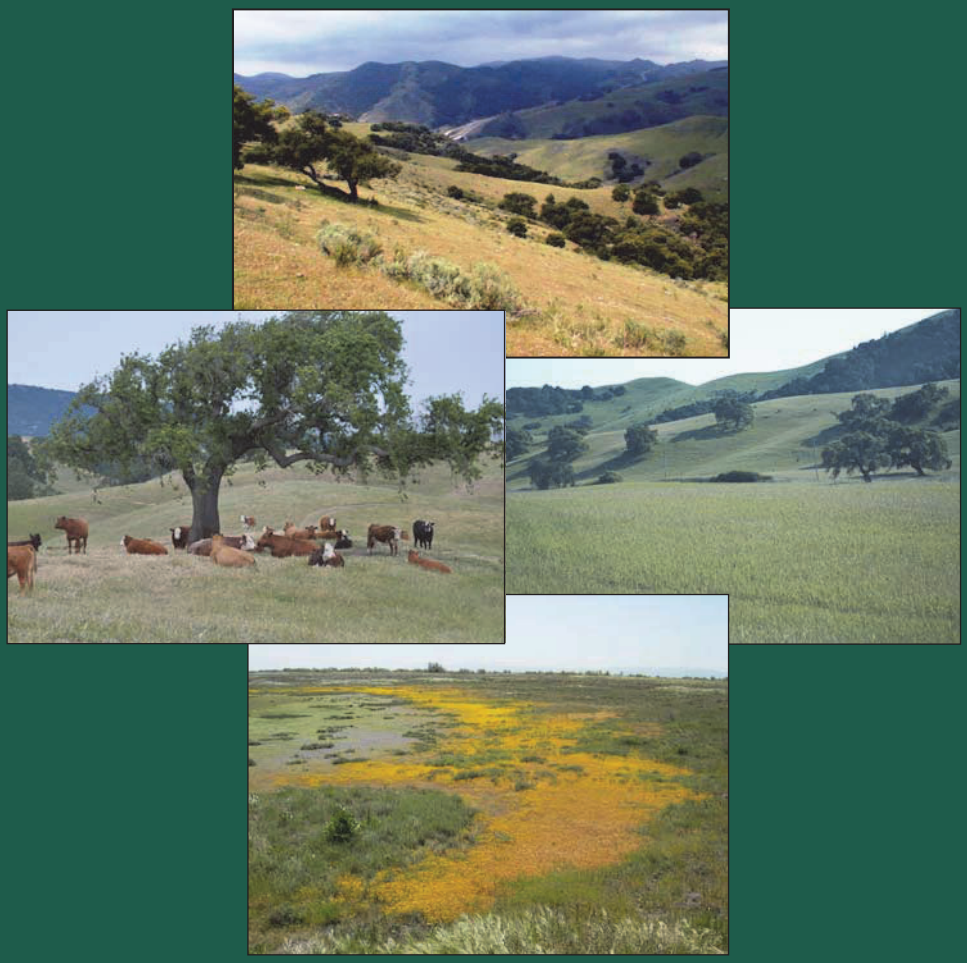
An Economic Analysis of the Benefits of Habitat Conservation on California Rangelands



CONSERVATION ECONOMICS WHITE PAPER

Conservation Economics Program

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There are several individuals from the ranching, university and conservation communities who assisted us in this research project by lending their valuable time, expertise and insights into the types and levels of benefits generated by California ranchlands. During the initial phases of this project we wanted to get a realistic view of what resource conservation practices are most commonly adopted by California ranchers, and their perceptions of benefits from and constraints to increased conservation activity. The ranching community, as represented by Bill Burrows and Larry Galper in Tehama County, and Scott Stone and David Batcheller in Yolo County shared their ranches and homes, conservation experiences, and views of conservation programs and ecosystem service markets that have influenced the policy proposals put forth in this report.

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The material we present in Chapter 2 of this report on the private financial impacts of adopting resource conservation practices could not have been written without the critical input (through numerous meetings and conference calls) from two individuals working with the USDA Natural Resources Conservation Service. Hal Gordon, program economist with the regional NRCS office in Portland, Oregon, provided his Grazing Economic Analysis model that we used to generate ranch-level estimates of the production and economic impacts of implementing resource conservation practices. Jon Gustafson, the California State Rangeland Management Specialist based in Davis, California, gave invaluable advice on what biophysical values/parameters we should use in the Grazing Economic Analysis model and which NRCS-funded conservation practices were most likely to be adopted by California ranchers. In addition, we would like to thank Josh Davy of the Livestock and Natural Resources Program at the University of California Cooperative Extension office in Red Bluff, California for his insights into the costs and benefits associated with resource conservation practices on ranchlands in Tehama, Glenn and Colusa Counties.

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Executive Summary

This report presents the results of a study to investigate the private and *public* economic benefits associated with the conservation of wildlife habitat and other natural resources on rangelands in California's Central Valley. There are over 11 million acres of grasslands within and encircling the Central Valley and the interior Coast Range, much of which are privately owned and managed as rangelands for livestock production. These ranches provide the last, best remaining habitats for many of what previously were wider-ranging species, including freshwater fish, wintering birds and waterfowl, invertebrates and mammals. There are 75 plant and animal species associated with California grasslands that are listed as threatened or endangered under the Endangered Species Act. These same grasslands are located in some of the state's fastest-growing counties and are under severe threat from conversion and development. California lost 105,000 acres of grazing lands to urbanization between 1990 and 2004 and it could lose 750,000 acres more by 2040. Biodiversity conservation in the Central Valley is inextricably linked to the continuation of private ranching landscapes that are sustainably managed through the adoption of resource conservation practices.

The sustainability of the ranching industry in California is linked to improved access to resource conservation incentives and the development of financial incentive mechanisms for ecosystem services in the form of markets and payment programs. We analyze two types of economic benefits associated with the adoption of selected conservation practices - riparian fencing, water development, prescribed grazing and re-vegetation of riparian areas with native plant species, re-establishment or afforestation of oak trees, and restoration of rangelands with native grasses - that are known to have positive impacts on wildlife. The first analysis addresses the *private financial* costs and benefits accrued by ranchers from the adoption of selected resource conservation practices. The second analysis identifies, and to the extent possible, quantifies the *public benefits* of enhancing ecosystem services that result from rangeland conservation practices. These services include carbon sequestration, water quality, biodiversity conservation and pollination, among many others.

Our analysis of the *private* financial costs and benefits from adopting various combinations of conservation practices indicates that private financial and resource conservation objectives can be compatible. Under the assumption of a common 50% USDA cost-share for the installation of various combinations of conservation practices, key financial performance indicators including the internal rate of return (IRR), benefit/cost ratio, and net present value (NPV) are positive for the analyzed practices under given empirically-based assumptions about increases in forage availability and carrying capacity. At the same time, practices associated with prescribed grazing, fencing of riparian areas and water point development can produce substantial ecosystem service benefits related to water quality and wildlife habitat. However, assuming no cost-share assistance, none of the conservation practice scenarios is financially viable for the rancher given the current market conditions for cow-calf operations that dominate rangeland use in California. Without cost share, private installation costs almost double in all cases, the IRRs are zero, the benefit-cost ratios are below 1 (where costs and benefits are equal), and the NPV is negative for all conservation practice scenarios. These results suggest that (1) private investment in rangeland conservation practices with no cost-share assistance is not likely, unless markets are created and are accessible by ranchers for various ecosystem services resulting from conservation practices but for which currently there frequently is no compensation, and unless the prices

on those markets are sufficiently high to make ecosystem service provision financially competitive with development; and (2) the financial attractiveness of adopting conservation practices depends on the economic, biophysical and product marketing conditions faced by *each individual* rancher.

