NATIVE TREE RESPONSE TO RIPARIAN RESTORATION

TECHNIQUES IN COASTAL NORTHERN CALIFORNIA

by

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ABSTRACT

Ranchers, farmers and land managers have implemented riparian restoration projects over the last few decades, working with resource agency staff and restoration practitioners; however, quantified regional assessments of trajectory and method effectiveness were not available. I conducted a retrospective post project assessment using a cross-sectional survey of riparian revegetation projects (n=89) and non-restored sites (n=13) on working and historic ranches in north coastal California. I measured composition of woody flora along alluvial stream reaches using belt transects with sampling plots corresponding to floodplain topography. I determined species planted and bank stabilization methods utilized at each project site. Non-restored sites surveyed (n=13) were comparable to pre-project vegetation condition. The study design allowed an assessment of population trajectories, or recovery timelines, ranging from 4 to 39 years since project implementation. I used a count-based statistical approach to analyze the number of live, established trees per plot for ten common genus groups - tree *Salix*, shrub *Salix, Populus, Alnus, Pseudotsuga, Fraxinus, Acer, Umbelullaria,* evergreen *Quercus*, and deciduous *Quercus*. I also quantified the density and trajectory of woody vegetation functional groups (native tree, exotic tree, native shrub, and exotic shrub) and composition by frequency observed at restoration sites.

I found significant effects of restoration on all ten groups assessed depending on the technique of restoration utilized. Passive restoration included large herbivore management using exclusionary fencing or livestock management techniques. Active revegetation methods included tree planting and bioengineering (deflectors, baffles or willow walls). Five groups were positively affected by passive restoration alone while three groups colonized bioengineering structures significantly ($p<0.05$). Active restoration had a greater effect than passive on nine of the ten groups analyzed. Direct planting increased the abundance of all ten groups. The effect of restoration on each group (using regression coefficients) was negatively affected by diaspore mass and positively affected by direct planting practices ($R^2=0.55$, $p<0.0001$). Population trajectory analysis found significant positive effects of project age for five of the groups analyzed. These recovery models further validate restoration outcomes as self-sustaining populations and guide quantified monitoring objectives. Project planning should continue to follow site-specific approaches to riparian restoration and the environmental factors assessed in this study, such as the relative affect of perennial stream flow and channel morphology, provide further insight for this process.

Chair:

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Introduction

Riparian forests provide critical habitat and hydrologic functions, while contributing to viable agricultural production systems and recreational opportunities; however, riparian forests have been widely degraded (Hobbs 1993). In response to this situation, billions of dollars have been spent in the United States on stream and river restoration (Palmer et al. 2005). The number of river restoration projects in the United States has steadily increased over 20 years since the 1980's from near 100 projects to over 4000 projects per year. California ranked as the third highest state for relative spending on stream restoration with $5,953,951 per 1000 kilometers (Bernhardt et al. 2005).

A common objective in restoration of riparian systems is the establishment of native plant populations and forest cover (Bernhardt et al. 2005, Palmer et al. 2005). California's ranchers and farmers have worked with local resource agencies and restoration consultants to restore riparian areas over the last three decades to meet multiple resource management objectives. Exclusionary fencing, livestock management and land use control/preservation are common passive methods. Common active restoration methods included bank stabilization, tree planting and instream habitat enhancement (McIver and Starr 2001).

Qualitative protocols promoted by state and federal agencies have been shown to be viable options for rapid assessments of stream health in California (Platts et al. 1987, Ward et al. 2003). Quantified studies have found riparian vegetation recovery resulting from passive revegetation methods such as livestock exclusion (Platts 1981, Kauffman et al. 1997), deer exclusion (Opperman and Merenlender 2000), wolf reintroduction (Ripple

and Beschta 2003) and livestock management (Ward et al. 2003). Following 20 years, streams carrying high amounts of bedload in alluvial reaches had low success of active instream enhancement structures in Oregon and Washington streams (Frissell and Nawa 1992).

While some studies have evaluated success of active and passive restoration, few have compared the long-term results from active and passive revegetation designs on a relative basis (McIver and Starr 2001, Thayer et al. 2005). When to implement best management practices depends on understanding where objectives may be reached with limited resources compared to the more expensive alternatives, which may be reserved for certain locations within the watershed (McIver and Starr 2001).

Restoration project monitoring has generally focused on survival of planted individuals within a contracted three to five year period, and surveys have rarely studied influences of revegetation practices on the resulting plant community dynamics (Hourdequin 2000). Post project analyses have provided valuable feedback for the design of future projects (Kondolf 1995, Kondolf et al. 2001), however, limited documentation has been available comparing long-term effectiveness of riparian management decisions (Palmer et al. 2005).

Performance standards have been challenged to provide realistic expectations given site-specific physical characteristics and temporal trajectory (Palmer et al. 2005). A large number of well documented restored sites may be more useful than reference sites for setting quantified objectives of stream restoration at a regional scale (Hughes et al. 2005) with the range of common site conditions may be accounted for statistically (Conroy and Svejcar 1991). Recovery models describing the range of potential outcomes at “restored”

project sites provide opportunities for further understanding of community structure, spatial arrangement in the context of ecosystem processes (Jelinski and Kulakow 1996; Falk et al. 2006).

Ecological attributes offer practical tools of restoration metrics to guide adaptive management decisions and communicate natural resource concepts to the public (SER 2002, Falk et al. 2006); however this depends on spatial and temporal context. For example, seed mass (Baraloto et al. 2005), seed size (Allen 1997, Gabbe et al. 2002) and dispersal mode (Shear et al. 1996, Takahashi and Kamitani 2004) have affected the ability for certain species to colonize restoration project sites and natural generation has been shown to not produce diversity similar to historical bottomland hardwood forest composition (Allen 1997, Gabbe et al. 2002). Conservation, restoration and management planning has increasingly utilized physiology and life history traits to explain research results and inform policy (Wikelski and Cooke 2006).

Restoration trajectory models offer a useful tool to validate whether project effectiveness is self-sustaining (Palmer et al. 2005) and achieving the desired outcome over time (Hobbs 1993, Hobbs and Norton 1996, Zedler and Callaway 1999, Choi 2004, Ruiz-Jaen and Aide 2005, Falk et al. 2006). Post-project assessments have demonstrated mixed results including successes and failures. Rapid recovery in riparian areas has been associated with physical factors such as floodplain access while a slow trajectory was due to degradation beyond thresholds for populations to establish (Pimm 1991, Lindig-Cisneros and Zedler 2000, Sarr 2002, Jordan 2003).

In order to compare long-term results of active and passive riparian restoration, I conducted a retrospective, cross-sectional survey of riparian revegetation projects at 102

survey sites in coastal northern California. I measured the abundance of woody species and physical characteristics at restored and non-restored sites in order to model correlations with common stream restoration practices. The passive restoration treatment was large herbivore management, which included livestock and deer control with exclusionary fencing (Schultz and Leininger 1990, Opperman and Merenlender 2000) or managing livestock number and/ or season of access (Conroy and Svejcar 1991, Ward et al. 2003, Allen-Diaz et al. 2004) usually following a few years of rest (Popolizio et al. 1994). Active methods investigated were tree planting and bank stabilization, which utilized bioengineering technology. I surveyed non-restored sites where feasible to quantify the relative effect of passive and active restoration methods. Rather than compare dissimilar species, I analyzed tree abundance per plot by genus for eight common native tree genera.

The project sites I surveyed were restored over a period of nearly four decades and thus represent a continuum of stages in riparian habitat recovery for the survey area. I was primarily interested in three fundamental research questions related to the abundance of common tree genera observed. 1) How did passive (large herbivore management) and active restoration methods (direct planting, bioengineering) affect the density of each tree genera? 2) Were native tree populations increasing in abundance over time? 3) What physical factors affected passive restoration potential?

I assessed native tree abundance using site-specific and landscape-scale environmental factors, amount of time since restoration and restoration techniques implemented. Diaspore mass, size and type are highly variable in woody riparian flora and dispersal mechanisms include combinations of water, wind, and/or animal (Baker

1972). These species-specific life history traits justified performing multiple analyses by genus. Colonization by diaspores and establishment depend on environmental factors, such as a particular site's hydrologic regime, channel morphology and groundwater availability. Flood inundation effects the dispersal of both seeds and sediments regulating the potential for recruitment of riparian species at restoration sites (Mitsch et al. 1998, Alpert et al. 1999, Andersson et al. 2000, Thayer et al. 2005).

I expected the abundance of tree genera that produce light weight diaspores would be most affected by passive restoration methods because they produce copious small seeds which have adapted to long distance migration (Greene and Johnson 1993). In contrast, I expected the heavier diaspore producing species would reestablish population abundance where direct planting was utilized. I anticipated that bioengineering projects not only increased the density of *Salix* species, but also increased colonization by other tree species as flood debris become entrained by branches and fine sediment is deposited (Wehren et al. 2002).

I used the effect of project age to indicate a relative increase over time in population abundance at project sites and indicate a positive population trajectory. This provided a measure of self-sustaining reproduction and quantified rates of recovery for each genus. The analysis of tree response to restoration efforts given temporal and spatial factors provides further guidance in site-specific project designs and realistic project monitoring objectives (Hughes et al. 2005, Palmer et al. 2005).

The history of participation by private landowners in riparian restoration since the 1970's in coastal California made this survey feasible. The landowners I visited for site access permission were generally interested in the survey and many had concerns

regarding site management and maintaining project goals over the long-term. Though riparian vegetation planting and bioengineering stream bank stabilization methods have undergone phases of field implementation and refinement (CDFG 1998, Wehren et al. 2002, CRP 2004), quantified riparian forest management outcomes over multiple decades were unknown (Palmer et al. 2005).

Methods

Study Area

I surveyed riparian vegetation and bank stabilization project sites north of the San Francisco Bay, California. As a northern coastal region, the study area is a Mediterranean climate with cool, wet winters and warm, dry summers. Forest vegetation is characterized by redwood, mixed evergreen, savanna, and mixed hardwoods (Hickman 1993). Numerous small intermittent and perennial streams drain the Coast Range. The, low gradient valleys were the most accessible and were the first locations settled for ranching historically. Intensive natural resource management such as continuous open grazing was common (Opperman and Merenlender 2003). The high erosion potential of the coastal mélange Franciscan geology was a concern following large floods during the 1960's. As stream banks continued to unravel, riparian revegetation was a primary tool to stabilize sensitive areas. Current land management is a mixture of private beef ranching, dairy, forestry and vineyard agriculture as well as public recreational areas.

I worked with Resource Conservation District and Natural Resource Conservation Service staff, in collaboration with restoration consultants, to identify potential projects to be included surveyed. The local Watershed and Range Management Advisors from the

University of California Cooperative Extension provided assistance contacting private landowners for site access permission. I surveyed each site that met the following parameters.

1. Restoration project was completed at least four years ago;
2. Projects with known installation dates and species planted;
3. Alluvial, gravel substrate stream reaches;
4. Mixed oak woodland, savanna or grassland vegetation types;
5. Tributary drainages of primarily first, second and third order streams;
6. Minimal tree cover prior to project installation;

I surveyed 102 sites along 19.4 km. of stream in Marin (n = 23 sites), Mendocino (n = 38), and Sonoma (n = 41) Counties (Appendix A). Surveyed project sites received combinations of restoration methods, which included 1) herbivore management (n = 89), and 2) planting of common native tree species (n = 53), and/ or 3) bioengineering (n = 37). All 89 surveyed project sites received varying degrees of large herbivore management. This included livestock or deer enclosures with exclusionary fencing as well as reducing the total number of livestock accessing the site. Herbivore management alone without active revegetation was considered a passive restoration technique (n = 37) and specifically included livestock exclosure or removal (n = 18), deer exclosure (n = 12), and livestock management (n = 7). Active restoration methods employed were direct tree planting and bioengineering bank stabilization project components, which treated multiple locations within a project site. Bioengineering techniques utilized live plant material with willow wall and/or deflector/ baffle construction designs (Flosi et al. 1998, Wehren et al. 2002, Gerstein and Harris 2005).

Non-restored sites were surveyed where anecdotal history and site access enabled an opportunistic survey (n = 13). I confirmed site history from restoration consultant and landowner interviews regarding vegetation structure similar to the adjacent restoration project site before revegetation occurred. The non-restored sites provided a conservative context of quantified pre-restoration conditions because most project sites treated the most degraded locations in the watershed (Wehren et al. 2002).

Sites surveyed ranged in project age from 4 to 39 years since restoration. This range in project age provided the continuum over multiple decades to represent riparian recovery. Most sites had seasonal stream flow and were dry in the summer (60 of 102). Other physical factors were summarized to characterize the population of inference (Table 1).

Data Collection

I characterized survey sites according to the following components: 1) History of restoration activities; 2) Physical conditions; and 3) Woody species abundance. I summarized project design information from past reports as well as anecdotal surveys of landowners and restoration practitioners for the 89 restored sites. I recorded species planted, bank stabilization structures installed and various management activities. More detailed information such as number of each species planted was not available for most sites to perform comparable analyses.

Current plant species abundance was determined from plots (n=2435) along transects perpendicular to the channel. Based on the site length measured from walking the stream channel, I located three stratified cross-sections at equal intervals in each site surveyed. Therefore, a total of six, 7.3 meter wide belt transects extended up the stream bank from

the thalweg. Plot location was based on channel morphology and the extent of comparable landform until the upper bank was sampled in the final plot of each transect. Plot length was variable. I used the lowest bankfull elevation indicator (break in slope of a flat depositional surface flooded every 1-2 years on average) and floodprone elevation (2 x bankfull depth) to place floodplain plot locations (Rosgen, 1996).

