

Deer Herbivory as an Ecological Constraint to Restoration of Degraded Riparian Corridors

Jeff J. Opperman¹
Adina M. Merenlender^{1,2}

Abstract

Ungulate herbivory can impact riparian vegetation in several ways, such as by reducing vigor or reproductive output of mature plants, and through increased mortality of seedlings and saplings. Much work has focused on the effects of livestock grazing within riparian corridors, while few studies have addressed the influence of native ungulate herbivory on riparian vegetation. This study investigated the effect of deer herbivory on riparian regeneration along three streams with degraded riparian corridors in Mendocino County, California. We utilized existing stream restoration efforts by private landowners and natural resource agencies to compare six deer exclosures with six upstream control plots. Livestock were excluded from both exclosure and control plots. Three of the deer exclosures had been in place for 15 years, one for 6 years, and two for 4 years. The abundance and size distribution of woody riparian plant species such as *Salix exigua*, *S. laevigata*, *S. lasiolepis*, *Alnus rhombifolia*, and *Fraxinus latifolia* were quantified for each exclosure and control plot. The mean density of saplings in deer exclosures was $0.49 \pm 0.15/\text{m}^2$, while the mean density of saplings in control plots was $0.05 \pm 0.02/\text{m}^2$. Within exclosures, 35% of saplings were less than 1 m and 65% were greater than 1 m; within control plots, 97%

of saplings were less than 1 m in height. The fact that little regeneration had occurred in control plots suggests that deer herbivory can substantially reduce the rate of recovery of woody riparian species within degraded riparian corridors. Exclusionary fencing has demonstrated promising results for riparian restoration in a region with intense deer herbivory.

Key words: grazing, exclusionary fencing, native ungulates, *Salix* spp., *Alnus rhombifolia*, Mendocino County.

Introduction

Riparian corridors are systems of high biotic, structural, and functional diversity (Gregory et al. 1991). Along smaller streams, riparian vegetation contributes much of the energy and nutrients for aquatic food webs through allocthonous inputs of leaf litter and branches. Overhanging vegetation provides shade that maintains the lower water temperatures necessary for survival of cold-water fish species (Meehan et al. 1977; Vannote et al. 1980; Barton et al. 1985). The input of larger branches and trunks creates instream structure and habitat, and this large woody debris plays a major role in channel-forming processes, such as pool formation and gravel bar stabilization (Abbe & Montgomery 1996). Riparian vegetation contributes to bank stability through root systems that anchor soil and by increasing roughness to slow the velocity of high flows (Kondolf & Curry 1986).

In the western United States, the extent of riparian ecosystems has been considerably reduced, and remaining habitats are often highly degraded or fragmented by a variety of human activities (National Research Council 1992; Kondolf et al. 1996). A degraded riparian zone can be defined as one that lacks the capacity to provide ecosystem functions such as bank stability, maintenance of water temperatures and stream flows, and habitat features (U.S. Department of the Interior, Bureau of Land Management [BLM] 1993; Kauffman et al., 1997). Grazing by livestock has been implicated in the decline of riparian forests (Keller & Burnham 1982; Platts & Wagstaff 1984; Knapp & Matthews 1996). Livestock can compact soil, exacerbate bank erosion, and consume seedlings and saplings of woody riparian species (Platts 1991; Fleischner 1994). Armour et al. (1994) estimate that 50% of Western riparian corridors are degraded due to livestock grazing. In the western United States, the realization that riparian degradation has contributed to the decline of anadromous fisheries has prompted much interest in the protection and restoration of these systems (Meehan et al. 1977; National Research Council 1992).

Techniques for riparian restoration include planting of riparian woody species, irrigation, and channel modification (Briggs et al. 1994; Kauffman et al. 1995). A key com-

¹Environmental Science, Policy, and Management, University of California, Berkeley, CA 94720-3110, U.S.A.