Our review of the ecosystem services and associated benefits provided by California Central Valley rangelands and by rangeland conservation practices yields several general conclusions. First, California rangelands generate a wide range of services that carry considerable total economic value. These services support benefits like livestock production, wildlife- and water-based recreation activities, drinking and irrigation water, species conservation, aesthetic benefits in the form of scenic views, and avoided damages to health, private property and public infrastructure. The second finding is that while some of the value generated by the services provided by rangelands can be captured by landowners – for example, improvements in forage quantity, quality or availability and in carbon sequestration – a substantial portion of the overall benefits accrues off-site. The latter include avoided water treatment and dredging costs, avoided health costs and property damages, passive use values for threatened, endangered or rare species, and aesthetic benefits associated with scenic views. In such cases, ranchers are unable to prevent others from enjoying the benefits that their resource conservation efforts produce and fail to reap the full value of these benefits. Ranchers therefore do not have an incentive to take the full value of these benefits into account when making rangeland management decisions.

For the re-establishment or afforestation of blue oaks (*Quercus douglasii*) on grasslands and the restoration of native perennial grasses and riparian areas, private costs exceed private benefits. Cost share programs in some cases can reverse this result, as is true for some riparian fencing or restoration measures. To the extent that the public benefits from these conservation practices exceed the increases in cost share levels needed to make these practices attractive to landowners, increased public funding for these practices would yield positive net benefits for society as a whole. The alternative approach would be to promote the establishment of viable ecosystem service markets for public goods associated with mitigation of greenhouse gasses through carbon sequestration, biodiversity conservation and water quality. Which of these two approaches for generating increased incentives for the provision of ecosystem services is the preferred one depends on the characteristics of the particular ecosystem services that are being generated by a conservation practice. Those practices that primarily generate public-benefit services like biodiversity and endangered species conservation generally are not suitable to commodification and thus to being traded in markets. Therefore, the primary financial incentive mechanisms for managing private rangelands for the provision of these services are public conservation payments and cost share programs. Those services which are amenable to commodification because they can be quantified and because rationable demand for them exists in principle can be promoted through ecosystem service markets. Carbon and water quality are two prime examples of such services. In many cases, the demand for these services likely will need to be created through regulatory drivers like greenhouse gas emission limits or clean water regulations. Overall, both payment programs and markets for ecosystem services will play important roles in achieving rangeland conservation and the increased provision of ecosystem services from these lands.

Major Conclusions

Ranches in the California Central Valley provide the last, best remaining habitats for many of what previously were wider-ranging species, including freshwater fish, wintering birds and waterfowl, invertebrates and mammals. These lands are also critical for wildlife adaptation to climate change.

Private, ranch-level, financial costs and benefits from adopting various combinations of conservation practices indicate that private financial and public resource conservation objectives can be compatible.

Our results suggest that (1) private investment in rangeland conservation practices with no cost-share assistance is not likely, unless markets are created and are accessible by ranchers for various ecosystem services resulting from conservation practices but for which currently there frequently is no compensation, and unless the prices on those markets are sufficiently high to make ecosystem service provision financially competitive with development; and (2) the financial attractiveness of adopting conservation practices depends on the economic, biophysical and product marketing conditions faced by *each individual* rancher.

With respect to the provision of public-benefit ecosystem services, ranchers are unable to prevent others from enjoying the benefits that their resource conservation efforts produce and fail to reap the full value of these benefits. Ranchers therefore do not have an incentive to take the full value of these benefits into account when making rangeland management decisions. Consequently, without cost share programs or without the creation of service markets, these services will not be provided at the levels that would generate the highest benefits for society as a whole.

Under current economic conditions, the re-establishment or afforestation of blue oaks on grasslands and the restoration of native perennial grasses and riparian areas, private costs exceed private benefits. Cost share programs in some cases can reverse this result, as is true for some riparian fencing or restoration measures. Increased private market prices for carbon sequestration and water quality could induce ranchers to supply more of these public benefits.

Both payment programs and markets for ecosystem services, as well as other types of incentive measures, will play important roles in achieving rangeland conservation, the preservation and recovery of endangered species, and the increased provision of ecosystem services.

1. Introduction

A. Background

This white paper constitutes our final report to the Richard and Rhoda Goldman Fund to present the results of a study of the economic benefits associated with the conservation of natural resources on grasslands and oak woodlands in California's Central Valley and surrounding foothills. The study has been a priority of the California Rangeland Conservation Coalition, a unique collaboration between Defenders of Wildlife, the California Cattlemen's Association and more than 90 other organizations, agencies, and local government entities. We report our findings with respect to the private and public economic benefits associated with the adoption of selected natural resource conservation practices, including benefits associated with wildlife habitat protection and restoration.

California has more than 34 million acres of rangelands within and encircling the Central Valley and the interior Coast Range that are grazed. This area is a unique and valuable natural resource for California as it includes a mix of oak woodlands, open grasslands, vernal pools and wetland habitats. Much of the Central Valley grasslands and foothills are privately owned and managed as rangelands for livestock production (Huntsinger et al., 1997). Many sites on these private rangelands are the last, best remaining habitats for what were previously wider-ranging species.

In general, California grasslands are divided into two plant communities: the Coastal Prairie dominated by perennial grasses, and the Annual Valley Grasslands (Valley Grasslands) dominated by annual grasses (Bartolome et al., 2007). This report focuses on the environmental and economic benefits provided by Valley Grasslands.

California grasslands are located in some of the state's fastest-growing counties and are under severe threat from conversion and development. As a result, California lost 105,000 acres of grazing lands to urbanization between 1990 and 2004, according to the state Department of Conservation. The California Oak Foundation projects it could lose 750,000 acres more by 2040. Managed appropriately, private ranches can benefit California's plants, fish and wildlife. The conservation of biodiversity in the Central Valley and surrounding foothills is inextricably linked to the continuation of private working landscapes and the sustainable stewardship of private grasslands by ranchers.

B. Wildlife Benefits from California Rangelands

Ecosystem services provided by rangelands include supporting services such as wildlife habitat, carbon sequestration, water quality and pollination (Daily et al., 1997). Rangelands also provide provisioning services such as forage for livestock and wildlife (Havstad et al., 2007) and other ecosystem services like open space and cultural amenities (Brunson and Huntsinger, 2008). In this report, we address the importance of California grasslands to selected supporting services (carbon sequestration, water quality) in addition to wildlife habitat because of the co-benefits that can accrue to wildlife when these supporting services are enhanced. Although we provide a more in-depth examination in Chapter 3 of the benefits to wildlife, and particular species, from the adoption of resource conservation

practices, we briefly describe here the importance of current rangeland ecosystems and selected management practices for conserving native wildlife species and their habitats.

Grasslands in the Central Valley of California support a wide variety of wildlife species including freshwater fish, birds, invertebrates and mammals (Hunting, 2003). The Central Valley grasslands not only provide forage for livestock, but also for wildlife populations that rely on these areas for a significant portion of their diet (George et al., 2001; Heady, 1988). There are 75 species associated with California grasslands including 10 vertebrates, 14 invertebrates and 51 plants that are listed as threatened or endangered under the Endangered Species Act (Jantz et al., 2007). In the four-county study area covered in this report, there are 10 listed species in Butte, 5 in Glenn, and 7 in Shasta and Tehama counties (California Department of Fish and Game, 2009). We describe a sampling of some important species that inhabit California rangelands below.

A number of reptile species breed in Valley Grassland habitats including the western fence lizard, common garter snake, and western rattlesnake (Basey and Sinclear, 1980). Reptile species that depend on grassland habitat for food include the San Joaquin whipsnake (California Department of Fish and Game, 2009). Amphibians like the threatened California red-legged frog are also present on annual grasslands and rely on the presence of livestock stock water ponds for habitat (Fellers and Kleeman, 2007).