Data gathered within each plot included species composition (Hickman 1993) by age-form class (BLM 1996). Because riparian plant community composition varies by geomorphic zone (Harris 1999, 1987), I classified the elevation above the thalweg for each plot and calculated relative plot height as the number of bankfull heights from the plot's vertical midpoint. Stream flow was characterized as perennial or intermittent with no flow during the summer. This was used to indicate summer groundwater availability at each site surveyed which is often a limiting factor for riparian species (Opperman and Merenlender 2003). I assessed soil particle size from each site to account for soil water holding capacity (Gee and Bauder 1986) using four composite samples stratified up the bank given landform distribution. These physical factors allowed plot-scale tree abundance data to be linked to specific cross-section morphology and account for within site variation.

Project site location was used to assess coarse spatial data using Geographic Information System software (ArcGIS 9.1). Ambient temperature and precipitation data were summarized from Parameter-elevation Regression on Independent Slopes Models (PRISM) as described by Daly (1997) over 30 years of data from 1971 to 2000 (Climate Source, 2001). I gathered data for each survey site within a 100 meter buffer using the intersect tool available in ArcGIS Spatial Analyst. The mean maximum summer

temperature indicated the degree of potential relative heat stress encountered at each survey site. I utilized the cover type data available from California Land Cover Mapping and Monitoring Program (CDF 2005) to calculate percent forested as the sum of conifer, hardwood and mixed woodland in the landscape surrounding each site for 100 meters. This offered a relative amount of seed source available from woody species to each site (Appendix A).

Data Analysis

I summarized the overall effects of restoration on riparian forest structure in the study area by comparing restored sites to non-restored sites using the abundance of tree and shrub/ vine functional groups by native and exotic origin to north coastal California. These data did not fit a normal distribution and was skewed to the left dominated by zero values. Therefore, I used count-based analysis negative binomial regression (Intercooled Stata v.8.0) to test for significant differences between non-restored and restoration sites. Site was included as a "cluster variable" in the analysis to account for spatial co-dependence between sites surveyed. I utilized plot size as an "exposure variable" in order to statistically account for variable sample area (Long and Freese 2006).

I analyzed the trajectory of woody vegetation functional groups by using the age of each project to predict abundance over time. Separate analyses were performed on each of the four functional groups– native tree, exotic tree, native shrub/ vine, and exotic shrub/ vine. The regression coefficient results allowed transformation by density (individuals/hectare) for facilitating interpretation. I utilized the standard error and confidence interval (95%) of regression coefficients to graphically represent the variability of native tree abundance over time.

I summarized the woody species encountered at restoration project sites by calculating the frequency observed in survey plots as the % of sites present. These results indicate riparian forest composition of the four functional groups.

Since the establishment of tree species was the focus of riparian restoration efforts, I focused my analysis of restoration methods on eight genera of small and large trees. They were chosen for analysis because of their wide distribution in riparian areas of the study area and frequent planting in revegetation efforts. The light diaspore mass (USDA 1974) genus groups I analyzed were tree *Salix* (red & shining willow), shrub *Salix* (arroyo & sandbar willow), *Populus* (Fremont's & black cottonwood), and *Alnus* (white & red alder). The heavy diaspore mass (USDA 1974) genus groups analyzed were *Fraxinus* (Oregon ash), *Acer* (box elder & big leaf maple), *Umbellularia* (bay), evergreen *Quercus* (coastal, canyon, & interior live oak), deciduous *Quercus* (valley, Oregon, & black oak), and *Pseudotsuga* (Douglas-fir). I chose to utilize genus and functional groups instead of individual species for my response variables in order to reduce the number of analyses conducted and increase the sample size while still documenting regional forest composition. I assumed response was similar within each genus except for *Salix* and *Quercus*, which I subdivided into functional groups because of their respective species richness and ecological importance.

My analysis of the 10 genus groups consisted of four restoration treatment combinations and six covariates as predictor variables. I used the number of established individuals in each genus group per plot (e.g. number of *Populus* observed in the sample) to determine population response to restoration practices. Established trees were considered taller than browse height (>4 feet tall). These data were not distributed

normally when analyzing by genus and were dominated by zero values. I used the negative binomial regression (Intercooled Stata v.8.0) to test for significant relationships within each genus group (Long and Freese 2006). The sample size for each treatment level depended on the number of sites where each group was planted. I categorized the following restoration treatment combinations in addition to non-restored sites (HM-, P-, B-).

- Herbivore management, genus not planted, not bioengineered (HM+, P-, B-);
- Herbivore management, genus planted, not bioengineered (HM+, P+, B-);
- Herbivore management, genus not planted, bioengineered (HM+, P-, B+);
- Herbivore management, genus planted, bioengineered (HM+, P+, B+).

I selected predictor variables *a priori* to avoid autocorrelations (Quinn and Keough 2003). Six covariates were added as predictor variables to the restoration treatment combinations. They included: 1) project age, 2) clay soil particle size, 3) maximum summer ambient temperature, 4) percent forested near site, 5) relative plot height above thalweg, and 6) perennial stream flow. Site was included as a "cluster variable" in the negative binomial regression model to account for spatial co-dependence between sites surveyed. I utilized plot size as an "exposure variable" in order to statistically account for the variable size sampling area (Long and Freese 2006) and transformed model results for understanding interrelationships of effects on tree abundance.

Given the variability between and within sites, multiple predictor variables were included in order to reduce the variability due to spatial and temporal spurious effects. I used backward step-wise regression at 90% confidence level (*P*-Value <0.10) in order to account for the high variability inherent in such a retrospective study design (Mapstone

1995, Quinn and Keough 2003). I wanted to avoid missing a treatment effect (Type II statistical error) and no correction for multiple comparisons tests was performed; however, a conservative interpretation may add a Bonferroni procedure to correct for cumulative Type I statistical errors and apply a *P*-Value of 0.01 for interpreting model results (Quinn and Keough 2003). I utilized the traditional 0.05 *P*-Value threshold for assigning significant model effects.

Analysis of response over time, or population trajectory analysis, utilized the years since project installation as a project age effect. I transformed count-based regression coefficients back to density (individuals per hectare) for graphical representation of trajectory results and physical effects. The oldest project site surveyed that planted the tree group set the maximum range of x-axis project age values for each genus groups.

For analysis of the effect of seed morphology on restoration response, I utilized the natural logarithm of median diaspore mass (USDA 1974) for each genus group to predict the effect of restoration for the four restoration treatments compared to non-restored sites as quantified by negative binomial regression coefficients. The regression coefficients with *P*-Value>0.10 were considered to have no effect on restoration and zero values were given. I tested for normal distribution and homogeneous slopes with the analysis of covariance (ANCOVA) statistical test (JMP 5.1 software).

Results

Woody Vegetation Trajectory

Mean project age was 13 years since restoration was implemented, and ranged from 4 to 39 years. Woody vegetation structure was significantly affected by riparian restoration

in the study area for each functional group (Figure 2). I found over nine times greater abundance of native trees ($p=0.004$) and nearly six times greater abundance of native shrubs and vines ($p<0.001$) at riparian restoration sites compared to non-restored sites. The abundance of exotic trees ($p=0.010$) and exotic shrub/ vines ($p<0.001$) was also significantly greater at restored sites than non-restored sites.

Trajectory analysis of woody vegetation structure found the four woody vegetation functional groups increased in abundance over time since restoration was implemented. Native tree abundance rapidly increased immediately following restoration at a greater density than the other functional groups; however, following five years exotic shrubs became more abundant and after 15 years native shrubs were relatively more abundant than native trees (Figure 3). Native tree abundance had the weakest relationship with project age over multiple decades of the four functional groups ($p=0.052$). Plus, I found high variability in the native tree trajectory (Figure 4).

The composition of woody flora at restoration sites included 19 native tree species (Table 2). The most common woody species encountered was *Salix lasiolepis* (arroyo willow). The four exotic tree species encountered occupied few of the sites. The shrub and vine species included 18 native species and 6 exotic species (Table 3). Most exotic shrub species occupied few of the sites; however, *Rubus discolor* (Himalayan blackberry) was a widespread shrub species.

Passive & Active Methods

I summarized the negative binomial regression results and mean density by restoration treatment for the genus groups producing light diaspore (Figure 5) and heavy diaspore mass (Figure 6). The significant effects between restoration treatments within

each genus group ($p<0.05$) were indicated by different letters. I provided the complete model results with regression coefficients, which quantify the effect of each predictor variable on tree abundance, for the treatments and covariates (Table 4) as well as the results matrix of the full model comparisons (Appendix B).

I organized the effects on tree abundance from passive (herbivore management alone) and active (planting and bioengineering) restoration methods by genus group (Table 5). Passive restoration positively affected the abundance of five groups and bioengineering significantly affected four of the 10 groups analyzed. Specifically, *Salix, Alnus*, *Populus* and *Fraxinus* significantly established by passive restoration methods alone. *Salix* and *Alnus* significantly ($p<0.05$) colonized bioengineering structures where they were not planted while *Fraxinus* and *Pseudotsuga* showed nearly significant results ($p<0.10$).

Of the active restoration methods, direct planting significantly affected the abundance of all tree groups analyzed. *Alnus* and shrub *Salix* abundance were most affected by the combination of both active methods bioengineering and planting (Figure 5). *Populus* response to restoration treatments showed a different pattern. Direct planting was the most effective method to establish the greatest abundance for seven groups - *Populus, Pseudotsuga, Fraxinus, Acer, Umbellularia,* evergreen *Quercus* and deciduous *Quercus* (Figures 5 and 6). The effect of planting for these seven groups was confirmed by comparing to the passive restoration effects. There was a significant difference between planting and herbivore management for the abundance of *Populus, Pseudotsuga, Fraxinus, Acer, Umbellularia,* and *Quercus*.

Population Trajectory

The temporal effect of project age had a positive affect on the abundance of five tree genera analyzed. Shrub *Salix* (Figure 7), *Fraxinus* (Figure 8), *Acer*, and both *Quercus* groups (Figure 9 and 10) were statistically significant while Umbellularia was nearly significant. These population trajectory models were based on the project age and restoration treatment regression coefficients while holding the other significant covariates constant (Table 4).

In general, the temporal effect of project age was stronger for the larger diaspore producing groups. Most light diaspore producing trees did not show a positive trajectory except for shrub *Salix*. After 35 years, shrub *Salix* density was approximately 700 trees per hectare where planting and bioengineering was combined. Sites that planted *Fraxinus* had more than 50 trees per hectare compared to non-planted sites with about five per hectare by year 20. Deciduous *Quercus* showed the slowest significant population increase over time with 10 trees per hectare if planted and about two per hectare where not directly planted following 25 years.

Site Physical Factors

Environmental variables produced patterns in relationships to the abundance of tree genera analyzed. The relative height on the bank of each plot had a significant negative effect on all four light diaspore tree groups (Table 6). For example, *Alnus* abundance significantly decreased as the relative bank height above thalweg increased (Figure 11). In contrast, *Acer*, *Umbellularia* and deciduous *Quercus* increased abundance as the height on bank increased (Table 6).

Clay soil particle size was positively correlated with shrub *Salix*, evergreen *Quercus* and *Pseudotsuga* abundance while *Populus* had a nearly significant trend. In contrast, tree *Salix* abundance had a significant negative relationship with clay soil particle size (Figure 12).

Maximum summer temperature positively correlated to the abundance of five tree groups. As temperature increased the abundance increased for shrub *Salix*, *Populus* (Figure 13), *Fraxinus*, evergreen *Quercus*, and deciduous *Quercus*. Forested near the project site had a negative effect on shrub *Salix* while a positive effect on *Alnus*, *Pseudotsuga*, *Fraxinus*, *Acer*, and *Umbellularia*.

Perennial stream flow at sites had a positive effect on the abundance of *Salix*, *Populus*, *Alnus*, *Fraxinus*, *Umbellularia*, and evergreen *Quercus*. I used the combination of stream flow and bank height for both tree *Salix* and *Populus* to compare and contrast their relationships. Both had the greatest abundance at sites with perennial stream flow and in plots at one bankfull height above thalweg. Tree *Salix* abundance was more affected by bank height than stream flow while *Populus* was more affected by stream flow than bank height.

Diaspore Mass

The effect of restoration on the abundance of each genus group correlated to diaspore mass and depended on the restoration method utilized (Figure 14). The lighter the diaspore, the greater the affect of restoration (R^2=0.55, $p<0.0001$). Direct planting had the greatest effect on tree abundance of the restoration treatments investigated.

Discussion

Woody Vegetation Trajectory

Riparian restoration had a significant effect on woody vegetation structure in north coastal California. The relative abundance of the four woody vegetation functional groups increased as a result of restoration activities. The intended result of riparian restoration was reestablishing native trees and this was highly successful. I documented greater than a nine-fold increase in native tree abundance at restored sites compared to non-restored sites. These outcomes document riparian resilience and long-term management tradeoffs.

I also documented a recovery timeline for the four functional groups (Palmer et al. 2005). Native tree abundance showed a weak trajectory. Some species may take longer than 40 years to measure an increase while other species may establish rapidly and decline following 10-20 years. The variability over time of these data questions the self-sustainability of restored forests regarding which restoration methods were successful on which species at site and landscape scales (Palmer et al. 2005).

The understory shrub and vine layer colonized after trees established and increased their abundance at a greater rate than native trees. The exotic woody understory species established relatively faster than native understory species (Lambrecht-McDowell and Radosevich 2005). This result quantifies a management tradeoff between balancing the abundance of native tree and exotic shrub/vine over time. Managing riparian forests over multiple decades should entail vegetation management of exotic shrubs in particular.

I found relatively more native tree and shrub species than exotic woody species at restored sites; however, the high frequency of *Rubus discolor* was an unintended outcome of the projects surveyed though few implemented control practices. *S. lasiolepis* presence may facilitate colonization and establishment by *R. discolor* which offers wildlife habitat in patches, but poses potential landscape-scale concerns because of its tendency to become an unmanageable bramble. Wild fire connectivity may need to be investigated as well as herbaceous plant diversity. Perennial exotic species have undesirable consequences in northern California riparian areas such as reducing colonization by native species (Alvarez and Cushman 2002, Gaffney 2002).