²Address correspondence to A. Merenlender, Ecosystem Science Division, University of California, Berkeley, Department of Environmental Science, Policy, and Management, 151 Hillgard Hall, Berkeley, CA 94720-3110, U.S.A.

ponent of any successful restoration is the identification of stressors that are contributing to the decline of the system or preventing system recovery. The failure to address such stressors will often render other restoration efforts ineffective (Briggs et al. 1994; Kauffman et al. 1997). Grazing pressure is an example of a stressor that can prevent recovery of a riparian corridor. Many studies have documented vigorous growth of riparian vegetation following the elimination of livestock grazing (Briggs et al. 1994; Green & Kauffman 1995; Kauffman et al. 1995).

In Mendocino County, California, riparian corridors on several streams that had been degraded due to livestock overgrazing did not recover following the removal of livestock. Biologists from state agencies believed that herbivory from *Odocoileus hemionus columbianus* (black-tailed deer) may have been responsible for the slow response. This hypothesis was based on direct observations of the impact of deer herbivory on regenerating riparian vegetation, as well as observation of greater growth of riparian vegetation in areas that were not accessible to deer (J. Booth, California Department of Fish and Game (CDFG), personal communication). To address this perceived impact, deer exclosures were erected by landowners and resource agencies on three streams in the upper Russian River watershed.

This study assessed the response of the woody vegetation within these exclosures to the elimination of deer herbivory. The riparian corridors of the study sites have not been grazed by livestock while exclosures have been in place, and, thus, these projects allow the quantification of the effect of wild ungulate herbivory. The goal of this research was to assess the results of existing restoration projects implemented by private landowners and state resource agencies. Although restoration projects are generally not designed as experiments (e.g., they do not have true replication or random assignment of treatments), they provide a source for opportunistic study of ecological processes and the efficacy of restoration strategies; every attempt should be made to learn from both successes and failures (Kondolf 1995). This approach ensures that science is addressing practical applications of restoration and providing insight to improve project effectiveness.

Study Sites

This study utilized exclosures on Parsons, Robinson, and Feliz Creeks, located within the upper Russian River watershed, in the North Coast Range of Mendocino County, California (Fig. 1). The region has a Mediterranean climate with cool, wet winters and hot, dry summers. Yearly precipitation averages 95 cm, with the majority falling as rain in late autumn and winter. Major land uses in the region include logging, sheep and cattle grazing, orchards, and vineyards. Many riparian corri-

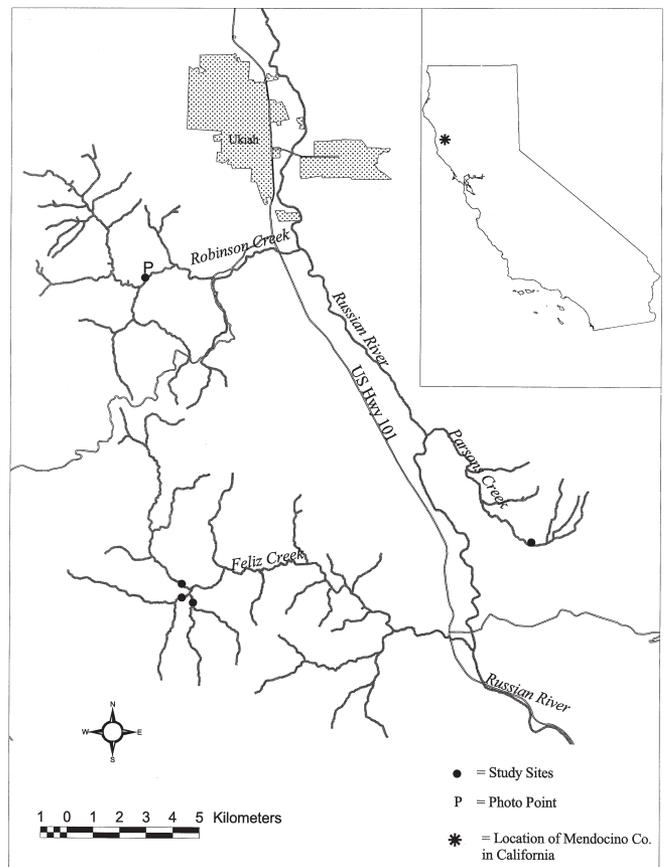


Figure 1. Map of Feliz, Parsons, and Robinson creeks, within the upper Russian River watershed, Mendocino County, California.