California Valley Grasslands provide arguably the most important wintering habitat for raptors in the North America (Pandolfino 2006). Twenty-one species of hawks, eagles and owls are regularly recorded on Christmas Bird Counts in the Central Valley, including species of continent-wide conservation concern like Burrowing Owl, Northern Harrier, Golden Eagle, and Ferruginous Hawk (FWS 2002). Grassland birds, more than any other guild, are in decline across the country (Pettyjohn and Sauer 1999, Sauer et al. 2005). Those declines are mirrored in the Central Valley with significant, long-term decreases in both breeding and wintering populations of very common species like Killdeer, Horned Lark, Lark Sparrow, and Western Meadowlark (Sauer et al. 1995, Pandolfino 2006). The Swainson's Hawk, listed as Threatened in California, forages in Central Valley grasslands in spring and summer. California Bird Species of Special Concern (Shuford and Gardali 2008) which use these habitats extensively include Northern Harrier, Mountain Plover, Burrowing Owl, Loggerhead Shrike, Grasshopper Sparrow, and Tricolored Blackbird.

Vernal pools on Central Valley Grasslands provide unique habitats because of special topographic and environmental characteristics that support a multitude of endemic species. These seasonal pools provide habitat for a diverse flora of native plant and several threatened and endangered animal species including invertebrates like the vernal pool fairy shrimp and the tadpole shrimp (Helm, 1998) and vertebrates like the California tiger salamander (United States Fish and Wildlife Service, 2004). Vernal pools in the California Central Valley also support a number of wintering migratory birds (Silveira, 1998, 2000).

In the Central Valley there are 28 native fish species including four different stocks (runs) of Chinook salmon and 40 introduced species (Moyle, 2002). Some important native species found on California rangelands include the California roach, Chinook salmon, rainbow trout/steelhead, Sacramento pikeminnow, Sacramento sucker and speckled dace (Thompson

et al., 2006). There are also several non-native game fish species like the green sunfish, largemouth bass, and smallmouth bass (Thompson et al., 2006).

Mammal species that depend on Valley Grassland habitat include the black-tailed jackrabbit, California ground squirrel, Botta's pocket gopher, western harvest mouse, California vole, badger and coyote (White et al., 1980). Central Valley Grasslands also provide habitat for the endangered San Joaquin kit fox (United States Fish and Wildlife Service, 1983). The Tulare grasshopper mouse is a species of concern that typically inhabits arid grass and shrub land habitats that are mostly in ranching (United States Fish and Wildlife Service, 1998).

Pollination by bees and other animals is an essential ecosystem service that increases the yield, quality and stability of 75% of globally important crops (Klein et al., 2007). It has been estimated that the value of crop pollination in the United States by the honey bee ranges from $5-$14 billion dollars annually (Morse and Calderone, 2000). Wild bee species are also important for the provision of pollination services (Greenleaf and Kremen, 2006; Winfree et al., 2007).

Recent studies stress the importance of rangelands for pollinators. Pollinator activity depends on the availability of nesting sites (Frankie et al., 2002; Hraniz et al., 2009) and the diversity and abundance of floral resources (Murray et al., 2009; Kremen et al., 2002), both of which are provided by California rangeland that is adjacent to cropland. Numerous studies report that the ability of wild and introduced bees to pollinate crops decreases as distance to natural and semi-natural habitat increases (Kremen et al., 2002, 2004; Ricketts et al., 2004; Greenleaf and Kremen, 2006; Morandin and Winston, 2007). In a review of case studies around the world, Ricketts et al. (2008) found a pattern of decreasing pollinator abundance in crop fields with increasing distance from natural or semi-natural habitats. A study in Canada showed that bumble bee and wild bee abundance was positively correlated with the amount of surrounding pastureland (Morandin and Winston, 2007).

In California, Kremen et al. (2002) have found a strong positive relationship between nearby natural habitat (much of which is in rangeland) and pollinator activity of native bees on croplands. A study of native bees in Northern California showed that both the amount and the stability of pollination services increased with increasing area of upland habitat (riparian forest, chaparral and oak woodland) which, in Northern California, is mostly provided by private ranches (Kremen et al., 2004). A species of native bee (*Bombus vosnessenkithat*) that pollinates tomato plants relies on adjacent natural habitat for nesting sites (Greenleaf and Kremen, 2006). The same authors concluded that farmers benefit from having adjacent rangelands which are essential for the health of native pollinator populations because which in turn helps reduce their costs of managing commercial honeybees (Kremen et al., 2004). A meta-analysis of the effects of anthropogenic disturbance on wild bees showed that habitat loss was the most important factor decreasing population abundance and richness, contributing to their recent decline in the United States (Winfree et al., 2009). With increased losses of California rangelands, it is likely that the state's pollinator bee populations will also decrease, which could have very adverse consequences for California's crop production.

Although we provide a more detailed analysis in Chapter 3 of the benefits to ecosystem services, including wildlife habitat, resulting from the adoption of conservation practices on California rangelands, we briefly illustrate here why this topic is important. Some of these

practices include stock pond maintenance, managed grazing, and fencing and restoration of riparian areas.

Ranch stock ponds (not analyzed in this report) play an increasingly important role in providing habitat for amphibians. In the Bay Area, livestock stock water ponds on ranchlands provide up to 50% of the remaining habitat for the threatened California tiger salamander (United States Fish and Wildlife Service, 2006). The California red-legged frog also thrives in stock ponds (Fellers and Kleeman, 2007). These two species also seem to benefit from livestock grazing around or near these ponds (DiDonato, 2007).

Managed grazing by ranchers is a cost-effective and natural tool for managing vegetation and enhancing wildlife habitat. It is also a tool that is being used by several land management agencies in California (Huntsinger et al., 2007). For example, it has been demonstrated that managed grazing can improve habitat for threatened and endangered species such as the Bay checkerspott butterfly, considered an umbrella species for grassland ecosystems (Murphy and Weiss, 1988). Through the removal of competition from non-native annual grasses, cattle grazing increases the abundance of native forb dwarf plantain (*Plantago erecta*) that the checkerspot feeds on (Weiss, 1999).

Recent research in California also suggests cattle grazing can be an essential management tool in maintaining native vernal pool ecosystems. Cattle contribute to maintaining biodiversity by selectively grazing on the invasive exotic grasses, reducing evapo-transpiration, and thereby extending the inundation period of the pools which allows the invertebrates to complete their life cycle (Marty, 2005; Pyke and Marty, 2005).

A few wildlife species on California rangelands benefit from having a shorter vegetation structure, which grazing creates, thereby increasing chances of wildlife finding available prey or allowing them to avoid predation (Barry et al., 2006). The San Joaquin Kit fox also benefits from grazing because the species favors flat and open space to avoid predators (Warrick and Cypher, 1998; United States Fish and Wildlife Service, 1998). Habitat enhancement by sheep grazing has been shown to increase populations of the Stephens kangaroo rat, a federally and state listed species (California Department of Fish and Game, 2009). Other species that benefit from livestock grazing are the California ground squirrel (Fehmi et al., 2005) and the burrowing owl (Barry et al., 2006).

C. Purpose

Defenders of Wildlife, along with the California Cattlemen's Association, has worked with California ranchers, environmentalists and agencies to create an agreement, titled The California Rangeland Resolution (Appendix 1). The Resolution is based on a multi-party effort to conserve and enhance private working landscapes and wildlife habitat within the Central Valley, surrounding foothills and interior Coast Range. Together, signatories to the Resolution formed the California Rangeland Conservation Coalition (Coalition). The long-term goal of the Coalition is to conserve private grasslands and promote habitat enhancement projects on these lands for the benefit of listed and unlisted species. The Central Valley and surrounding foothills encompass more than 28 million acres of rangelands. Those areas that were identified as the most critical for conservation are shown in Figure 1.1.