Passive & Active Methods

The results quantified relative effectiveness of passive and active approaches to riparian restoration (Falk et al. 2006). Understanding the successes and limitations of passive restoration outcomes is useful for efficient watershed planning and effective landscape scale effectiveness. Active rehabilitation methods may be implemented when and where passive ones have not produced satisfactory results (McIver and Starr 2001).

Passive restoration using large herbivore management methods was successful to recover half of the groups analyzed (shrub *Salix*, tree *Salix, Populus*, *Alnus*, *Fraxinus*). This ability to establish following restoration without planting indicates the impact large herbivores have on riparian forest structure and species composition in north coastal California. The dominant component of surveyed sites was shrub *Salix* which was able to grown into high density thickets.

Active restoration had a greater effect than passive on nine of the ten groups analyzed. Direct planting alone was effective at establishing all ten groups analyzed. It

was the most effective restoration method to establish the greatest abundance of the heavy diaspore genera as well as *Populus,* which has a narrow recruitment box depending on environmental factors (Busch and Smith 1995).

The combination of both active methods (bioengineering and direct planting) correlated to the greatest abundance of shrub *Salix* and *Alnus*. Bioengineering structures offered stable stream banks on which 3 groups regularly established without planting (shrub *Salix*, tree *Salix*, *Alnus*) and 2 groups had a tendency to colonize (*Pseudotsuga, Fraxinus*). This active restoration method had indirect consequences on riparian forest composition and structure.

These genus-specific differences in response to restoration were confirmed by utilizing the passive restoration sites to compare the effectiveness of active restoration methods. For example, the tree Salix group showed an increase in abundance from all restoration methods but there was no difference between methods. In contrast, the heavy diaspore trees had greater abundance where planted than at restoration sites where they were not planted. Thus, tree *Salix* may not benefit greatly from planting methods or they were not planted frequently enough to be able to measure an effect. Regardless, future restoration may provide a greater benefit to riparian forest composition by planting more tree *Salix* where appropriate.

Deciduous *Quercus*, in comparison, did not establish successfully where it was not planted. This finding agrees with previous research which found marginal passive natural regeneration and long-term population declines of *Q. lobata* in California (Tyler et al. 2006). Other studies have shown passive means alone could not be counted on to

establish all woody forest species (Allen 1997). It is clear from my results that planting deciduous *Quercus* was effective and the practice should continue.

Restoration monitoring programs are challenged to measure the effectiveness of restoration efforts. Non-restored sites offer one tool for comparison as a potential "control". Passively restored sites offer another form of statistical "control" to help interpret the relative performance of active methods and confirm their success. Care should be taken during project implementation to set aside both types of controls for future reference that are comparable to the project site.

Population Trajectory

The trajectory concept has been used to represent a predictable guaranteed outcome, but this has not always occurred, similar to the Clements' climax model (1936). This wishful thinking has been used to justify mitigation assuming that restoration methods would lead to the desired results (Zedler and Norton 1996). Documentation of a positive trajectory for each population is one of the most useful assessments of project validation for riparian revegetation because it offers a measure of self-sustaining forest composition and quantifies multiple timelines for recovery (Palmer et al. 2005).

The range of project ages I surveyed allowed quantification of the effect of restoration method over time since project installation. I inferred that the population was reproducing at project sites in the survey area if a positive trajectory was found statistically. This restoration trajectory shows that population abundance at project sites followed a relative increase over time which provided a measure of population expansion over time (Hobbs 1993, Hobbs and Norton 1996, Zedler and Callaway 1999, Choi 2004, Ruiz-Jaen and Aide 2005, Falk et al. 2006). Planted individual trees growing larger

without reproduction brings into question the resilience and sustainability of planted forests.

I found strong relationships with project age for five of the 10 groups analyzed. Most of these were the heavy diaspore producing tree species that are also slower growing, except for the shrub *Salix* group – the only one of the light diaspore trees able to reproduce clonally into drier locations of the riparian corridor.

The groups that did not show a positive trajectory may be due to multiple factors such as missing elements in community composition, isolation from seed sources, and/or functional ecosystem changes. These effects may not be reversible using passive restoration methods alone (Opperman and Merenlender 2003, Jordan 2003, Faulk et al. 2006). Other factors include very rapid colonization during the first ten years by species like *Alnus*, which proceed to thin their density as they grow into mature trees and abundance declines.

Revegetation technology and techniques have continually evolved. For example, recent projects since the late 1990's layout shrub *Salix* plantings in clumped zones to form a mosaic physiognomy rather than continuously through the entire project site. The tree forms of *Salix* are important components given their gallery riparian forest structure. In contrast, the shrub forms of *Salix* rapidly grow into a thicket forest structure. The tree forms are harder to find at most degraded riparian corridors which may cause them to be planted less frequently than the more common shrub *Salix* populations; however, the tree forms are preferred by many ranch managers and flood control engineers because of their upright form. Potential opportunities for improving future project design should be to plant a greater frequency of tree *Salix* species where appropriate. Restricting the planting

of shrub *Salix* species outside bend locations along streams and eroding landforms where their clonal reproduction strategy increases bank stability.

The temporal effect of project age indicates which genera have established populations at restoration sites and offers guidance for project monitoring. The trajectory models may be useful to set quantified objectives that are species-specific. Riparian project monitoring should design sampling intervals for every five to ten years for ensuring recovery of the fast growing species is documented. These intervals offer opportunities to assess project performance and implement adaptive management. Such standards for project success that contain specific timelines for recovery may become more useful than the elusive reference site, which was helpful for project design; however, the restoration community can now look back upon their accomplishments to guide long-term management by setting hypothesis-based goals that account for variability over space and time (Hughes et al. 2005).

Site Physical Factors

The physical site conditions had large effects on the response of tree genera to restoration and any one of them could be responsible for an unsuccessful project at any specific site. Spatial factors were also critical to understanding the response from restoration, such as a particular site's hydrology affected how floods disperse both seeds and sediments. Incorporating them and other environmental data was important for increasing the confidence in model results for specific restoration treatments (Harris 1999, Thayer et al. 2005) and understanding how results apply to watersheds throughout California and the western United States.

Site-specific effects offer practical use for adapting project designs to local stream morphology. Channel morphology at cross-sections was an important factor influencing the water dispersed tree genera because their abundance depended on floodplain dynamics. The greater floodplain area near bankfull elevation increased the population size of *Alnus* and both *Salix* groups. Therefore, sites with severely incised channels were slow to recover these floodplain dependent tree species. In contrast, three of the six heavy diaspore producing groups had greater abundance as the vertical distance above bankfull channel increased.

Stream flow has been found to be a good predictor of passive restoration potential (Opperman and Merenlender 2003). Project sites on stream reaches with perennial flow produced larger populations for seven of the 10 groups analyzed than sites where the stream dries up in the summer. The sites with summer flow had greater abundance of all four light diaspore groups which responded significantly to passive restoration. Thus, at sites with both perennial flow and accessible floodplain, passive restoration should be successful to establish riparian forest structure.

The combination of perennial flow and bank height provided insight into the autecology of *Populus* and tree *Salix*, which offers guidance on adapting project design to site-specific attributes. Restoration practitioners can expect *Populus* to be abundant where flow is perennial and tree *Salix* to be abundant where floodplains are accessible. Specific objectives may be set based on this understanding and landowners requesting these species can have confidence in achieving successful project outcomes.

Fine substrate dominated watersheds often have greater percent clay soil particle size in the soil and tree *Salix* was less abundant in these locations. In contrast, the shrub *Salix*

group was better adapted to high clay soil content as was evergreen *Quercus* and *Pseudotsuga*. This information is useful to keep site objectives realistic and understand that if tree *Salix* is desired in an upland gully, it will take a lot of effort that would be better used on more appropriate species.

Intact remnant forest near the project site was indicative of potential seed sources (Takahashi and Kamitani 2004). Five groups had an increase in abundance as the amount of forest increased. In contrast, shrub *Salix* did better near less forest cover. Surprisingly, both *Quercus* groups were not affected by forest cover. Thus, one should not assume *Quercus* will establish without planting from relict seed sources.

Summer ambient temperature effects were surprising. I expected a negative relationship with abundance but I found a positive effect on abundance for half of the groups assessed. Their abundance was greatest in hot locations which are often furthest from the cooler coastal climate. *Populus* was most affected by this factor and it does not grow along the immediate coast. This took into account locations on the landscape that were not suitable habitat for these five groups. Presumably, these groups will benefit from global warming.

Diaspore Mass

Overall, the effect of restoration was greatest on the trees that produced light mass diaspores. Similar results have been found in bottomland hardwood forests (Allen 1997) and tropical rainforest (Lamb et al. 2005) restoration assessments. Small diaspores are able to travel greater distances using multiple dispersal mechanisms and greater fecundity (Greene and Johnson 1993). The lighter diaspore mass producing trees are considered

early seral colonizers with full sun habitat requirements and high growth rates (Trowbridge et al. 2005).

The greater the diaspore weight, the more important it is to plant that species regardless of bioengineering. Diaspore mass was found to be less important than dispersal method near intact forest complexes (Takahashi and Kamitani 2004); however, forest fragmentation reduces animal dispersal mechanisms so successful establishment is episodic and driven by sedimentation processes (Florscheim and Mount 2002). Most of the sites I surveyed were relatively barren at the time of restoration and my trajectory results confirm the slow establishment over multiple decades of the heavy diaspore producing trees.

Conclusions

I documented riparian forest composition and structure at restoration sites and compared active revegetation techniques to the inherent ecosystem resiliency of passively restored sites. I found the desired affects were being accomplished such as self-sustaining native tree populations. I also documented unintended consequences such as exotic shrub abundance increased faster than other functional groups over multiple decades. These long-term management tradeoffs must be considered to effectively implement projects throughout a watershed and enlist broad public support from private landowners.

I found significant effects of restoration on all ten genus groups assessed depending on the technique of restoration utilized. Passive restoration included large herbivore management using exclusionary fencing or livestock management techniques and was successful at establishing riparian forest structure. Active revegetation methods included

tree planting and bioengineering (deflectors, baffles or willow walls) and were effective at enhancing forest structure and composition. Five groups were positively affected by passive restoration alone while three groups colonized bioengineering structures significantly ($p<0.05$). Active restoration had a greater effect than passive on nine of the ten groups analyzed. Direct planting increased the abundance of all ten groups.

Population trajectory analysis found significant positive effects of project age for five of the groups analyzed. These recovery models further validate riparian restoration outcomes as self-sustaining populations and guide quantified monitoring objectives within agricultural landscapes. I found slow and fast recovery timelines of eight common native tree genera in the study area. The tree *Salix* and *Populus* recovery did not show a significant trajectory and depended on physical environmental factors. In contrast, shrub *Salix* abundance reestablished rapidly and continued to increase in abundance over time while deciduous *Quercus* population recovery was slow and abundance was much greater at sites where it was planted.

Diaspore mass explained patterns in the effect of restoration across all genera assessed. The effect of restoration on the abundance of each was negatively affected by diaspore mass and positively affected by direct planting practices. Understanding how such species-specific life history traits affect responses to stream rehabilitation further enhances the science of restoration ecology.

Benefits and limitations exist of the study design utilized. Similar retrospective study designs were employed by researchers to document stages of succession from multiple sites given their known date of agricultural abandonment (Oostings 1942, Tilman 1988). Future research should further document long-term response to riparian restoration and

test my results using repeated measurements from the same project sites over time (Harris et al. 2005). Such controlled and ideal studies are intensive and limited by funding to complete follow up surveys 20 years later and large sample sizes are needed to account for the variations in site characteristics of riparian systems. Determining the effect of a specific restoration method was challenging in disturbance dependent ecosystems over long time scales (Hourdequin 2000). The retrospective study design offered an efficient and statistically valid approach to comparing project effectiveness, restoration methods and management tradeoffs in a relative context to guide regional conservation options.

Overall, I found site-specific, outcome-based restoration strategies were successful. The process-based objectives highlighted in case studies, which remove levees (Florsheim and Mount 2002) or alter flow regime (Kondolf et al. 2006) to restore riparian habitat were not practical in the study area. Such idealized options for stream management are not the only successful techniques to accomplish restoration goals at a landscape scale. The results validate site-specific project designs and highlight environmental factors influencing restoration capacity and potential (Hughes 2005). Project planning should continue to follow site-specific approaches to riparian restoration and the environmental factors assessed in this study, such as the relative affect of perennial stream flow and channel morphology, provide further insight for this process. Watershed management can utilize these factors to design projects at landscape and watershed scales (Manning et al. 2007).

Figures and Tables

Figure 1: Example restoration project site at Chileno Creek. Photographic sequence documents site response following zero (top), two, and eight years (bottom) since restoration (images courtesy of Marin Resource Conservation District).

Table 1: Summary attributes of sites surveyed with mean, minimum, and maximum values.

Variable	Mean	(Min. - Max.)
Watershed Area (km.^{2})	23.5	(0.2 - 133.1)
Channel sustrate D50 (mm)	20.6	(0.01-62)
Elevation (m.)	145.3	(3.7 - 656.4)
Annual precip. (mm.)	1,019.0	(679 - 1,629)
Annual temp. (C.)	13.7	(12.0 - 15.1)
Summer max. temp. (C.)	28.07	(18.8 - 31.6)
Forested near site (%)	21.9	(0 - 100)
Clay soil texture (%)	4.7	(0.05-47.4)

Figure 2: Mean density (±1 Standard Error) of woody vegetation functional groups. Different letters indicate significant effects ($p<0.05$) between restored (n=2146) and non-restored sites (n=289) using negative binomial regression (STATA v.8.0).