dors in the area have been cleared for agriculture or have become degraded due to livestock overgrazing.

Six deer exclosure sites were compared with control sites on Parsons, Robinson, and Feliz Creeks. Since the fenced creek exclosures were placed by individual landowners the sites were not selected according to random experimental design protocol. Parsons Creek is a second-order stream east of the Russian River. In the winter of 1992–93, the Hopland Research and Extension Center (HREC) of the University of California implemented a demonstration restoration project on the riparian corridor of Parsons Creek. Treatments included sheep exclosures, deer exclosures, and planting of riparian woody species. Deer exclosures have fences 2 m tall and also exclude sheep. Fencing for sheep exclusion does not exclude deer. For this study, two deer exclosures and two control plots excluding sheep only were compared. Thus, the only ungulate herbivores within control plots were deer. Neither deer nor sheep fencing excludes rodent or invertebrate herbivores. Plots were rectangular in shape and parallel to the channel, with the average plot size 1,000 m².

Robinson Creek is a third-order stream on the western side of the Russian River Valley. Fencing to exclude deer was constructed on a 300-m long section of the creek's riparian corridor in 1991, which encompassed roughly 2 ha of the riparian corridor. Although this property has not been grazed by livestock for many years, little natural regeneration of woody riparian species had occurred prior to the construction of the deer exclosures. Vegetation within this fenced area was sampled and compared to an unfenced control area upstream of approximately equal size. Both the fenced and control areas are under the same ownership and management.

Feliz Creek is a third-order stream also on the west side of the valley. The north, middle, and south forks (second-order streams) of Feliz Creek converge on the property of a single landowner. He erected deer exclosures on each of the three forks, as well as a portion of the main stem. The fenced area of the riparian corridor on the south fork was 54 m long by 10 m wide (5 m on each side of the stream). The fenced area of the north fork was 300 m long by 24 m wide (12 m on each side of the stream). The fenced area of the middle fork was 225 m long by 24 m wide (12 m on each side of the stream). These fences were constructed between 1980 and 1982. Each of these three fenced plots was surveyed and compared to an unfenced control section upstream on the same property. This property has not been grazed by livestock for the past 18 years.

Methods

Plots on the three streams were surveyed for woody vegetation during June and July 1997. Regeneration at the sites consisted primarily of saplings of *Salix exigua* (narrow-leaved willow), *S. laevigata* (red willow), *S. lasiolepis* (arroyo willow), *Alnus rhombifolia* (white alder), and *Fraxinus latifolia* (Oregon ash). Saplings were placed into two size classes, based on height and number of stems (U.S. Department of Agriculture, Forest Service 1992). Size class 1 included saplings less than 1 m tall with branching, woody stems or obvious new growth. Saplings greater than 1 m tall or with five or more stems were placed in size class 2. The density of saplings within a plot was determined either by a complete survey or estimated by sampling (described below). We elected to use density rather than percent cover as a measurement of recruitment; tree density is a more precise measurement and is easier to quantify in sparsely vegetated stream reaches with few or heavily browsed plants, as was the case in several of our plots.