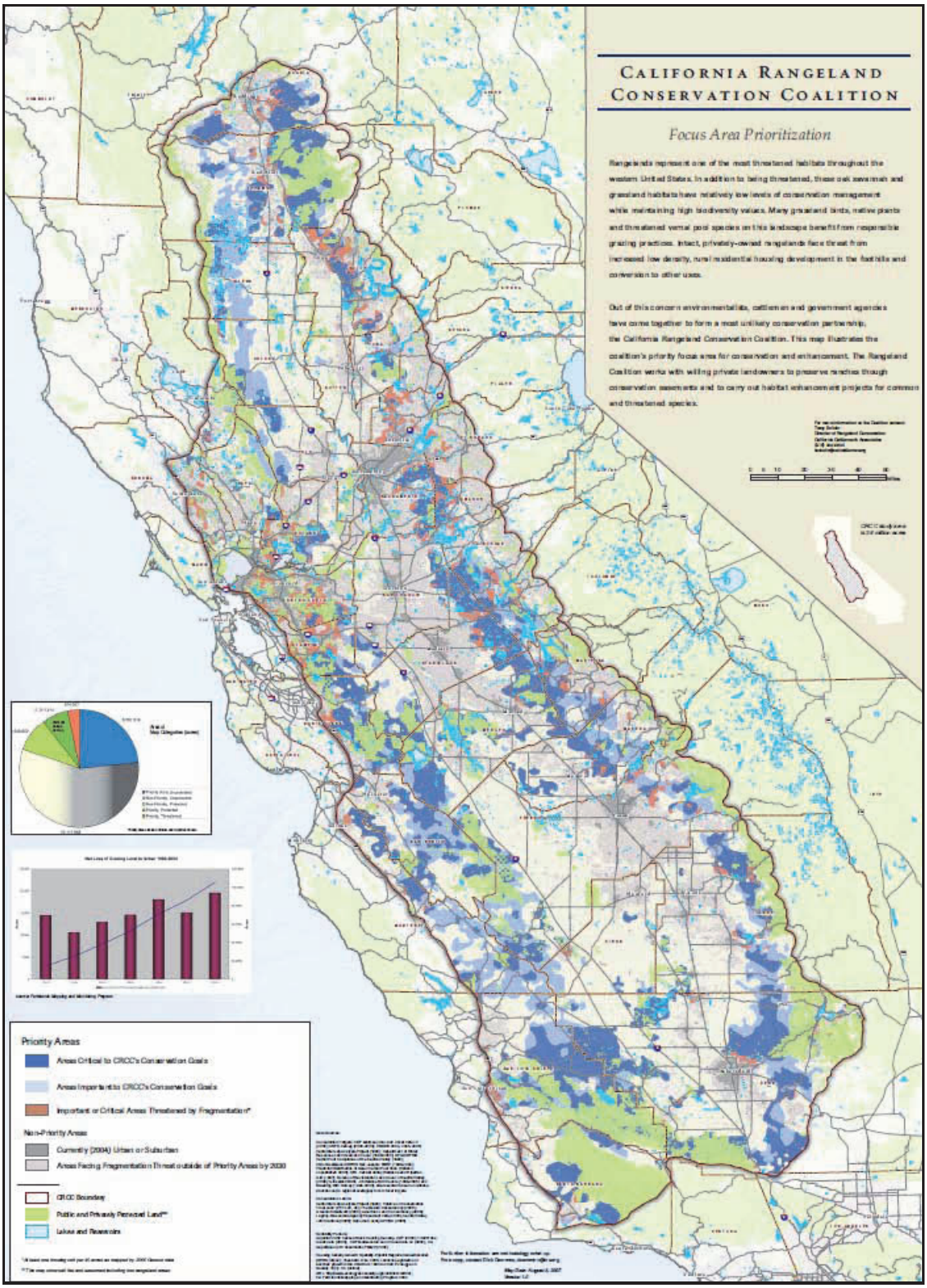

Figure 1.1: California Rangeland Conservation Coalition priority areas

The sustainability of the ranching industry in California is linked to the development of markets for ecosystem services such as wildlife habitat. More specifically, it is linked to the ability of ranchers to benefit from those markets to create incentives that will increase the likelihood of achieving conservation goals. In order to make the case for the development of markets for ecosystem services on and from California rangelands, this report describes and analyzes the prospective ecological and economic benefits that could result from the conservation and restoration of these important ecosystems. We investigate two types of economic benefits associated with the adoption of selected conservation activities: those that would or could be captured by private ranchers, and those public benefits that are derived from improved environmental conditions and/or ecosystem services.

This report consists of three types of integrated economic analyses. The first addresses the relative *private* financial costs and benefits of alternative resource conservation management regimes to improve wildlife habitat on working rangelands in California. We analyze these costs and benefits by investigating the economic and financial impacts resulting from the adoption of specific resource conservation practices on a "typical" ranch in our study area. We employ a Grazing Economic Analysis model (Gordon, 2008) that generates estimates of financial returns and benefit-cost ratios to the ranching operation resulting from the impact of conservation practices on forage production and livestock (cattle) productivity. These conservation practices include riparian fencing, water development, prescribed grazing and re-vegetation of riparian areas with native plant species.

The second type of economic analysis identifies and, to the extent possible, quantifies the value of ecosystem services that result from rangeland conservation practices that benefit wildlife. We address the question of what market and non-market ecosystem services, including wildlife habitat, will be affected and improved as a result of adopting conservation practices. We provide some preliminary estimates as to how much these services may be worth. The ecosystem services investigated include increased forage production, carbon sequestration, improved water quality and wildlife habitat. Practices investigated include re-establishment or afforestation of oak trees, riparian protection and revegetation and restoration of native grasses.

Lastly, we address the general economic conditions under which wildlife habitat conservation on working grasslands is profitable for private ranchers and identify potential policy mechanisms and/or incentives (private markets; public payments) that would allow ranchers to more fully capture the economic benefits from providing increased ecosystem services to the public. We also address the question of what policy/incentive instruments can be employed to compensate ranchers/landowners for ecosystem services provided, including greater access to public conservation programs.

Although the data employed in this report comes from ranching sites throughout the state of California, the economic analysis presented here is meant to reflect conditions in four counties in north-central California: Butte, Glenn, Shasta and Tehama (Figure 1.2). Relevant data and literature for other areas of the western U.S. was consulted and used wherever appropriate.

Figure 1.2: Four-county project focus area

D. Organization of the Report

The next chapter provides a ranch-level financial analysis of the costs and benefits from the adoption selected resource conservation practices that directly or indirectly result in improved wildlife habitat or species protection. Chapter 3 provides a description and, to the extent possible, quantifies selected market and non-market ecosystem service benefits generated by the adoption of rangeland conservation practices. Chapter 4 provides recommendations with respect to future public policy options to encourage habitat and species conservation on California ranchlands, including markets for ecosystem services and various other landowner incentive mechanisms.

2. Financial Analysis of Adopting Selected Conservation Practices on California Rangelands

This chapter addresses the question "What are the relative *private* financial costs and benefits of alternative treatments and management regimes to improve wildlife habitat and other natural resources on California rangelands?" Profitability at the ranch level, or even a neutral impact on ranch income, is a major factor in ranchers' decisions to adopt resource conservation practices, including those aimed at protecting wildlife species and their habitats.

We analyze the question of on-ranch profitability of conservation practice adoption through the use of a Grazing Economic Analysis model (Gordon, 2009) developed by the USDA Natural Resources Conservation Service (NRCS). Given the installation of a particular resource conservation practice, or suite of practices, the model estimates the financial/economic impact on a "typical" ranch operation as a result of implementation costs (labor and materials) and changes in forage availability, harvest efficiency and livestock carrying capacity. We define a "typical" ranch as a 900-acre cow-calf operation that is located in the four-country study area. The four conservation practices we consider, as defined through the California State NRCS office, are fencing of riparian areas, active restoration of native plants in riparian zones[1], prescribed grazing management, and water point development. We describe and analyze the following three resource conservation practice adoption scenarios through the model: Scenario I: adoption of a conservation practice package that includes fencing off riparian zones combined with water point development; Scenario II: the same as Scenario I, but with active re-planting of fenced riparian zones with native plant species; Scenario III: the combined adoption of water point development and prescribed grazing management. The conservation practices for each Scenario have been put in place by some ranchers, but adoption over a wider area could be promoted more extensively from a natural resource management perspective. Each of the practices investigated addresses several resource concerns simultaneously, and although they are not *directly* aimed at particular species, they do have beneficial impacts on both wildlife and their habitats. These *non-market* benefits are described in more detail in Section D of Chapter 3.