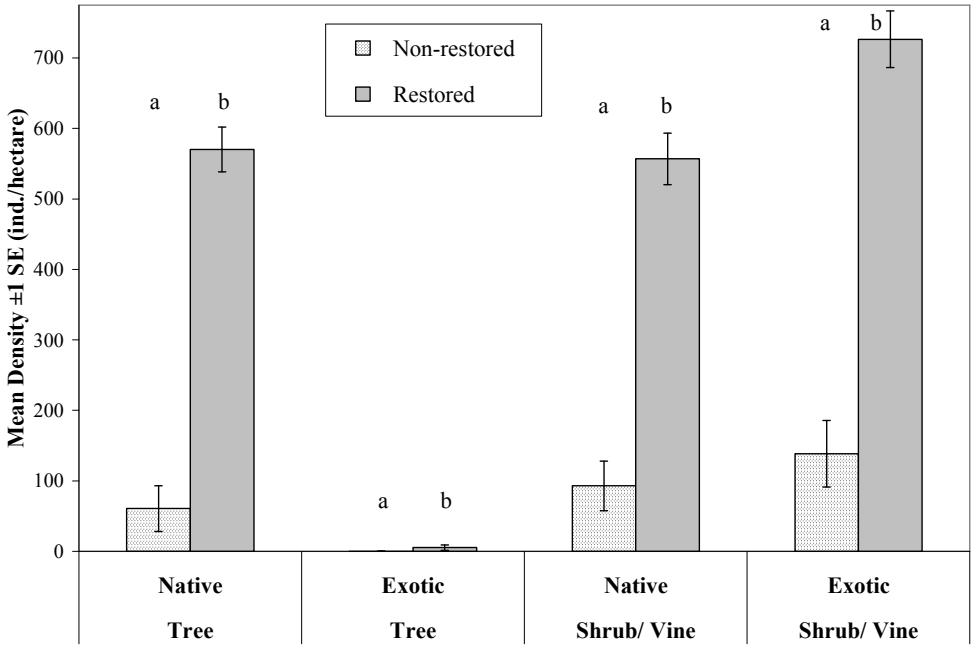


Figure 3: Trajectory of woody vegetation density by functional group – native tree (p=0.052), exotic tree (p=0.01), native shrub/ vine (p<0.001), and exotic shrub (p=0.001) - from negative binomial regression (STATA v.8.0).

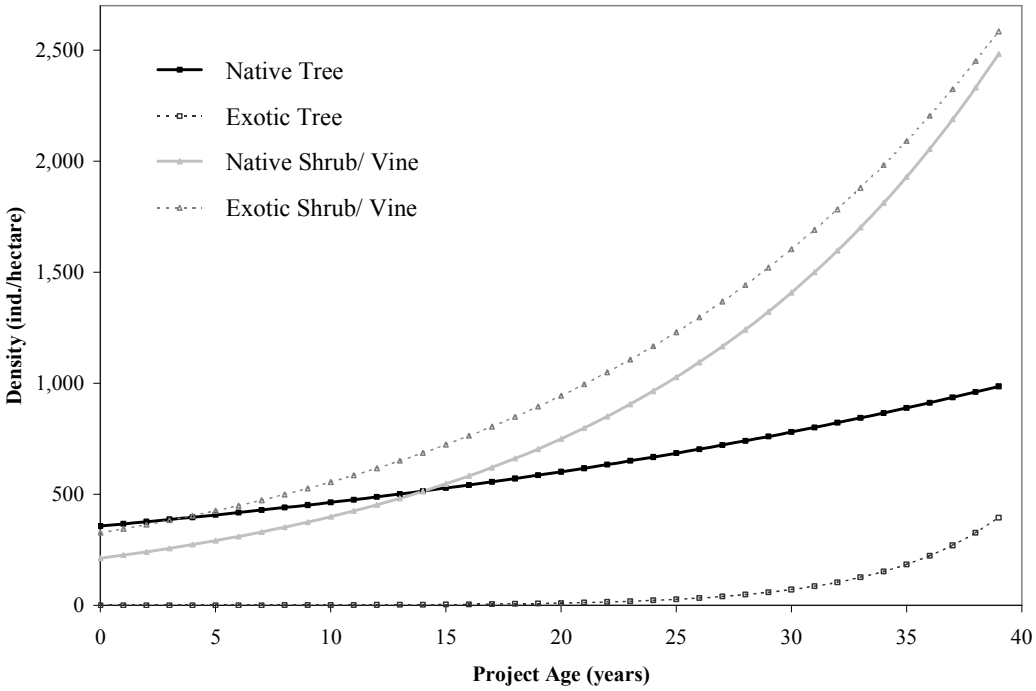


Figure 4: Native tree density, standard error and 95% confidence interval from negative binomial regression (STATA v.8.0).

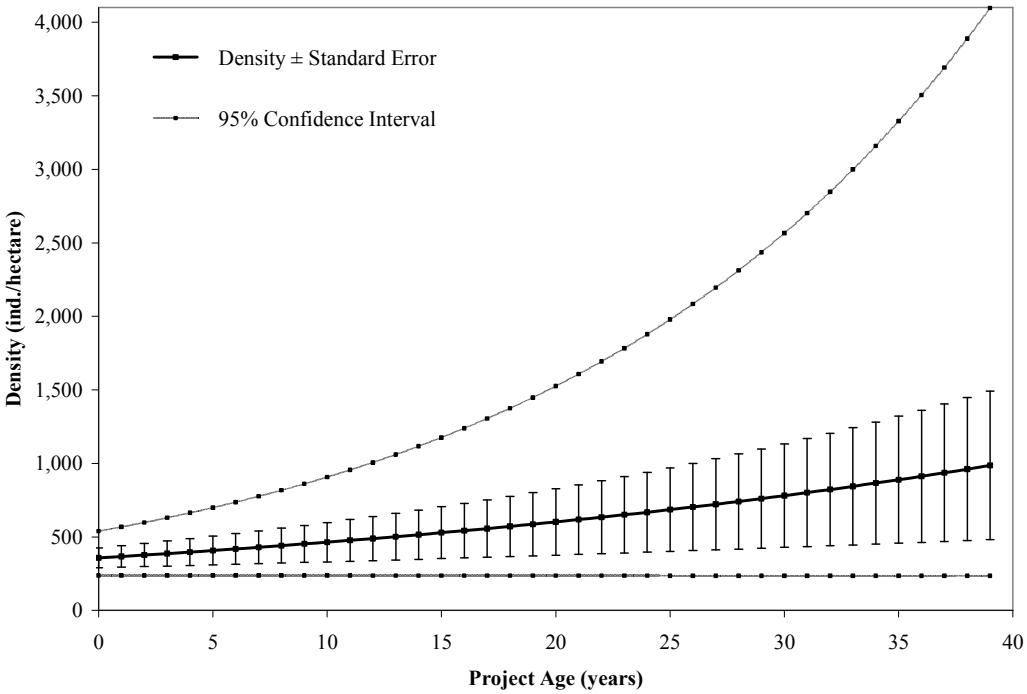


Table 2: Summary of tree species observed at restoration project sites by frequency (n=90).

Species Name	Frequency (% sites present)	Form	Origin
Salix lasiolepis	92.0%	small tree	native
Salix lucida	58.0%	tree	native
Salix laevigata	46.6%	tree	native
Fraxinus latifolia	44.3%	small tree	native
Quercus agrifolia	42.0%	tree	native
Umbellularia californica	39.8%	tree	native
Alnus rhombifolia	37.5%	tree	native
Salix exigua	30.7%	small tree	native
Quercus lobata	28.4%	tree	native
Quercus kelloggii	25.0%	tree	native
Alnus rubra	22.7%	tree	native
Acer macrophyllum	20.5%	tree	native
Aesculus californica	20.5%	small tree	native
Populus fremontii	17.0%	tree	native
Pseudotsuga menziesii	17.0%	tree	native
Arbutus menziesii	11.4%	tree	native
Sequoia sempervirens	11.4%	tree	native
Acer negundo	6.8%	tree	native
Populus balsamifera	5.7%	tree	native
Prunus cerasifera	5.7%	small tree	exotic
Acacia dealbata	3.4%	tree	exotic
Eucalyptus globulus	1.1%	tree	exotic
Pinus radiata	1.1%	tree	exotic

Table 3: Summary of shrub and vine species observed at restoration project sites by frequency (n=90).

Species Name	Frequency (% sites present)	Form	Origin
Rubus discolor	88.6%	shrub	exotic
Rubus ursinus	58.0%	shrub	native
Baccharis pilularis	53.4%	shrub	native
Toxicodendron diversilobum	46.6%	shrub	native
Rosa californica	31.8%	shrub	native
Symphocarpus albus	27.3%	shrub	native
Lonicera hispidula	17.0%	vine	native
Physocarpus capitatus	15.9%	shrub	native
Rhamnus californica	14.8%	shrub	native
Lonicera involucrata	14.8%	shrub	native
Calycanthus occidentalis	10.2%	shrub	native
Sambucus mexicana	10.2%	shrub	native
Cornus sericea	10.2%	shrub	native
Heteromeles arbutifolia	8.0%	shrub	native
Gensita monspessulana	8.0%	shrub	exotic
Crataegus douglasii	6.8%	shrub	native
Corylus cornuta	5.7%	shrub	native
Cytisus scoparius	5.7%	shrub	exotic
Hedera helix	5.7%	vine	exotic
Mimulus aurantiacus	4.5%	shrub	native
Ulex europaea	3.4%	shrub	exotic
Cercocarpus betuloides	2.3%	shrub	native
Myrica californica	2.3%	shrub	native
Sesbania tripletii	1.1%	shrub	exotic

Figure 5: Mean density (±1 Standard Error) of light diaspore tree groups. Different letters indicate significant statistical effects between each restoration treatment ($p<0.05$) using negative binomial regression (STATA v.8.0).

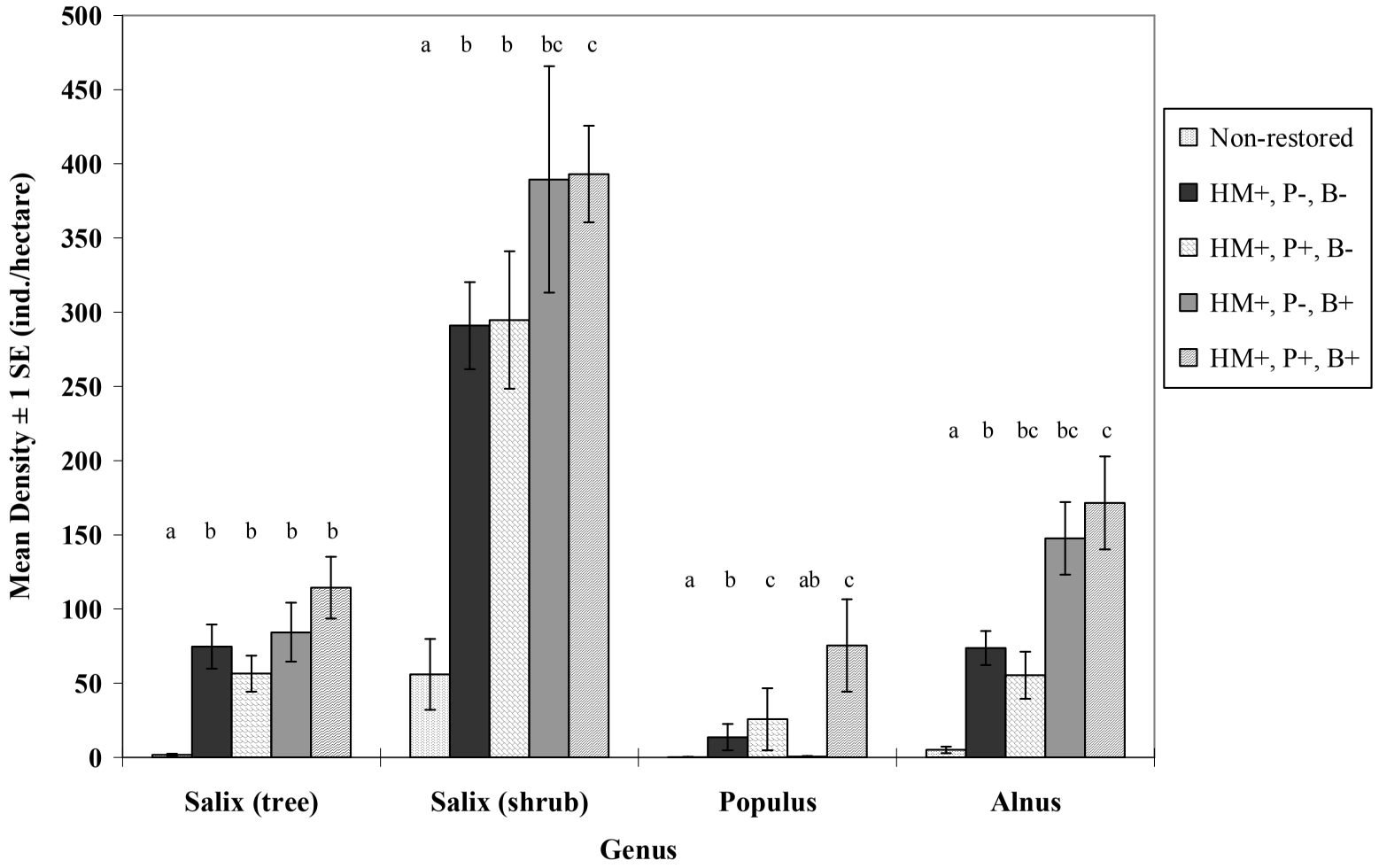


Figure 6: Mean density (±1 Standard Error) of heavy diaspore tree groups. Different letters indicate significant statistical effects between each restoration treatment ($p<0.05$) using negative binomial regression (STATA v.8.0).