Within the Parsons Creek plots all woody stems were surveyed. The area of suitable substrate for riparian regeneration (i.e., non-rock groundcover) within each

treatment was measured with a tape in order to calculate sapling density. Some of the plots at Parsons Creek had been planted with willow and alder in 1994. Therefore, a planted deer exclosure plot was paired with a planted control plot upstream of the fence, and an unplanted deer exclosure plot was paired with an unplanted control. To avoid counting planted individuals as natural regeneration, size class 2 saplings were recorded but not included in the data for either of the planted plots. This was a conservative choice because few planted individuals (<6%) survived into 1997 (J. Opperman, unpublished data), and it is therefore likely that many of the size class 2 saplings were from natural regeneration. High mortality of planted trees was attributed to scour during floods and drought stress (R. Keiffer, HREC, personal communication).

The exclosure on the south fork of Feliz Creek was relatively small, permitting a complete survey of woody vegetation using the same methods described for the Parsons Creek plots. The two other deer exclosures on Feliz Creek and the exclosure on Robinson Creek were much larger and, consequently, woody regeneration within these plots was sampled along transects. Four-meter wide belt transects were extended from the wetted channel to the beginning of upland vegetation perpendicular to the stream. Plots at Feliz Creek were sampled every 15 m of channel length, and plots at Robinson Creek were sampled every 20 m of channel length. The variable intervals were chosen such that approximately thirty transects could be placed within each plot, fifteen on each side of the channel. Transect length varied due to differences in topography, with a mean length of 13.2 m. Control plots were selected at Feliz and Robinson creeks; unfenced reaches with similar channel type and geomorphology to that found in the exclosure were sampled upstream of the exclosures using the same methods. The control plots were located within the same property boundaries as the exclosures and were under the same land management prior to, and following, the construction of exclosures.

Treatment and control sites were not randomly assigned. Treatment sites had already been determined by the landowners who implemented the restoration projects. Controls were placed adjacent, or nearly adjacent, to the treatment sites in order to minimize differences between confounding variables such as flow regime, seed source, geomorphology, and land ownership and management. Controls were placed upstream of treatments in order to avoid areas potentially influenced by the treatment. For example, vigorous vegetation within exclosures may have served as an abundant seed source for downstream reaches.

The amount and type of vegetation existing prior to fencing was not documented in either the exclosures or control plots. However, the property owners and others

involved with the projects (e.g., consultants, agency personnel) described the exclosures and control reaches as being quite similar, with little or no riparian vegetation prior to erecting exclosures. Additionally, photographs of the Robinson Creek project during its first year showed both exclosures and control reaches with a sparsely vegetated riparian corridor. The effect of fencing on riparian regeneration was tested using a paired sample *t*-test, with the six exclosure/control pairs as replicates.

Results

Density of regenerated saplings was greater in fenced plots than in control plots ($p \leq 0.015$, $df = 5$; Fig. 2). Mean density within the six exclosures was 0.49 ± 0.15 saplings/m², compared to mean density within control plots of 0.05 ± 0.02 saplings/m². Comparing regeneration within exclosures, density was lowest in the Parsons Creek plots, highest in the Robinson Creek plot, and intermediate in the Feliz Creek plots (Fig. 2).

The species and age classes of sapling regeneration varied between streams (Figs. 2 and 3). Regeneration in the four Parsons Creek plots was composed almost entirely of *S. exigua* and *S. laevigata*. Regeneration within the Robinson Creek exclosure was primarily *S. exigua*, *S. laevigata*, *S. lasiolepis*, and *Alnus rhombifolia*. Primarily size class 1 saplings of the same three *Salix* spp. were recorded in the Robinson Creek control plot, with a smaller component of size class 2 saplings of *S. lasiolepis*. The middle and south fork exclosures of Feliz Creek were com-