A. Grazing Economic Analysis Model

The Grazing Economic Analysis model is applied to a "typical" ranch from our four-county project study area (Butte, Glenn, Shasta and Tehama counties) in order to determine the private economic net benefits from the adoption of resource conservation practices. Key model parameter values were chosen to be representative of conditions in our study area, based on consultations with local NRCS experts. Parameter values may differ from other rangeland areas in California due to differences in environmental conditions. For example, the costs and the benefits generated by the model are dependent on several site and location characteristics (soil type, present condition of rangeland forage, trend in the condition of rangeland resources) and evaluation criteria (cost of practices, lifetime of treatment, interest rate, total acres to which conservation practices are applied, etc.). The model variables are described below and the results from each of the three Scenario outputs are provided in Appendix 2, Tables A2.1 to A2.3. The next sections of this chapter provide a general

[1] Riparian zones are mainly restored by planting native trees and shrubs and sometimes with an understory of grasses and forbs.

description of the model and the results of the Grazing Economic Analysis for the three conservation practice Scenarios.

B. General Model Description and Assumptions

The Grazing Economic Analysis model (Gordon, 2009) is a set of interactive spreadsheets that estimates the profitability and selected other financial performance indicators of a management practice or bundle of practices for a particular ranch, based on treatment costs and improved livestock carrying-capacity, forage availability and harvest efficiency expected to result from the adoption of the treatment. The other financial performance indicators include the break-even period for an investment, the internal rate of return, the benefit-cost ratio and the net present value per acre generated by the adoption of a suite of conservation practices. The value of these economic indicators determines whether a rancher should invest in these conservation activities, all of which benefit particular species or wildlife habitats. The key concept here is that for there to be widespread adoption of resource conservation practices by California ranchers that benefit wildlife or improve other natural resources such as water and air quality, these practices must either have a positive or neutral impact on ranch income.

The various parameters used in the model runs that represent site and location characteristics, evaluation criteria, and forage availability and utilization efficiency are shown in Appendix 2. The time-frame chosen for the analysis is 20 years. Site and location characteristics include descriptive identifiers such as county/state, soil type, present condition of rangeland forage, and the trend in forage production. None of these variables directly impact the model's output of key indicators, although they inform the choice of appropriate, empirically-based values of key model parameters such as forage availability and carrying capacity. Based on consultations with local NRCS experts, we assume that the present forage production condition of rangelands in the four-county study area is "good" and that the current trend in this condition is "stable".

Evaluation criteria for the model include information about the conservation practice or suite of practices to be adopted (called the grazing land treatment), the cost per acre of the practice(s), the duration of the life of the practices, the applicable interest rate, the value of an Animal Unit Month (AUM)[2], and the total acres treated (see Appendix 2). For each of the conservation practice Scenarios, we assume an average ranch size of 900 acres for the four-county area. Additionally, we hold constant the duration of the life of the selected practices (20 years)[3] and the interest rate (7%) to allow comparisons of the different suites of practices examined. Costs per acre are derived from a state-wide cost list that is maintained by NRCS for various conservation practices, and the basis on which NRCS will cost share. The only evaluation criteria that differ across Scenarios are the type of grazing land treatment (i.e., conservation practices adopted) and the initial treatment and annual maintenance costs per-acre.

[2] An Animal Unit Month (AUM) is defined as the amount of forage needed by an "animal unit" (AU) for grazing for one month. The quantity of forage required is based on a cow's metabolic weight, and the animal unit is defined as one mature 1,000 pound cow and her suckling calf. For this analysis we assume a standard forage amount of 790 lbs/AUM.

[3] Although we use a 20 year time frame, the duration of specific practices can be adjusted for in the model.

The variables under "Forage Utilization" include forage availability, pounds per AUM, forage harvest efficiencies both with and without the conservation practice, current forage availability, maximum carrying capacity and the number of months the ranch operation can be grazed each year (See Appendix 2). Average forage availability for the four-county study area is assumed to be 2000 pounds/acre.[4] Forage per AUM is a standard weight at 790 pounds. Harvest efficiency is the percent of annual above-ground biomass consumed by an animal unit. For each of the conservation practice Scenarios, we assume (based on consultation with local NRCS experts) a slight increase in harvest efficiency (from 20% to 25%) due to the adoption of the suite of resource conservation practices. Current forage availability and maximum carrying capacity are variables that are generated through formulas that draw on the data in the cells that indicate pounds of forage required per AUM and harvest efficiency percentages. The number of months that California rangelands can be grazed each year varies with climate, herd size and location. In our study area, rangelands are grazed, subject to climatic conditions, between four and eight months per year. For this analysis, we assume a grazing period of six months for each of the Scenarios.

A crucial variable that determines the size of the economic impacts of adopting resource conservation practices is the setting of the percentage change in carrying capacity of the ranch before and after the practices have been implemented (see Appendix 2). These percentages are based on the field knowledge of local range management specialists in California and vary by resource conservation practice.

C. Model Details and Results

Scenario I: Riparian Fencing and Water Development

The first conservation practice Scenario is based on the combined adoption of the practices of riparian fencing and water development. Appendix Table A2.1 shows the results of the Grazing Economic Analysis for these practices. We assume that a rancher pays 50% of the total cost of the practices adopted, which is what would typically be paid under the Environmental Quality Incentives Program that is managed by the USDA Natural Resources Conservation Service.[5] This conservation practice Scenario has been adopted by ranchers in California for (1) the purpose of protecting riparian areas and improving water quality, with potential benefits to aquatic habitat; and (2) to increase livestock access to under-utilized forage through a more even utilization of the property by livestock by developing additional watering points. The latter outcome has the impact of creating healthier range conditions, which in turn benefits several wildlife species, and can control the spread of exotic invasive weeds. There are also private benefits to ranchers from the adoption of fencing riparian areas and water development. These benefits take the form of improved forage utilization, better animal health and higher beef production due to better water quality, more reliable water sources, and erosion control in riparian zones (Belding et al., 2000).

[4] Personal communication. Dr. Melvin George, Rangeland Ecologist, University of California at Davis. July 2009.

[5] Personal communication. Jon Gustafson, State Rangeland Management Specialist, California NRCS office. July, 2009.

In Scenario I we assume that forage harvest efficiency is 20% before adoption of the conservation practices, and 25% in the first year following adoption through the end of the 20-year period. The improvement is due to increased access to forage in the areas near new watering points. Based on input from local range managers, we assume that there would be no change in the carrying capacity of rangelands over the 20-year period of analysis *without* adoption (before treatment) of the conservation practices.[6] However, *with* adoption (i.e., under the treatment), we assume that the carrying capacity of the ranch increases by 10% in the first year, and then increases by 20% for years 3 thru 20.

Scenario I model outcomes for primary production and economic indicators are shown in Table 2.1. The value of the indicators make it clear that, given the 50% cost share level, and assumed increases in harvest efficiency and carrying capacity, it is profitable for the rancher to implement fencing and water development practices.

Table 2.1 Economic and production results - conservation practice scenario I: Riparian fencing and water point development

Years to Break-Even on Investment:	0
Increase in Carrying Capacity (AUMs/Ac):	0.24
Increase in Stocking Rate, (#Head/Total Ac):	31
Total Installed Treatment Cost ($/Total Ac):	$21,236
Amortized Installed Treatment Cost ($/Ac/yr):	$2.23
Internal Rate of Return:	99%
Breakeven $/AUM	$9.14
Benefit/Cost Ratio:	1.36
Net Present Value ($/Ac) under 50% cost share:	$9.91

Stocking rate immediately increases from 76 to 107 head of cattle upon installation of the conservation practices and remains at the higher level for the life of the project. The total cost of the two practices is about $21,000/acre and includes the cost of foregone income from taking acreage out of forage production.