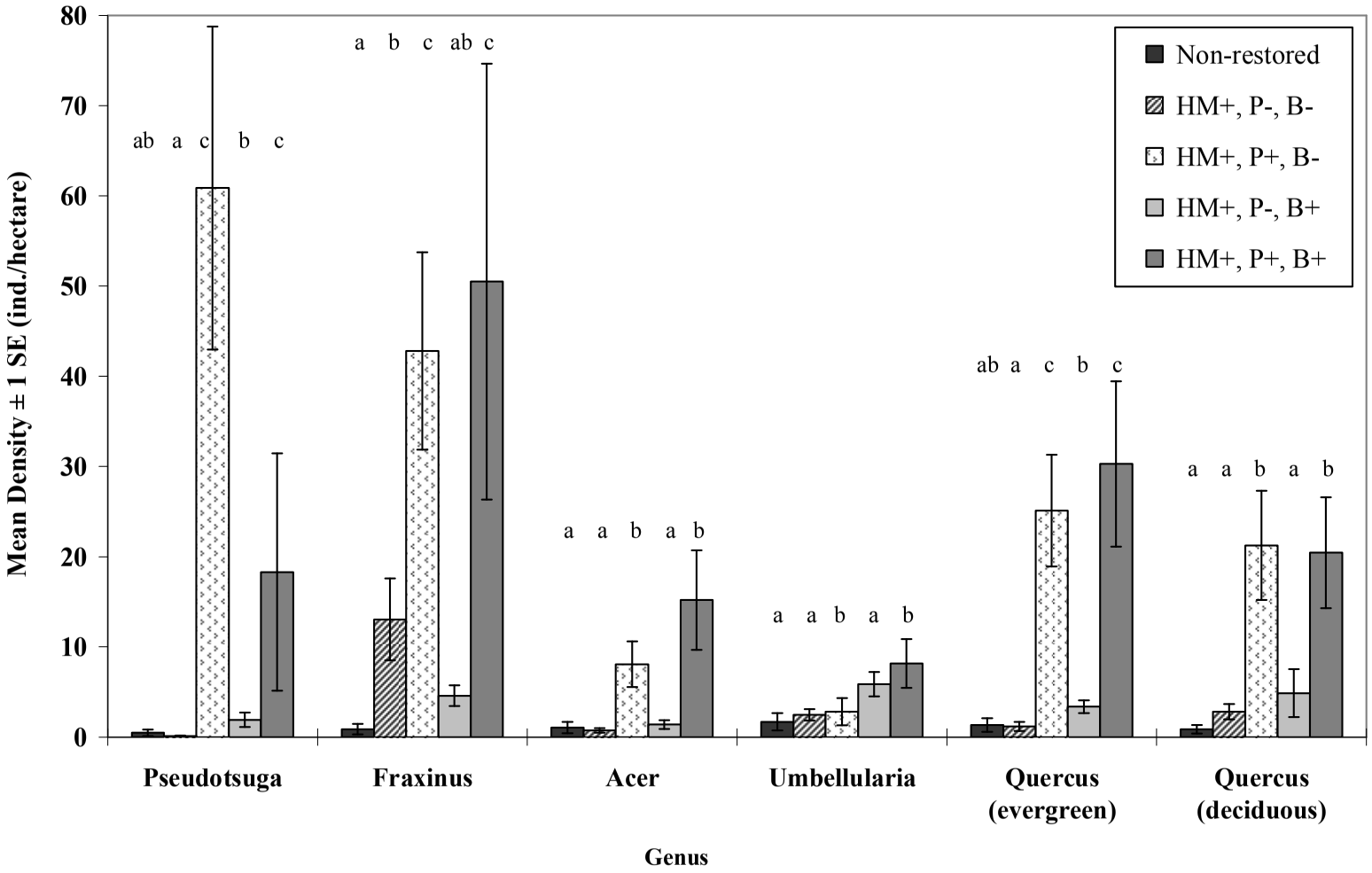


Table 4: Statistical results of negative binomial regression model by genus group. Restoration treatment regression coefficients quantify the effect of revegetation technique on the density of each genus group compared to non-restored sites (HM-, P-, B-). Predictor variables accepted in a backward stepwise process with P-value < 0.10 for a category (Intercooled Stata v.8.0).

Predictor Variables	***Salix* (tree)**		***Salix* (shrub)**		***Populus***		***Alnus***		***Pseudotsuga***	
	Coefficient (95% CI)	*P*- value	Coefficient (95% CI)	*P*- value	Coefficient (95% CI)	*P*- value	Coefficient (95% CI)	*P*- value	Coefficient (95% CI)	*P*- value
Restoration Treatment:										
HM+, P-, B-	3.19 (2.10, 4.29)	<0.001	1.50 (0.10, 2.90)	0.036	3.29 (0.10, 6.49)	0.043	1.94 (0.64, 3.25)	0.004	-0.67 (-2.43, 1.10)	0.458
HM+, P+, B-	3.34 (2.10, 4.58)	<0.001	1.60 (0.24, 2.96)	0.021	6.57 (2.96, 10.2)	<0.001	2.05 (0.53, 3.57)	0.008	5.62 (3.38, 7.86)	<0.001
HM+, P-, B+	3.33 (2.06, 4.59)	<0.001	1.79 (0.14, 3.44)	0.034	1.54 (-1.95, 5.03)	0.387	2.74 (1.20, 4.28)	<0.001	1.63 (-0.25, 3.52)	0.089
HM+, P+, B+	3.89 (2.75, 5.04)	<0.001	2.27 0.94, 3.61)	0.001	5.51 (2.47, 8.54)	<0.001	2.92 (1.42, 4.42)	<0.001	3.38 (1.63, 5.13)	<0.001
Covariate:										
Project age (years)	-	0.120	0.02 (-0.001, 0.04)	0.061	-	0.278	-	0.631	-	0.187
Clay (%)	-0.07 (-0.13, -0.003)	0.039	0.04 (0.007, 0.07)	0.016	0.05 (-0.008, 0.10)	0.096	-	0.317	0.06 (0.04, 0.08)	<0.001
Ambient temp. (C.)	-	0.732	0.11 (0.04, 0.17)	0.001	0.69 (0.27, 1.11)	0.001	-	0.672	-	0.831
Forested (%)	-	0.512	-0.01 (-0.02, -0.003)	0.007	-	0.887	0.03 (0.01, 0.04)	<0.001	0.04 (0.02, 0.05)	<0.001
Height (bankful #)	-0.35 (-0.46, -0.24)	<0.001	-0.31 (-0.37, -0.24)	<0.001	-0.25 (-0.42, -0.08)	0.004	-0.38 (-0.49, -0.27)	<0.001	-	0.776
Flow perennial	0.96 (0.23, 1.69)	0.010	0.77 (0.33, 1.21)	0.001	2.18 (0.44, 3.93)	0.014	2.18 (1.55, 2.80)	<0.001	-	0.153

Predictor Variables	***Fraxinus***		***Acer***		***Umbellularia***		***Quercus* (evergreen)**		***Quercus* (deciduous)**	
	Coefficient (95% CI)	*P*- value	Coefficient (95% CI)	*P*- value	Coefficient (95% CI)	*P*- value	Coefficient (95% CI)	*P*- value	Coefficient (95% CI)	*P*- value
Restoration Treatment:										
HM+, P-, B-	1.78 (0.25, 3.30)	0.022	-0.93 (-2.41, 0.54)	0.214	-0.05 (-1.71, 1.61)	0.950	-1.64 (-3.29, 0.008)	0.051	0.24 (-0.87, 1.35)	0.671
HM+, P+, B-	4.45 (2.82, 6.08)	<0.001	3.05 (1.44, 4.66)	<0.001	2.78 (1.04, 4.52)	0.002	1.92 (0.30, 3.55)	0.020	2.83 (1.08, 4.58)	0.002
HM+, P-, B+	1.56 (-0.04, 3.15)	0.056	-0.28 (-1.71, 1.14)	0.697	0.66 (-0.82, 2.14)	0.379	0.16 (-1.30, 1.63)	0.827	0.49 (-0.67, 1.65)	0.405
HM+, P+, B+	4.30 (2.60, 6.00)	<0.001	2.89 (1.08, 4.69)	0.002	2.07 (0.38, 3.76)	0.016	2.23 (0.61, 3.85)	0.007	2.69 (1.28, 4.10)	<0.001
Covariate:										
Project age (years)	0.09 (0.05, 0.13)	<0.001	0.08 (0.02, 0.15)	0.006	0.04 (-0.007, 0.08)	0.098	0.06 (0.03, 0.10)	0.001	0.07 (0.02, 0.13)	0.011
Clay (%)	-	0.936	-	0.489	-	0.390	0.03 (0.01, 0.05)	0.002	-	0.127
Ambient temp. (C.)	0.44 (0.26, 0.62)	<0.001	-	0.508	-	0.431	0.25 (0.10, 0.40)	0.001	0.56 (0.28, 0.83)	<0.001
Forested (%)	0.02 (0.01, 0.03)	<0.001	0.03 (0.02, 0.05)	<0.001	0.03 (0.02, 0.05)	<0.001	-	0.526	-	0.640
Height (bankful #)	-	0.603	0.21 (0.05, 0.37)	0.011	0.54 (0.41, 0.67)	<0.001	-	0.649	0.44 (0.28, 0.60)	<0.001
Flow perennial	1.27 (0.41, 2.14)	0.004	-	0.309	1.82 (0.94, 2.70)	<0.001	0.75 (0.12, 1.38)	0.020	-	0.362

Table 5: Summary of significant ($p<0.05$) and nearly significant ($p<0.10$) restoration effects with median diaspore mass by genus group (USDA 1974).

Genus Group	Statistical Effects				Seed Mass
	Passive	Planting	Bioengineering	Age	Median (mg)
Salix (tree)	↑	↑	↑	ns	0.04
Salix (shrub)	↑	↑	↑	~↑	0.06
Populus	↑	↑	ns	ns	0.53
Alnus	↑	↑	↑	ns	0.69
Pseudotsuga	ns	↑	~↑	ns	13.90
Fraxinus	↑	↑	~↑	↑	34.60
Acer	ns	↑	ns	↑	86.70
Umbellularia	ns	↑	ns	~↑	1,512.83
Quercus (evergreen)	~↑	↑	ns	↑	2,646.05
Quercus (deciduous)	ns	↑	ns	↑	4,534.32

Notes:

↑ = significant positive regression coefficient ($p<0.05$) compared to non-restored sites.

ns = not significant ($p>0.10$) regression coefficient.

~↑ = nearly significant ($p<0.10$) regression coefficient.

Figure 7: Shrub *Salix* density as a function of project age by restoration treatment with clay soil, ambient temperature, forested, perennial stream flow, and plot height constant.

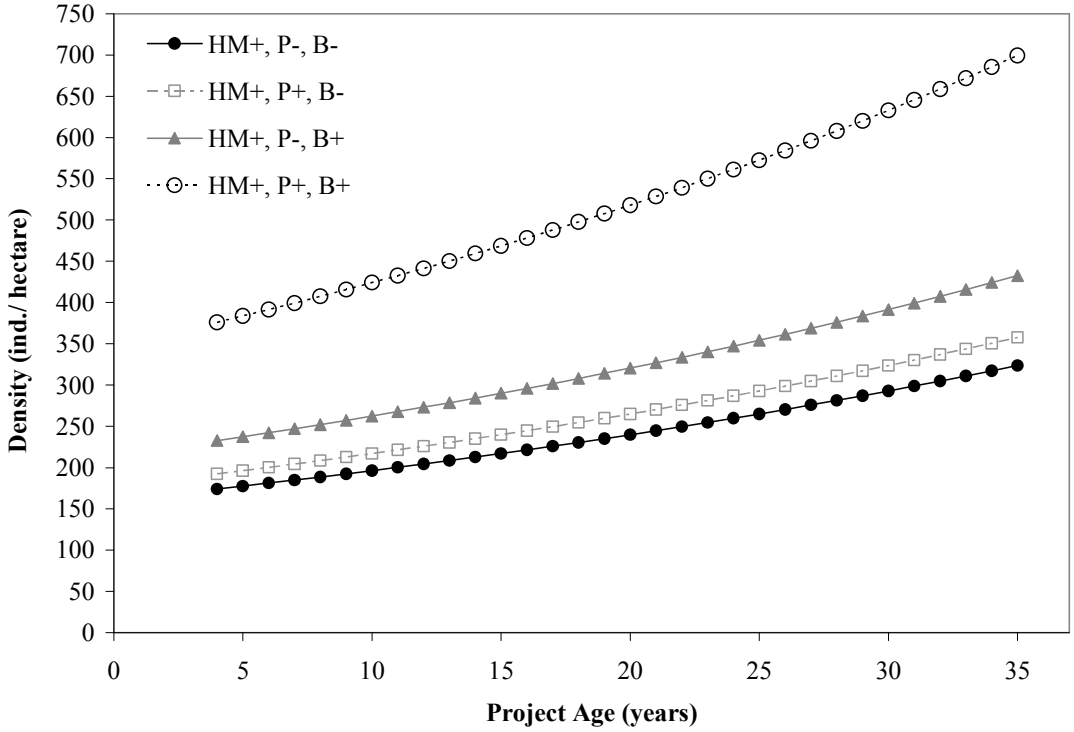


Figure 8: *Fraxinus latifolia* density as a function of project age by restoration treatment with forested, perennial stream flow, and ambient temperature constant.

