sites would have produced a clearer understanding of riparian vegetation and aquatic habitat trajectories following restoration, the substitution of space-for-time in our chronosequence comparisons provided useful insights that inform regional restoration efforts. This cross-sectional survey approach also offers an effective option for systematic, objective assessment of completed projects and postproject appraisals (Kondolf et al. 2007; Tompkins & Kondolf 2007). We suggest that stream restoration research further investigate the impact of establishing woody species on stream channel morphology, nutrient cycling, and overall plant diversity. This will prepare the restoration partnership to manage numerous objectives and ecosystem services over multiple decades.

Implications for Practice

- Site-specific riparian revegetation strategies were successful in maintaining native tree and shrub density, cover, and richness over multiple decades.
- Shrub control may be important for maintaining understory diversity at restored riparian sites, since the trajectory for exotic shrub abundance and variability in native herbaceous species indicated a need for vegetation management 10–20 years postrestoration.
- Although aquatic habitat improved following revegetation (e.g., more shade, more woody debris, and deeper pools), other important instream attributes such as fines

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and embeddedness did not recover over multiple decades and may be controlled by watershed factors.

- Monitoring of riparian revegetation projects should include bank stability, woody debris, channel width-to-depth ratio, and pool depth where appropriate, in addition to plant diversity and cover over time.

Acknowledgments

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