The internal rate of return (IRR), defined as the interest rate at which the present value of the income stream generated by the practice becomes zero, is 99%. The decision to invest in the conservation practices is made by comparing the 99% IRR to the cost of capital faced by the ranch. Needless to say, such a high rate of return far exceeds the cost of borrowing capital, indicating that the adoption of the practices represents a good investment for the rancher. Further evidence of the profitability of adopting the suite of practices, as illustrated by the benefit/cost ratio of 1.36, indicates that overall benefits exceed costs by about 36%. The IRR and the benefit/cost ratio illustrate that adoption of the Scenario I conservation practices, which have beneficial impacts on wildlife, can be profitable at the ranch level.

With the adoption of the riparian fencing and water development practices, the breakeven price at which the practice bundle becomes profitable is estimated at about $9.15/AUM, which indicates what the value per AUM needs to be for a rancher to recoup the costs of the conservation investment. The net present value (NPV) of the investment is estimated to be about $10.00/acre. NPV is the sum that results when the discounted value of expected costs

[6] Ibid.

is deducted from the discounted value of expected returns generated from the conservation investment. If the NPV is positive, then the investment is potentially worth making, depending on how this particular suite of conservation practices ranks with other Scenarios and investment opportunities the rancher faces.

In addition to the private financial benefits/returns from adopting riparian fencing and water development practices, there is a range of *non-market*, *public* benefits that could be jointly generated by these practices. These types of benefits could include improved water quality for both human and aquatic species' use, control of invasive exotic species, and improved habitat for both riparian and upland wildlife species.

Scenario II: Active Riparian Restoration (Native Species), Riparian Fencing, and Water Development

The conservation activities in Scenario II differ from those in Scenario I only by the addition of "active" restoration of riparian areas with native plant species, after fencing has been installed. With the exception of the added costs of native re-vegetation of riparian areas, ranch size, evaluation criteria and forage utilization variables assumed in Scenario I are unchanged (See Appendix A2.2). Restoring riparian areas with native vegetation is assumed to *not* have an appreciable impact on either harvest efficiency or an increase in carrying capacity. The average total cost of a riparian planting for our sample ranch of 900 acres is estimated at about $6.40/acre, based on NRCS cost data. With an assumed 50% cost share, the added increment for riparian re-vegetation is about $3.25/acre compared to Scenario I, which results in an overall treatment cost for the whole package of practices of $26.84/acre.

Table 2.2 provides the production and economic results of the combined practices of restoration of native plant species in riparian areas, riparian fencing and water development. Compared to Scenario I, the additional element of restoration of native plant species in riparian areas does not result in a decrease in carrying capacity. Thus, the ranch still enjoys augmented production as a result of adding a native species restoration practice that may have more of a public rather than private benefit. If planting native species resulted in increases in marketable sequestered carbon over some baseline, then both private (depending on market conditions) and public benefits would increase compared to Scenario II.

Table 2.2 Economic production results - conservation practice scenario II: Fencing and restoration of riparian zones and water development

Years to Break-Even on Investment	0
Increase in Carrying Capacity (AUMs/Ac):	0.24
Increase in Stocking Rate, (#Head/Total Ac):	31
Total Installed Treatment Cost ($/Total Ac):	$24,153
Amortized Installed Treatment Cost ($/Ac/yr):	$2.53
Internal Rate of Return:	53%
Breakeven $/AUM	$10.40
Benefit/Cost Ratio:	1.22
Net Present Value ($/Ac) under 50% cost share:	$6.67

Both the total and amortized costs of adopting the new suite of conservation practices rise, but only marginally on a per-acre basis. For example, total treatment costs with riparian re-vegetation are estimated to be about $24,000/acre, as compared to nearly $21,200/acre without riparian re-vegetation, or an increase of about 13%.

The additional costs associated with adding riparian re-vegetation to the suite of Scenario I conservation practices does not significantly impact the financial benefits that a rancher would receive for the additional investment. The IRR decreases from 99% in Scenario I to 53% in Scenario II, but is still significantly high to warrant the investment. The breakeven point as measured by the price of an AUM ($/AUM) increases from about $9.00/acre to $10.40/acre. The benefit/cost ratio of 1.22 shows that benefits exceed costs by over 20%. The NPV per acre decreases with riparian re-vegetation from about $9.90/acre to $6.70/acre, or by nearly 32%, but it is still positive and an indicator that restoring riparian areas with native vegetation as part of fencing riparian areas and water development would still be profitable. Given that riparian restoration with natives is expected to not have a significant impact on forage, and therefore livestock production, adoption of the restoration practice alone would probably not be expected to be attractive to ranchers, at least from a financial return standpoint.

The incremental addition of riparian restoration with native plant species to the Scenario I suite of conservation practices does not provide increased financial benefits to the rancher because the restored areas are not expected to be grazed. However, there may be public benefits associated with improved carbon sequestration, water quality, and wildlife habitat, and perhaps other ecosystem services as well.

Scenario III: Water Development and Prescribed Grazing Management

Our third conservation practice scenario is based upon the adoption of prescribed grazing management in combination with the water development practice that was included in Scenarios I and II. The details of prescribed grazing schemes can differ from ranch to ranch, depending on ecological and forage conditions, but the practice generally involves rotating cattle to prevent overgrazing, to improve and protect the long-term viability of forage species, and to improve cover and forage for wildlife. In some cases, the use of prescribed grazing may also be complemented by fencing and placing mineral supplements. This practice mostly involves increased labor costs to manage and move cattle and to monitor rangeland conditions to determine when cattle need to be herded to another location (see Table A2.3. in Appendix 2 for model details). Water development is required in order to move livestock to sections of the ranch that may not have watering wells or stream access.

As a new prescribed grazing management program is implemented, it is expected that there will be a gradual increase in forage availability over the 20-year period of analysis. This translates into an increase in livestock carrying capacity for the ranch. Based on consultation with local NRCS range experts, we assume a 5% increase in carrying capacity in the second year of the project, 15% in the third year, and then 20% in years four through the end of the project. With the exception of the carrying capacity and labor cost increases, and the active restoration costs, all other model parameters remain the same as in Scenarios I and II.

Table 2.3 shows the values of the production and economic indicators for Scenario III. The total installed treatment cost for the rancher, at a 50% cost share rate for water development and at the 75% EQIP rate for materials needed for monitoring grazing, is about $15,600/acre. The IRR is estimated at 216%, well above those for Scenarios I and II, and is an indicator that the investment in prescribed grazing and water development would be financially beneficial. The breakeven AUM price is estimated at about $7.00/AUM. At the given cost share levels, benefit/cost ratio and NPV are estimated at 1.61 and $14.00/acre, respectively. The values for the benefit/cost ratio and the NPV strongly indicate that it would be profitable for the rancher to invest in prescribed grazing management in conjunction with water point development.

Table 2.3 Economic and production results - conservation practice scenario III: Water development and prescribed grazing management

Years to Break-Even on Investment	0
Increase in Carrying Capacity (AUMs/Ac):	0.24
Increase in Stocking Rate, (#Head/Total Ac):	30
Total Installed Treatment Cost ($/Total Ac):	$15,600
Amortized Installed Treatment Cost ($/Ac/yr):	$1.64
Internal Rate of Return:	216%
Breakeven $/AUM	$6.80
Benefit/Cost Ratio:	1.61
Net Present Value ($/Ac) under 50% cost share:	$13.95

As in the case of Scenarios I and II, there are likely ecosystem service benefits that cannot be readily quantified in monetary terms. These benefits include improvements in water quality and wildlife habitat and are described in Section D of Chapter 3.