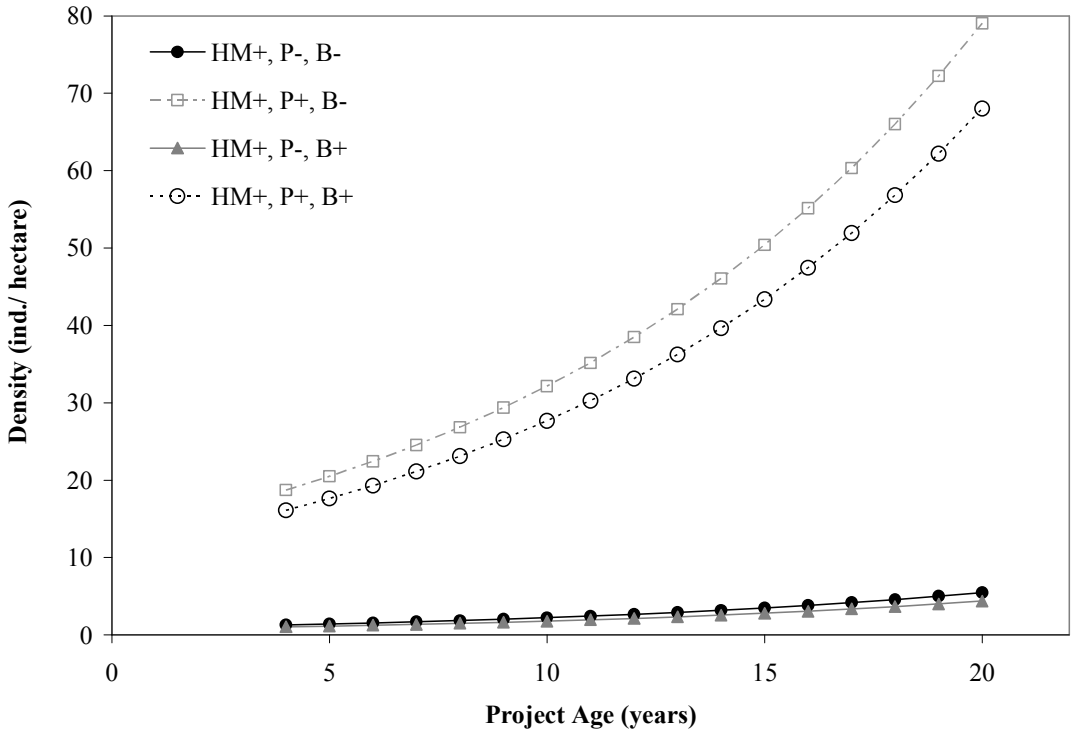


Figure 9: Evergreen *Quercus* density as a function of project age by restoration treatment with clay soil, perennial stream flow, and ambient temperature constant.

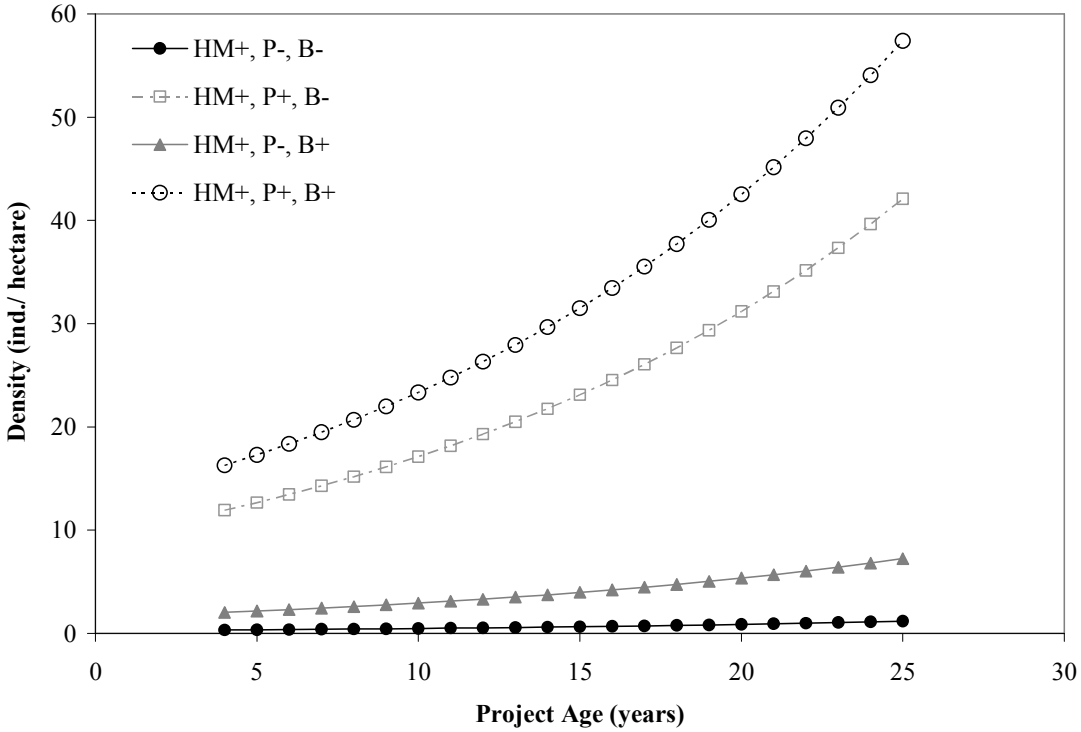


Figure 10: Deciduous *Quercus* density as a function of project age by restoration treatment with plot height and ambient temperature constant.

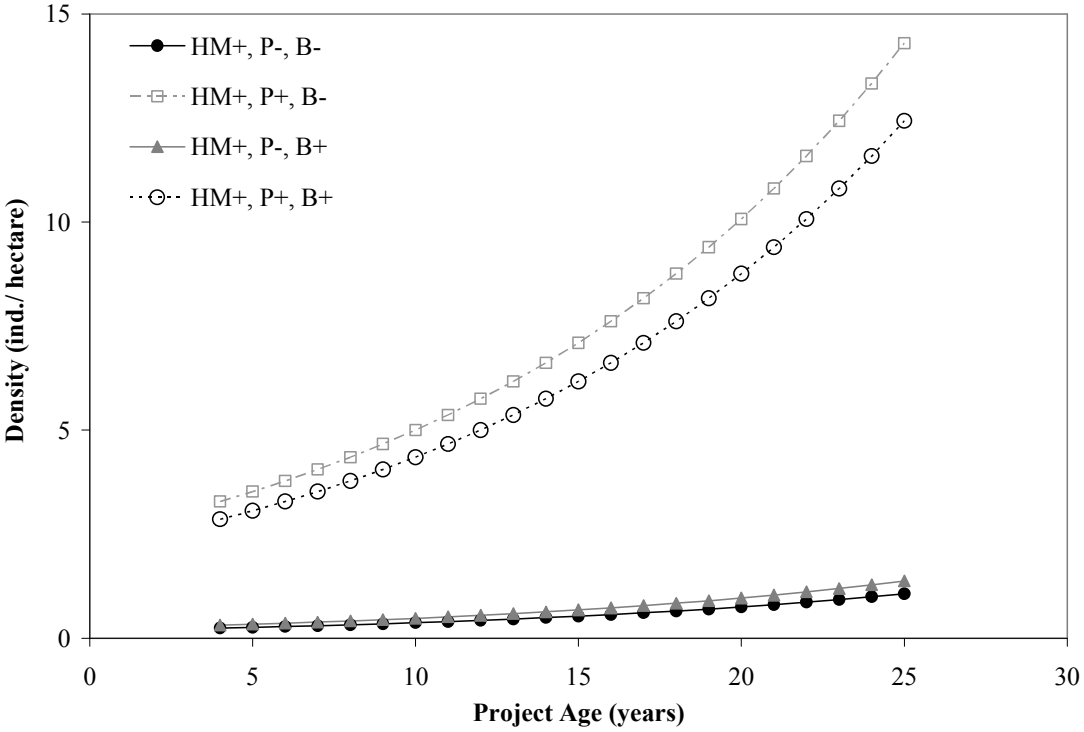


Table 6: Summary of significant ($p<0.05$) and nearly significant ($p<0.10$) physical effects by genus group.

Genus Group	Statistical Effects				
	Clay	Temp.	Forested	Height	Flow
Salix (tree)	↓	ns	ns	↓	↑
Salix (shrub)	↑	↑	↓	↓	↑
Populus	~↑	↑	ns	↓	↑
Alnus	ns	ns	↑	↓	↑
Pseudotsuga	↑	ns	↑	ns	ns
Fraxinus	ns	↑	↑	ns	↑
Acer	ns	ns	↑	↑	ns
Umbellularia	ns	ns	↑	↑	↑
Quercus (evergreen)	↑	↑	ns	ns	↑
Quercus (deciduous)	ns	↑	ns	↑	ns

Notes:

↑ = significant positive regression coefficient ($p<0.05$) compared to non-restored sites.

ns = not significant ($p>0.10$) regression coefficient.

~↑ = nearly significant ($p<0.10$) regression coefficient.

Figure 11: *Alnus* density as a function of relative plot height by restoration treatment with perennial stream flow and forested constant.

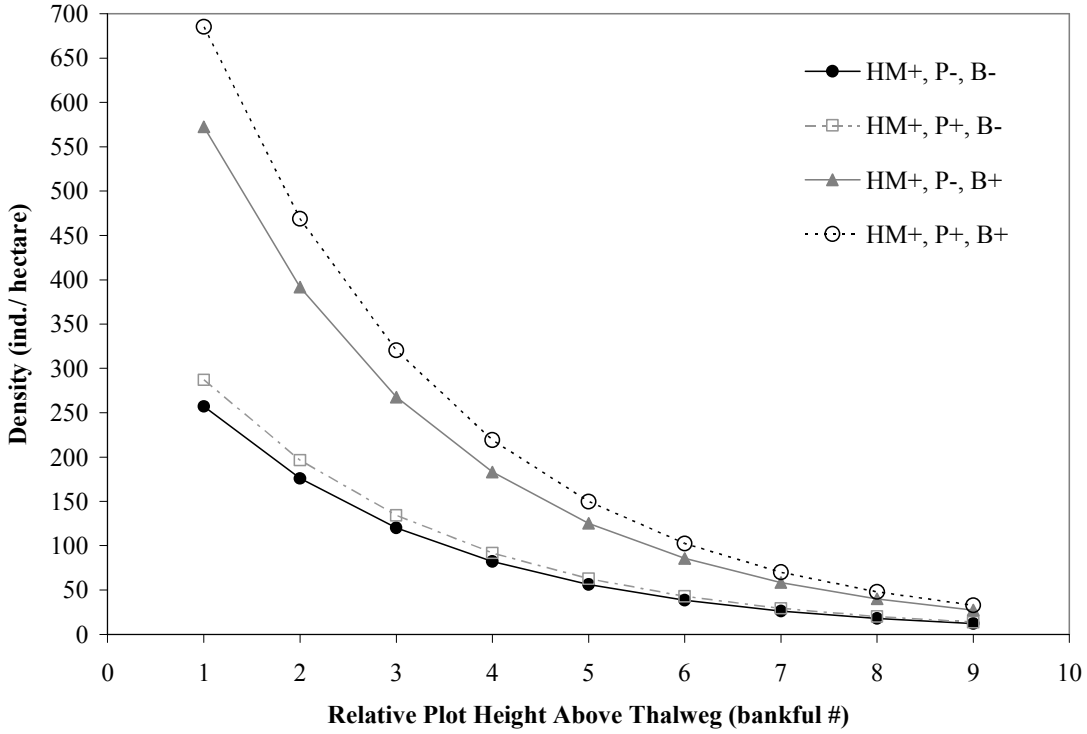


Figure 12: Tree *Salix* density as a function of clay soil particle size by relative plot height and stream flow with restoration treatment constant.

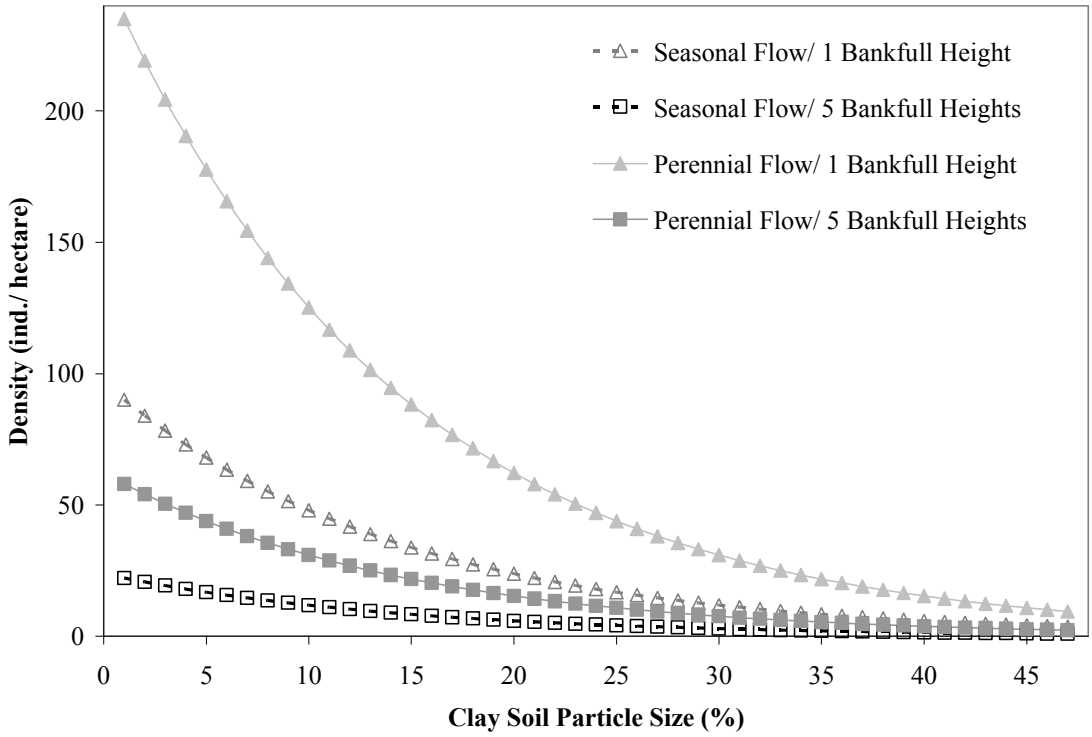


Figure 13: *Populus* density as a function of summer temperature by relative plot height and stream flow with restoration treatment constant.