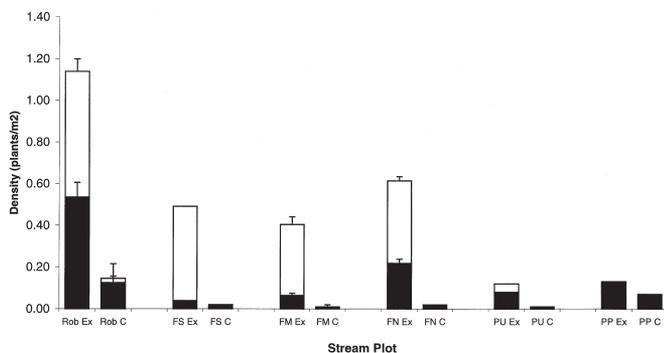


Figure 2. Density and size class distribution of riparian woody sapling regeneration within exclosure (Ex) plots and control plots (C). Size class 1 saplings are represented in the black portion of each bar and size class 2 saplings are in the white portion of each bar. Rob = Robinson Creek; FS = south fork of Feliz Creek; FM = middle fork of Feliz Creek; FN = north fork of Feliz Creek; PU = Parsons Creek unplanted plots; PP = Parsons Creek planted plots. Feliz North, Feliz Middle, and Robinson plots were sampled (standard error bars shown); all vegetation was recorded in Feliz South, Parsons unplanted, and Parsons planted plots.

posed of a near uniform canopy of size class 2 *A. rhombifolia* with a small understory component of *F. latifolia*. Regeneration within the north fork was composed of *S. exigua*, *S. laevigata*, *S. lasiolepis*, *A. rhombifolia*, *F. latifolia*, as well as a small component of other species (including *Populus fremontii* [Fremont cottonwood] and *Acer macrophyllum* [big-leaf maple]). Regeneration within the three Feliz Creek control plots was composed primarily of size class 1 *A. rhombifolia*.

High densities of *A. rhombifolia* and *Salix* spp. seedlings were found within all control plots, indicating that seedling establishment is occurring. Very few saplings were found in any of the controls and those that were found were primarily of the smaller size class: 97% of saplings found in control plots were size class 1 and 3% were size class 2. Within exclosures, 35% of saplings were size class 1 and 65% were size class 2. Many of the saplings found in the control plots displayed leaf and stem damage characteristic of deer browsing. For example, nearly every size class 2 *Salix* spp. sapling within the Robinson Creek control plot displayed signs of heavy browsing. Within the Feliz Creek control plots, *A. rhombifolia* saplings were found only in a prostrate, bushy form, indicative of repeated browsing.

Discussion

In Mendocino County, several creeks with sparsely vegetated riparian corridors responded to the removal of livestock with limited recruitment of woody vegetation. The lack of recovery did not meet expectations

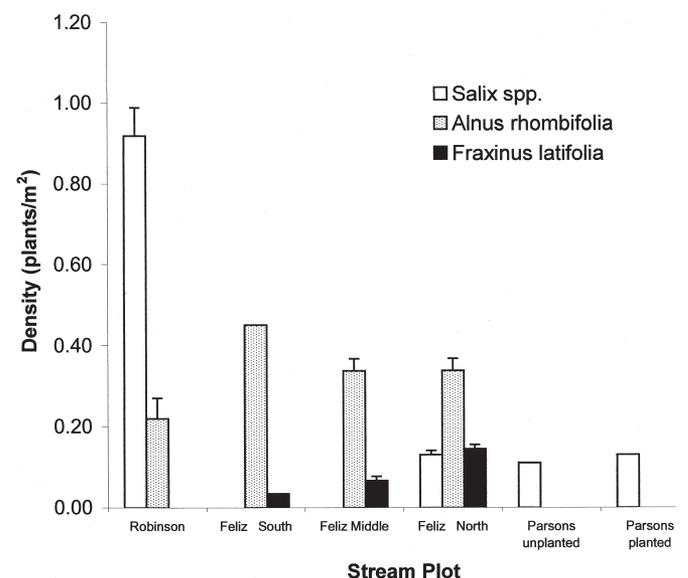


Figure 3. Comparisons between the species composition of woody riparian regeneration within six deer exclosures. *Salix* spp. includes *S. exigua*, *S. laevigata*, and *S. lasiolepis*.