D. Comparison of Conservation Practice Scenarios and Preliminary Conclusions

What are the relative *private* financial costs and benefits of alternative treatments and management regimes to improve wildlife habitat and other natural resources on California rangelands? We have approached this question by using a Grazing Economic Analysis model (Gordon, 2009) as a tool to estimate the financial impact on a ranching operation of three suites of conservation practices.

The analysis of the production and economic impacts on ranching operations in the four-county project area from the adoption of the selected conservation practices indicates that private financial and resource conservation objectives can be compatible. Although the conservation practices investigated do not target wildlife conservation *per se*, their effects on increasing water and land resource quality do benefit rangeland aquatic and terrestrial habitats.

Table 2.4 summarizes some of the major production and economic indicators for each of the conservation practice Scenarios for the 50% cost-share level, and compares each of these parameters for the three Scenarios to the case of no cost-share. The comparison is meant to illustrate how the financial viability of the various conservation practice packages would change drastically if ranchers had to bear the entire costs of implementation.

Table 2.4: Financial indicators for conservation practice scenarios at 50% and zero cost share

Indicator	*Scenario I*		*Scenario II*		*Scenario III*	
	50% CS	0%CS	50% CS	0% CS	50% CS	0% CS
Total Installed Treatment Cost ($/ Total Ac):	21,236	42,471	24,153	45,389	15,600	31,200
Internal Rate of Return:	99%	0	53%	0	216%	0
Benefit/Cost Ratio:	1.36	.73	1.22	.69	1.61	.92
Net Present Value ($/Ac):	9.91	-13.70	6.67	-16.93	13.95	-3.39

Under the 50% cost-share assumption, each of the conservation practice scenarios is a viable financial investment at the private ranch level. The combined practice of prescribed grazing management and water development (Scenario III) has the least cost and the highest economic returns in terms of the estimated IRR, benefit/cost ratio and NPV. Scenario I is the second-most attractive investment alternative, followed by Scenario II which includes restoring riparian vegetation with native plant species. All three conservation practice combinations will have beneficial impacts on wildlife habitat, which are described in Section D of Chapter 3. The selection of which conservation practices ranchers decide to ultimately adopt will depend on their particular financial situation, the ecological condition of their ranches, and expected environmental and product price conditions over the medium to long term. For example, the cost-effectiveness of a particular practice will depend on how the rancher will arrange for installation, that is, whether the rancher installs the practice with his own labor, or contracts out. Likewise, the economic indicators presented here do not account for the transactions costs faced by ranchers in enrolling in and implementing conservation programs, or the costs of obtaining permits, which could be substantial.

Table 2.4 also provides estimates for selected economic indicators under the assumption of no cost-share from the NRCS, thereby illustrating the case of what returns ranchers could gain if they were to invest in the given conservation practices entirely on their own. Under the no cost-share assumption, none of the conservation practice scenarios is financially viable for the rancher. Costs of installation almost double in all cases, the IRRs are zero, the benefit-cost ratios are below 1.0 (where costs and benefits are equal), and the NPV for all conservation practice scenarios is negative. There are three major implications of these findings. First, under the production and financial assumptions we employed, private investment in rangeland conservation practices with no cost-share assistance is not likely, especially under the current market conditions for maintaining cow-calf operations in California. This conclusion changes to some extent if and where markets exist and are accessible by ranchers for various ecosystem services (carbon sequestration, water quality, wildlife habitat) that are co-benefits of installing conservation practices but for which currently there frequently is no compensation.

A second implication is that there is likely some median point (not investigated here) at which the costs of adopting resource conservation practices are just offset by the private economic benefits achieved. However, the 50% cost share would appear to leave enough

room to accommodate any margin for error associated with assumptions behind the Grazing Economic Analysis on either the cost or the benefit side.

The third implication from the analysis presented here is that the financial attractiveness of each of the suites of conservation practices will depend on the economic, bio-physical, and product marketing conditions faced by *each individual* rancher. Overall installation costs for practices may be higher or lower than assumed by NRCS, depending on the physical characteristics of any particular ranching operation. Likewise, any increases in carrying capacity, and hence the potential for augmented income, will depend on the response in forage quantity and quality to improved management at any specific site.

Although we can generally conclude that there are private, on-ranch benefits from the adoption of the resource conservation practices illustrated here, we have also shown the important role that public investment has on financial viability of the rancher's decision to invest in these practices under current economic circumstances. Chapter 3 will provide an investigation of the types of *public* ecosystem service benefits that can be generated by specific conservation practices and the extent to which these benefits can be captured by private ranchers through market mechanisms.

3. Ecosystem Services Provided by California Grasslands

A. Introduction

Like all ecosystems, grasslands provide a wide array of goods and services that contribute to human wellbeing (White et al., 2000; Maczko and Hidinger, 2008). For reasons of convenience, we will refer to these outputs collectively as "ecosystem services."

The ecological literature provides several rather broad definitions of ecosystem services as natural processes and products that support human existence and enhance human well-being (Daily, 1997; Daily et al., 1997). The Millennium Ecosystem Assessment (2005) followed this broad definition, distinguishing between supportive services (those that lead to the maintenance of the conditions for life, such as nutrient cycling), provisioning services (those that provide direct inputs to human economy, such as food and water), regulating services (such as flood and disease control), and cultural services (such as provision of opportunities for recreation and spiritual or historical purposes).

While such broad definitions have many useful purposes, they lump together ecosystem functions or processes (such as nutrient cycling or habitat provision), ecosystem products (such as food, fiber, or water), and benefits (the economic value of a service, such as flood control or aesthetic beauty). This creates two problems when attempting to place an economic value on ecosystem services. First, since ecosystem products are the output of ecosystem processes or functions, counting both the products and the biophysical processes producing them will tend to raises issues of double-counting (Boyd and Banzhaf, 2007). Second, ecosystem functions and their outputs describe only the biophysical supply side of nature's outputs. The economic value of those outputs, however, is a function of both their supply and demand (Tallis and Polasky, 2009; McDonald, 2009). In other words, human beneficiaries are required for an output of nature to become an ecosystem service– there must be people who actually benefit from that output.

To avoid the problems associated with too broad and economically imprecise definitions of ecosystem services, Brown et al. (2007) argue that ecosystem services should be defined as "flows from an ecosystem that are of relatively immediate benefit to humans and occur naturally" (Brown et al., 2007:334). Boyd and Banzhaf (2007:619) suggest narrowing this definition even further to include only end-products – "components of nature, directly enjoyed, consumed, or used to yield human well-being."

A corollary of the foregoing is that ecosystem services are benefit-specific, that is, they are contingent on, and specific to, particular human activities or wants (Boyd and Banzhaf, 2007). For example, aquifer or surface water quality is an ecosystem service for the provision of drinking water, because humans directly value the quality of the water they drink. On the other hand, water quality is not an ecosystem service for sport fishing, because anglers do not generally value the quality of the water body *per se*. Rather, they value the fish themselves. Therefore, the ecosystem service in this case is the target species (e.g., trout, bass). In economic terms, water quality is but one of many inputs in the production function of the target fish species, and its contribution to the value of the output (the fish) is part of the total value humans assign to that output. The value of the quality of the surface water body is

embedded in the value that anglers assign to the target species. Another rangeland example that demonstrates how this benefit-specificity defines what qualifies as an ecosystem service is erosion control. People generally do not value avoidance of topsoil loss *per se*. Rather, ranchers value the productivity of their land for livestock production. Therefore, forage production is the ecosystem service for this benefit because it is the forage that is the immediate input to livestock production. Likewise, people downstream generally do not value erosion control *per se*. Rather, they value the avoidance of damages in the form of reduced reservoir dredging costs or flood damages. The ecosystem service producing these benefits is natural land cover, which controls erosion.