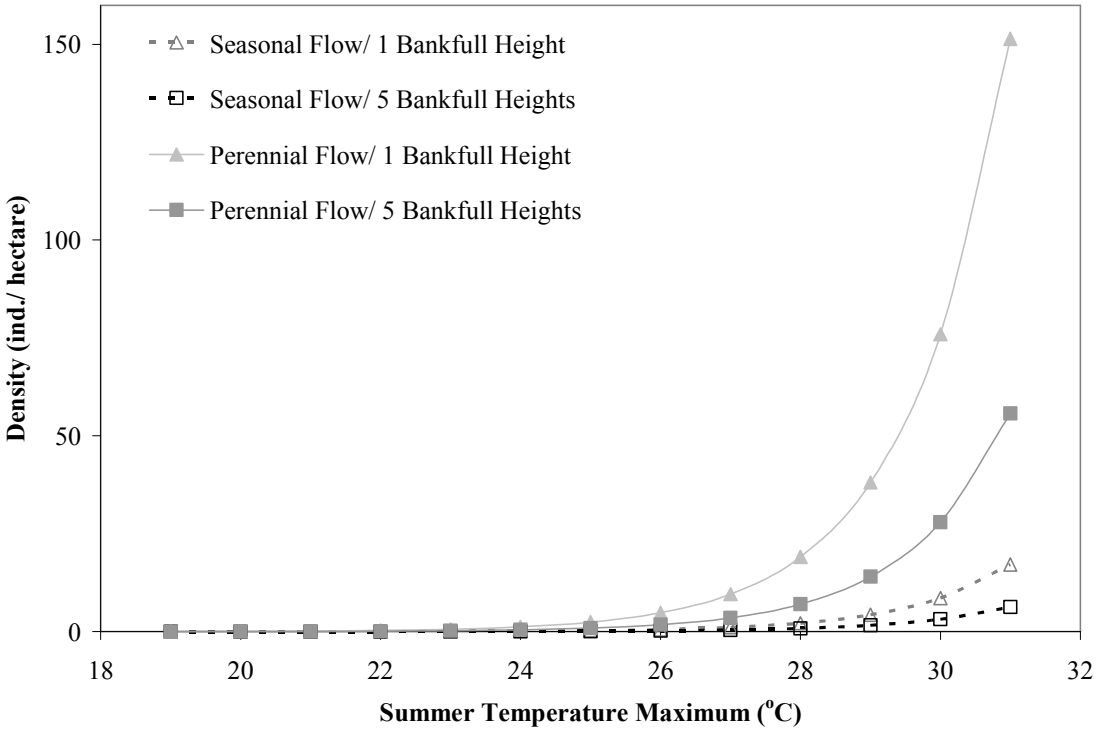
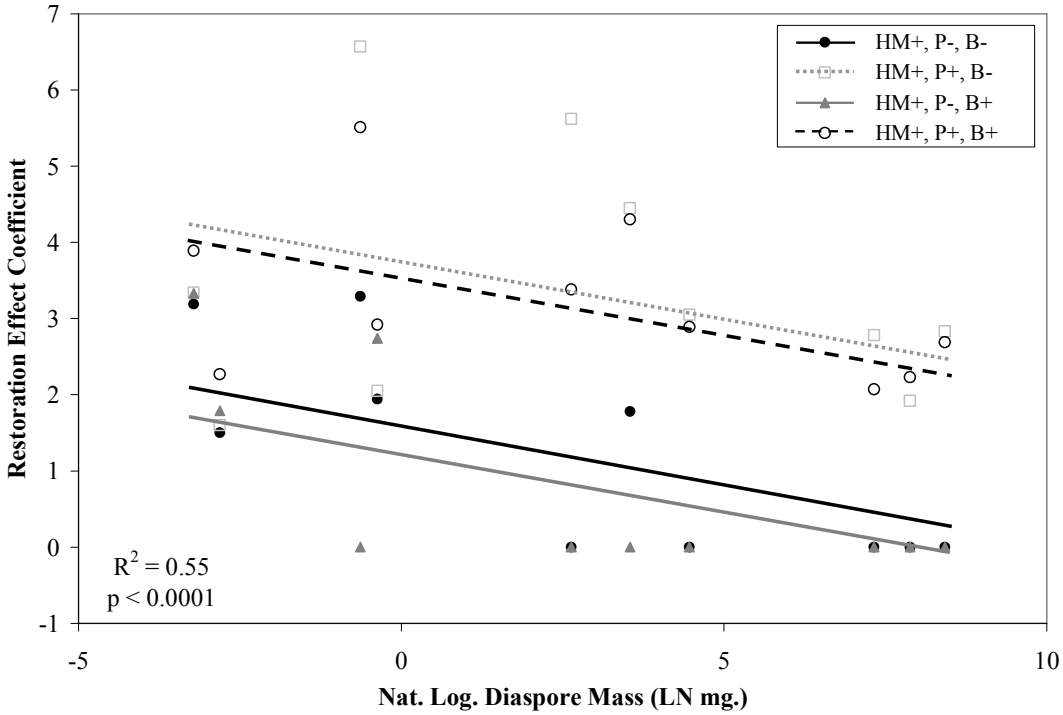


Figure 14: Restoration effect quantified by regression coefficients for each genus group by restoration treatment as a function of the natural logarithm of diaspore mass using an Analysis of Covariance statistical model (JMP 5.1) after testing for homogeneous slopes (p=0.393).



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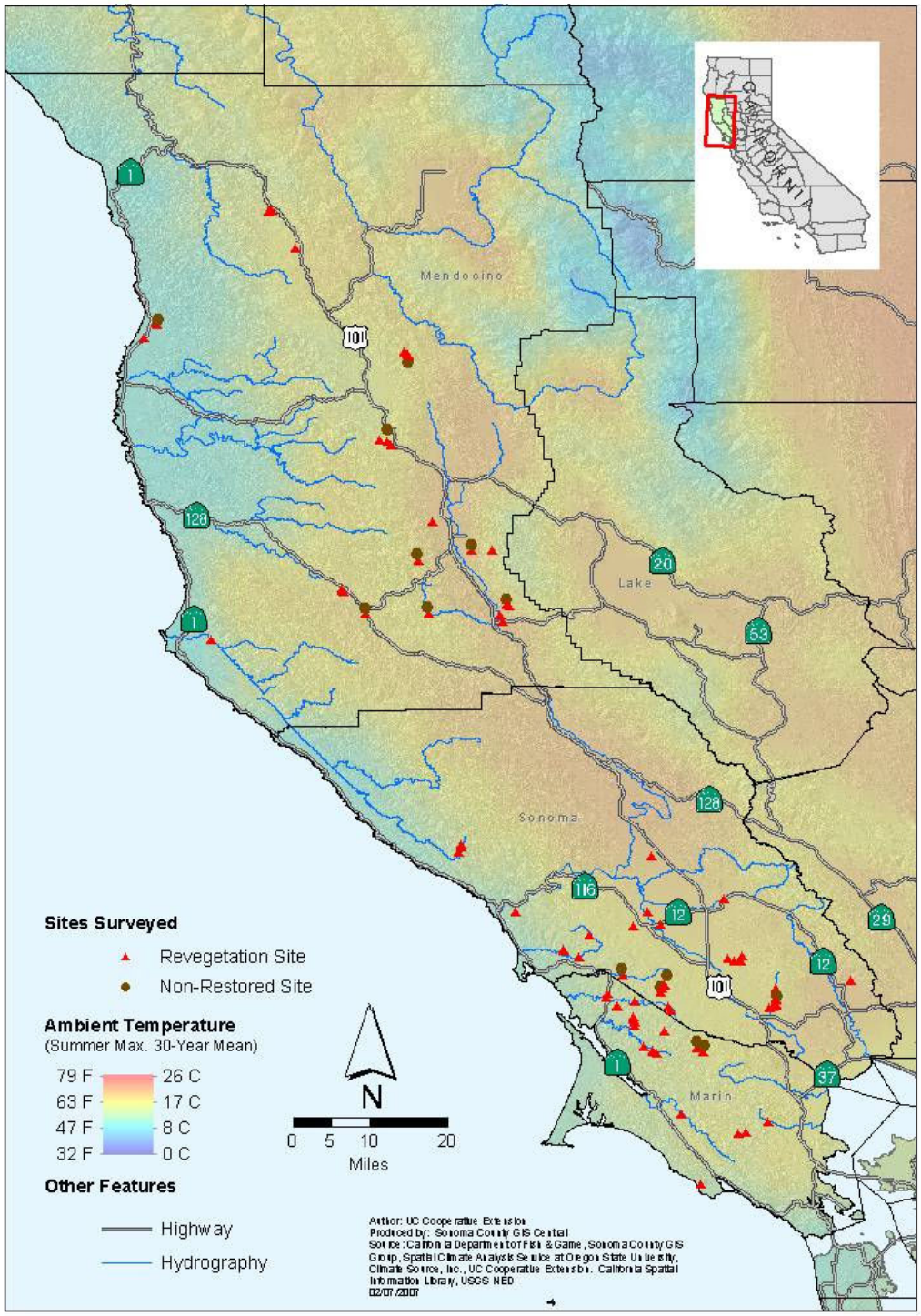
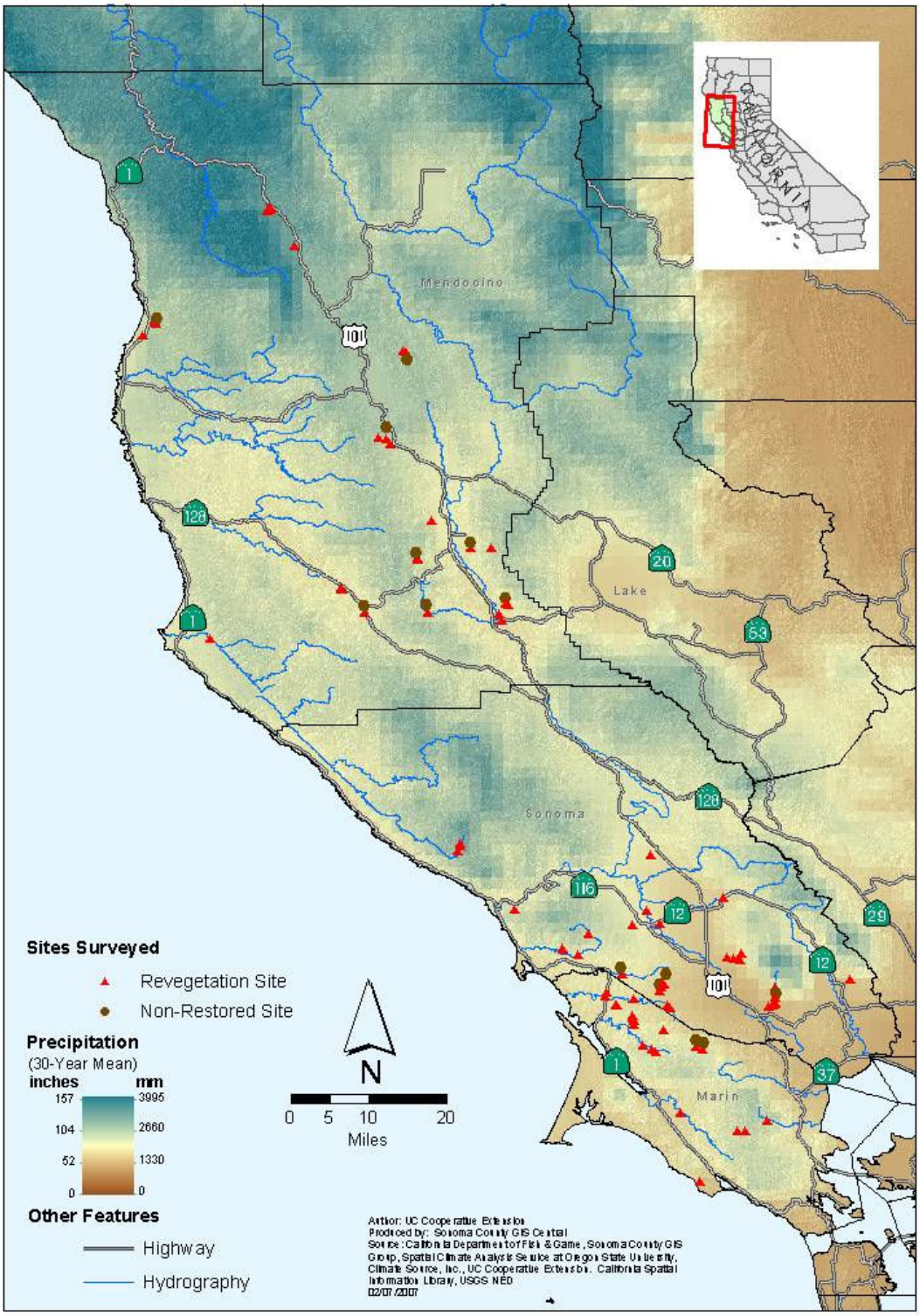
Appendix A: Maps of survey sites