Figure 4. Robinson Creek riparian corridor within deer exclosure. Fence line can be seen in the foreground. The photograph shows natural regeneration of riparian woody species 6 years after construction of the exclosure. This vegetation is primarily composed of *Alnus rhombifolia*, *Salix laevigata*, and *Salix exigua*.

based on results from other regions, where considerable regeneration of riparian vegetation has occurred following the cessation of livestock grazing (Briggs et al. 1994; Green & Kauffman 1995; Kauffman et al. 1995). Six deer exclosures were constructed on three streams in an attempt to facilitate riparian recovery by eliminating deer herbivory. The density of sapling regeneration was greater within all six deer exclosures than in the paired upstream control plots. These results indicate that herbivory by black-tailed deer may be significantly retarding or preventing the regeneration of riparian vegetation within this region.

The recovery of vegetation within deer exclosures on Robinson and Feliz Creeks was quite rapid with considerable growth observed after 2 years (D. Meda, landowner, personal communication; R. Morris, landowner, personal communication). Currently, the streams within the exclosures have a continuous riparian corridor. The density of regeneration recorded in this study appeared to be higher at Robinson Creek than Feliz Creek due to the differences in tree age and size. The exclosure at Robinson Creek has been in place for 6 years and, therefore, the regeneration consists of dense stands of young saplings (Fig. 4). Exclosures on Feliz Creek have been in place for 15 years, and the trees are much larger (e.g., greater average height and dbh) and more widely spaced. Control plots on Feliz and Robinson Creeks had little growth of riparian vegetation, and very few saplings in the larger size class were found at either stream (Fig. 5).

The density of regeneration within the Parsons Creek exclosures was much lower compared to the other streams. This may be attributed to more harsh abiotic conditions for establishment and survival within the Parsons Creek ripar-



Figure 5. Robinson Creek control site upstream of the deer exclosure.

ian corridor. The plot reaches on Parsons Creek become dry in May or June of most years, whereas the plot reaches on Feliz and Robinson Creeks generally flow year round. Seedlings of many riparian species (e.g., *Salix* spp.) require contact with the water table during the growing season (McBride & Strahan 1984; Braatne et al. 1996). Seedling mortality due to drought stress may result if the water table at Parsons Creek declines too quickly for seedlings' root systems to maintain contact with water. Working in the Russian River basin, McBride and Strahan (1984) found that 63% of seedlings along reaches with surface water survived through the summer, while 16.5% of seedlings survived along reaches that dried out by the end of the summer.

Much work on riparian restoration has focused on the effects of livestock fencing on regeneration (Briggs et al. 1994; Green & Kauffman 1995; Kauffman et al. 1995). However, few studies have considered the role that native ungulate herbivory plays in these systems and to what extent this herbivory may be limiting recovery of degraded systems. Case and Kauffman (1997) compared growth of riparian woody species within deer and elk exclosures to growth outside of the exclosures along a stream in northeast Oregon. Livestock had been removed from the riparian corridor prior to the study. They observed marked differences in crown volume, height, and willow catkin production. During their two year study, the mean height growth of existing woody riparian plants within the exclosures was 47 ± 6 cm, compared to 16 ± 4 cm in the controls. Within the exclosures, 34% of willows produced catkins, while only 2% within the controls did so. Crown volume of willows within the exclosures increased 550%, compared to 195% outside. Kay (1995, 1997) and Kay and Chadde (1992) report significant effects of elk herbivory on willows in Yellowstone National Park's northern range. The mean height of willows within long-term exclosures was 274

cm, compared to 34 cm in controls. They found an average of 307,000 seeds/m² of willow canopy within exclosures, while no seed production was observed outside the exclosures. Also working in Yellowstone, Keigley (1997) hypothesized that herbivory by native ungulates may eliminate cottonwoods in the park's northern range.