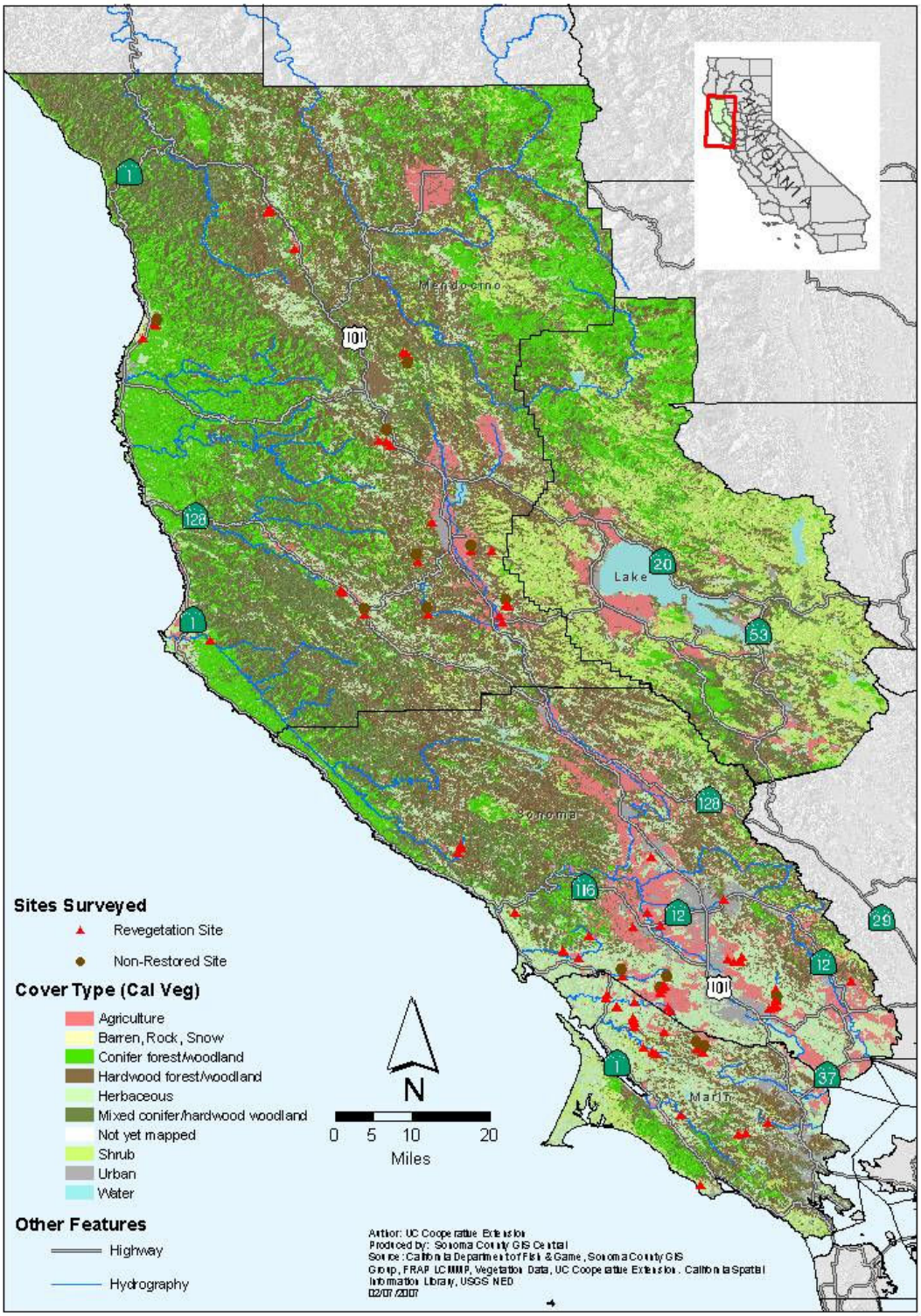
Study area including surveyed restoration projects (red) and non-restored sites (brown) with mean summer maximum temperature.



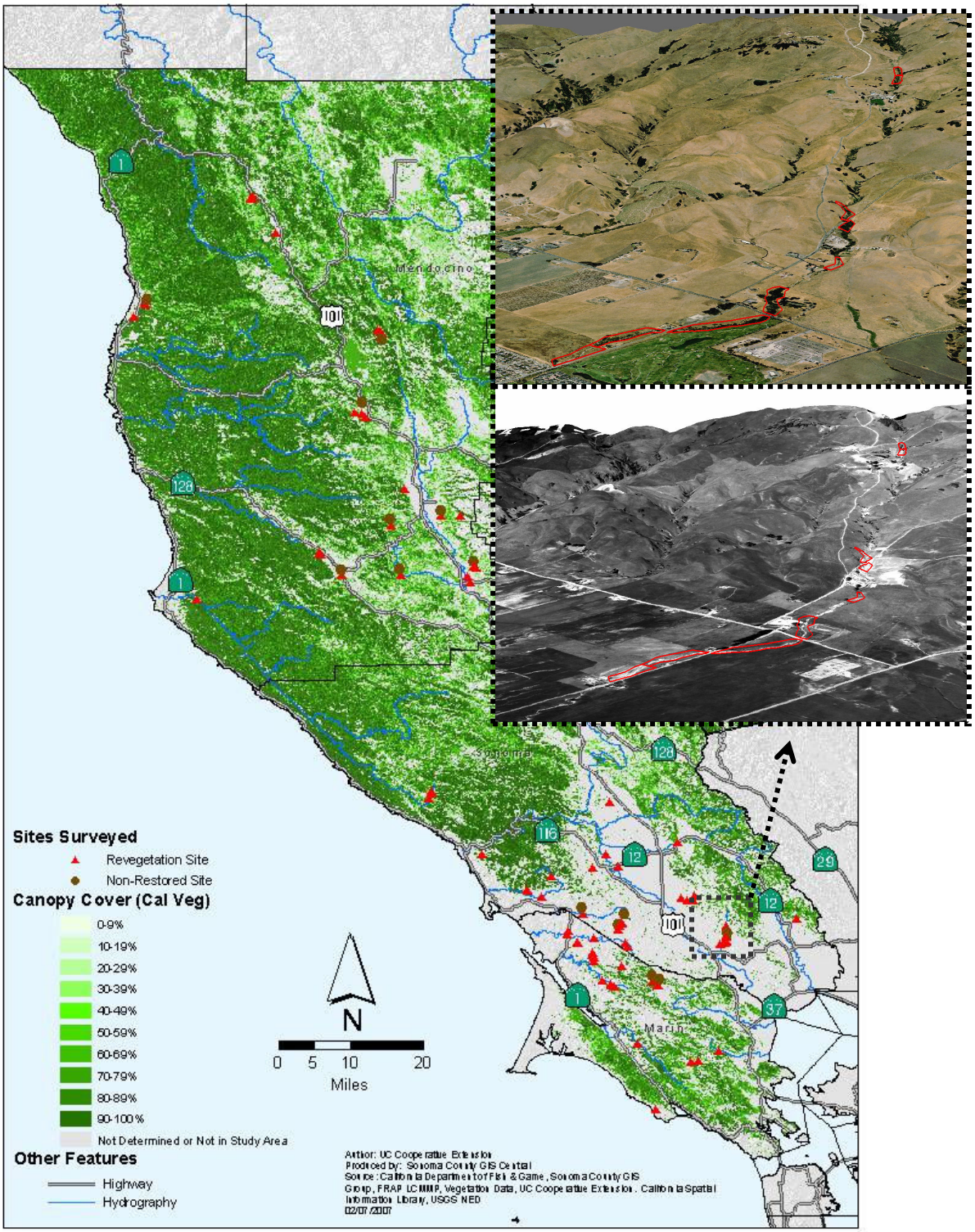
Mean annual precipitation (Climate Source 2001) over the study area with restoration sites (red) and non-restored sites (brown).



Cover type of dominant vegetation types (CDF 2005) over the study area with restoration sites (red) and non-restored sites (brown).



Canopy cover (CDF 2005) over the study area with restoration sites (red) and non-restored sites (brown). The top right images focus on the Adobe Creek watershed showing survey site boundaries over aerial photographs from 2004 (top) and 1971 (bottom).



Appendix B: Mean density with number of sites surveyed by genus group and management treatment.

Genus	Treatment	N site #	Mean Density (ind./hec. ± 1 SE)	Non-Restored *P*- value	HM+, P-, B- *P*- value	HM+, P+, B- *P*- value	HM+, P-, B+ *P*- value	HM+, P+, B+ *P*- value	FMC *P* <0.05
***Salix* (tree)**	Non-Restored	13	1.74 ± 0.75	-	<0.001	<0.001	<0.001	<0.001	**a**
	HM+, P-, B-	44	74.6 ± 14.9	<0.001	-	0.752	0.787	0.092	**b**
	HM+, P+, B-	13	56.5 ± 12.2	<0.001	0.752	-	0.980	0.259	**b**
	HM+, P-, B+	11	84.4 ± 19.9	<0.001	0.787	0.980	-	0.286	**b**
	HM+, P+, B+	21	115 ± 20.9	<0.001	0.092	0.259	0.286	-	**b**
***Salix* (shrub)**	Non-Restored	13	55.9 ± 23.9	-	0.036	0.021	0.034	0.001	**a**
	HM+, P-, B-	39	291 ± 29.3	0.036	-	0.709	0.474	0.001	**b**
	HM+, P+, B-	17	295 ± 46.3	0.021	0.709	-	0.699	0.010	**b**
	HM+, P-, B+	7	389 ± 76.3	0.034	0.474	0.699	-	0.258	**bc**
	HM+, P+, B+	26	393 ± 32.6	0.001	0.001	0.010	0.258	-	**c**
Populus	Non-Restored	13	0.16 ± 0.16	-	0.022	<0.001	0.391	<0.001	**a**
	HM+, P-, B-	54	13.7 ± 8.85	0.043	-	<0.001	0.076	0.037	**b**
	HM+, P+, B-	3	25.7 ± 20.8	<0.001	<0.001	-	<0.001	0.100	**c**
	HM+, P-, B+	24	0.62 ± 0.36	0.387	0.080	<0.001	-	<0.001	**ab**
	HM+, P+, B+	8	75.5 ± 31.1	<0.001	0.048	0.171	0.001	-	**c**
Alnus	Non-Restored	13	5.06 ± 2.27	-	0.004	0.008	<0.001	<0.001	**a**
	HM+, P-, B-	48	73.7 ± 11.4	0.004	-	0.830	0.105	0.027	**b**
	HM+, P+, B-	10	55.4 ± 15.9	0.008	0.830	-	0.268	0.146	**bc**
	HM+, P-, B+	17	147.6 ± 24.4	<0.001	0.105	0.268	-	0.779	**bc**
	HM+, P+, B+	14	172 ± 31.3	<0.001	0.027	0.146	0.779	-	**c**
Pseudotsuga	Non-Restored	13	0.48 ± 0.35	-	0.554	<0.001	0.090	<0.001	**ab**
	HM+, P-, B-	50	0.11 ± 0.07	0.458	-	<0.001	0.021	<0.001	**a**
	HM+, P+, B-	6	60.9 ± 17.9	<0.001	<0.001	-	<0.001	<0.001	**c**
	HM+, P-, B+	28	1.91 ± 0.81	0.089	0.036	<0.001	-	0.018	**b**
	HM+, P+, B+	5	18.3 ± 13.2	<0.001	0.002	<0.001	0.018	-	**d**
Fraxinus	Non-Restored	13	0.88 ± 0.58	-	0.022	<0.001	0.056	<0.001	**a**
	HM+, P-, B-	46	13.1 ± 4.54	0.022	-	<0.001	0.644	<0.001	**b**
	HM+, P+, B-	9	42.8 ± 10.9	<0.001	<0.001	-	<0.001	0.789	**c**
	HM+, P-, B+	27	4.59 ± 1.15	0.056	0.644	<0.001	-	<0.001	**ab**
	HM+, P+, B+	7	50.5 ± 24.2	<0.001	<0.001	0.789	<0.001	-	**c**
Acer	Non-Restored	13	1.06 ± 0.63	-	0.836	<0.001	0.454	<0.001	**a**
	HM+, P-, B-	47	0.74 ± 0.26	0.214	-	<0.001	0.179	<0.001	**a**
	HM+, P+, B-	8	8.08 ± 2.52	<0.001	<0.001	-	<0.001	0.890	**b**
	HM+, P-, B+	28	1.38 ± 0.49	0.697	0.179	<0.001	-	<0.001	**a**
	HM+, P+, B+	6	15.2 ± 5.52	0.002	<0.001	0.890	<0.001	-	**b**
Umbellularia	Non-Restored	13	1.70 ± 0.95	-	0.950	0.002	0.379	0.016	**a**
	HM+, P-, B-	50	2.48 ± 0.64	0.950	-	<0.001	0.096	<0.001	**a**
	HM+, P+, B-	7	2.82 ± 1.50	0.002	<0.001	-	0.001	0.271	**b**
	HM+, P-, B+	25	5.87 ± 1.37	0.379	0.096	0.001	-	0.008	**a**
	HM+, P+, B+	7	8.15 ± 2.70	0.016	<0.001	0.271	0.008	-	**b**
***Quercus* (evergreen)**	Non-Restored	13	1.35 ± 0.77	-	0.047	0.020	0.804	0.009	**ab**
	HM+, P-, B-	43	1.18 ± 0.48	0.051	-	<0.001	<0.001	<0.001	**a**
	HM+, P+, B-	12	25.1 ± 6.20	0.020	<0.001	-	<0.001	0.660	**c**
	HM+, P-, B+	25	3.36 ± 0.70	0.827	<0.001	<0.001	-	<0.001	**b**
	HM+, P+, B+	9	30.3 ± 9.16	0.007	<0.001	0.660	<0.001	-	**c**
***Quercus* (deciduous)**	Non-Restored	13	0.87 ± 0.47	-	0.671	0.002	0.405	<0.001	**a**
	HM+, P-, B-	39	2.80 ± 0.85	0.671	-	0.001	0.648	0.001	**a**
	HM+, P+, B-	16	21.3 ± 6.05	0.002	0.001	-	0.002	0.852	**b**
	HM+, P-, B+	24	4.87 ± 2.67	0.405	0.648	0.002	-	0.001	**a**
	HM+, P+, B+	10	20.4 ± 6.14	<0.001	0.001	0.852	0.001	-	**b**

Notes:
Negative binomial backward regression predictors accepted with P-value<0.100.
The full means comparison (FMC) used this matrix of probability values to find significant effects between each treatment.
Significant efffects between restoration treatments are indicated by different letters (p<0.05).