While based on a relatively small sample size, the results of this study indicate that herbivory by deer is severely limiting to natural riparian regeneration on these streams; regeneration density of woody riparian species within deer exclosures was approximately ten times greater than regeneration density within controls. Further, 97% of saplings found within control plots were of the smaller size class, and many displayed signs of heavy browsing. Possible explanations for such strong effects of deer on riparian regeneration in Mendocino County include:

- (1) *Deer densities.* Although quantitative data are not available, deer densities are estimated to be quite high in Mendocino County (J. Booth, CDFG, personal communication). High deer population densities have been implicated in significant changes in vegetation composition in the eastern United States (Alverson et al. 1988; Tilghman 1989) and may amplify the effects that deer have on riparian regeneration.
- (2) *Mediterranean climate—dry season pressure.* In the eastern United States, the greatest impacts due to deer herbivory occur during the winter, when little other browse is available and deer preferentially feed on certain woody species (Alverson et al. 1988; Tilghman 1989). The Mediterranean climate of Mendocino County results in a comparable season of low food availability during the dry months. Generally, little rain falls between May and October, and during the late summer and early fall, riparian corridors are one of the few sources of green vegetation. The annual drought may, thus, result in a seasonal bottleneck on riparian seedling and sapling survival due to herbivory.
- (3) *Threshold effects.* Systems that have been damaged due to a stressor may recover after that stressor has been removed. However, recovery may not be possible if the degree of degradation exceeds a certain threshold (Hobbs & Norton 1996). The riparian corridors of the streams in this study may have crossed such a threshold for recovery, due to the near complete removal of riparian vegetation. In other words, deer herbivory may not be sufficient to significantly influence a riparian corridor that has been lightly disturbed, but the same level of deer herbivory may be able to prevent a degraded riparian corridor from recovering toward its original condition. The streams in this study had almost no riparian vegetation prior to the construction of deer exclosures. The

lack of other riparian vegetation would have increased the browsing pressure on any seedlings and saplings that did establish outside the exclosures.

Conclusions

The results of this study emphasize the importance of monitoring and documenting restoration projects in order to learn from the results. In the upper Russian River watershed, removal of livestock from the riparian corridors of three streams was not sufficient to promote regeneration of woody vegetation. Landowners and agency biologists believed that deer herbivory might have been responsible for the lack of recovery, and they implemented fencing projects to address this possible stressor. Following the elimination of deer herbivory, riparian corridors on Feliz and Robinson Creek responded with vigorous regeneration. The regeneration at Parsons Creek has not been as dramatic but provides further evidence that deer herbivory may act as a stressor to the recovery of degraded riparian systems. Although the six exclosures were not originally established as a scientific experiment, we believe that much can be gained from documenting the results of these ongoing stream restoration projects; lessons learned can be disseminated to other landowners, agencies, and ecologists. The influence of deer herbivory should be considered when planning a riparian restoration project in this region and potentially in other regions with similar patterns of degradation, deer density, and/or seasonal dynamics. Preliminary fencing projects would be recommended to determine if deer herbivory is limiting riparian regeneration at a specific site.

Acknowledgments

The authors would like to thank R. Keiffer, C. Vaughn, and G. Giusti for providing information on the Parsons Creek Project, K. Heise for help with fieldwork, and C. Brooks for assistance with mapping the study sites. Landowners D. Meda and R. Morris provided access to their properties and essential information about the restoration projects on their creeks. We also want to thank Tom Schott, Natural Resources Conservation Service, and Jack Booth, California Department of Fish and Game, for their assistance. W. Silver, D. McCullough, R. Harris, T. Dudley, and S. Mansfield gave much-appreciated feedback on earlier versions of this paper. Edie Allen, Tom Griggs, and an anonymous reviewer also helped improve this manuscript for publication. Funding for this research was provided by an internship from the University of California's Department of Agriculture and Natural Resources through the Renewable Resource Ed-

ucation Act and a National Science Foundation Graduate Fellowship.

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