Developing a definition of ecosystem services that complies with economic and accounting principles is not merely an academic exercise. Rather, a precise definition of ecosystem services that identifies the latter as discrete, countable and identifiable end-products is a necessary condition for their quantification, which in turn is a prerequisite for the establishment of ecosystem service markets. It is not surprising that all such markets or market-like arrangements that have developed are for services that conform to the definition advocated here – directly valued end-products such as water, sequestered carbon stocks, trees or species (Kroeger and Casey, 2007). Table 3.1 provides a non-exhaustive list of the services provided by California rangelands and the economic benefits they support.

Table 3.1: Selected benefits and associated ecosystem services provided by California rangelands; the services discussed in this report are underlined

Benefit	*Ecosystem Service*
Livestock harvest	Forage production, water availability
Crop harvests (nearby properties)	Pollinator populations
Recreation – hunting, fishing	Relevant species populations; natural land cover
Recreation – wildlife viewing	Relevant species populations
Wildlife passive use benefits	Relevant species populations (threatened/ endangered/rare species and habitats)
Drinking water provision – Avoided treatment cost	Aquifer and surface water quality (run-off nutrient absorption)
Drinking water provision – Avoided pumping/transport cost	Aquifer and surface water availability (aquifer infiltration)
Aesthetic benefits (open space property value premiums; outdoor recreation)	Natural land cover in view shed
Damage avoidance – Health benefits	Drinking water quality (nutrient and bacterial control)
Damage avoidance – Property	Natural land cover (trees, grasses), soils, wetlands (climate change, rain storm events)
Damage avoidance – Harvests (forage)	Native plants resistant to invasion by unpalatable weeds (e.g., cheat grass)
Damage avoidance – Reservoirs, stream channel dredging	Natural land cover

Sources: Boyd and Banzhaf (2007); Maczko and Hidinger (2008); Valerie Eviner (personal communication).

In this report, we focus only on a few of these ecosystem services, underlined in Table 3.1, that are produced by a sample of rangeland conservation measures: Forage production for livestock; threatened, endangered or rare species and their habitat; natural land cover (trees and grasses) and soils that sequester atmospheric carbon dioxide; drinking and aquifer and surface water quality associated with prevention of nutrient and bacterial runoff; and erosion control. These services are, to varying degrees, provided by all rangelands. However, implementation of particular rangeland conservation measures can increase the provision of some of these services from a particular area. Importantly, all of the rangeland conservation practices we examine in our analysis also have been documented to benefit wildlife. Thus, adoption of these practices in many cases may make economic sense for ranchers while also generating economic value for third parties, including neighboring agricultural lands – which can benefit for example from services provided by pollinators that rely on rangeland habitat – and society at large – which benefits from scenic views, cleaner water and the preservation of species, to name but a few of the benefits provided by intact rangelands.

In the following sections, we discuss some of the benefits provided by oak reestablishment and afforestation, restoration of native perennial grasses, grazing management and riparian fencing and restoration. Whenever available data permit, we generate quantitative estimates of the benefits and associated values brought about by these rangeland conservation practices. In many cases, available data and the limited scope of this analysis do not allow quantification of all of the benefits produced by the set of conservation measures analyzed in this report. In those cases, we at least identify the benefits and briefly point out how their value could be estimated. Where possible, we also generate cost estimates for these conservation practices. The chapter closes with a comparison of the economic benefits and costs associated with the practices that highlights the existing discrepancies between private and public perspectives on the economic feasibility of implementing the selected practices.

B. Oak Reestablishment and Afforestation

Oak reestablishment on and afforestation of range grasslands has the potential to generate a wide range of economic and ecological benefits, including increased carbon sequestration, forage production, provision of wildlife habitat and improved downstream water quality. In this chapter, we discuss these benefits in some detail and develop quantitative estimates for some of them.

Forage impacts of oak reestablishment on grasslands

From the late 1950's through the early 1970's, several studies reported that palatability and production of forage in the understory of blue oak was low when compared to forage in open grassland areas (Johnson et al., 1959; Burgess, 1987; Kay, Burgess and Leonard, 1980; Murphy and Berry, 1973; Murphy and Crampton, 1964). These studies formed the basis for statewide "rangeland improvement" activities that resulted in the removal of blue oak from grazing areas and the loss of 1 million acres (0.4 million ha) of woodlands (Bolsinger, 1988; IHRMP, 1998). According to some assessments, these losses may lead to a long-term decrease in soil quality and forage productivity (Dahlgren et al., 1997; Camping et al., 2002).

In contrast, more recent studies (Table 3.2) found that blue oak cover did not decrease forage production (Bartolome et al., 1994), at least for canopy cover levels of up to 40-60 percent (Battles et al., 2008; Connor and Willoughby, 1997), or actually did increase forage production (Frost and McDougald, 1989; Frost et al., 1991; Ratliff et al., 1991). Furthermore, oak cover was found to significantly increase forage quality, with one study reporting that forage under oaks was higher in crude protein concentration and lower in acid detergent fiber and lignins, and exceeded livestock crude protein requirements for six months compared to one month on open grasslands (Frost et al., 1990).

Several hypotheses have been offered to resolve the discrepancy of the effects of blue oak on forage production between early and later studies. The reasons advanced for variation in forage impacts include tree density, climate and soil factors (Duncan and Clawson, 1980; Kay, 1987; Menke, 1987). In particular, mean annual precipitation has been identified as a factor influencing the relationship between forage yield and oak canopy, with oak canopy reported to reduce forage yields only where mean annual precipitation exceeds 20 inches (McClaran and Bartolome, 1989) but increasing understory production on dry sites (Bartolome, 1987; Ratliff et al., 1991). The latter would be consistent with the observation that canopy shading becomes extremely valuable in drought years by reducing moisture loss from evapotranspiration (Frost et al., 1989). Another study (Callaway et al., 1991) indicates that blue oaks with shallow, fine roots inhibit understory production, which may be partially attributable to allelopathic blue oak root exudates as well as competition for water and nutrients. Variations in root morphology may therefore explain differences in understory production of blue oak.

Most studies examining the forage impact of oaks do not address the question of causality. Is higher productivity under blue oak canopies due to the inherent properties of the sites occupied by oaks? Some authors, finding that the removal of oaks did not significantly decrease forage production, argue that oak may simply be found on sites that are

Table 3.2: Recent literature findings on the impact of oaks on forage production

Author	*Study Location*	*Rangeland Type*	*Mean Rainfall (mm/yr)*	*Soil Type*	*Study Details*	*Forage Quantity and/or Quality Impacts*
Bartolome et al. (1994)	W. of Paso Robles 35°40' N, 120°37' W	Blue oaks w/ understory of annual herbs	360	Dibble clay loam	Removed all blue oaks from sample plots to test effect on understory production	Forage production: No significant effect. 36.6 g/m^2 (1989), 60.2 g/m^2 (1990) and 58.1 g/m^2 (1991). Herbaceous cover: significantly increased (24.3% uncut, 32.6% in openings); Species composition: remained relatively stable, except for increase in Redstem filaree (*Erodium cicutarium*)
Connor and Willoughby (1997)	Sierra Nevada foothills (UC SFREC) Yuba County	Small open grasslands, savannas, and dense oak woodlands	724	rocky loams	Measured forage clippings at 4 sites w/ varying % of blue oak canopy	No consistent relationship btw. oak canopy and forage production,; strong relationship btw. yearly rainfall and yearly forage production.
Frost et al. (1989)	San Joaquin Exp. Range, 25 mi NE of Fresno	Savanna w/ >400 annual herb species, 21% tree cover	587; but study in drought year)	coarse-loamy mixed thermic	Measured herbaceous production under canopy and in open grasslands	Blue oak consistently yielded higher forage production (kg/ha) every month (Nov-May). Herbaceous production was higher during higher rainfall years.
Ratliff et al. (1991)	San Joaquin Experimental Range, 25 mi NE of Fresno	6% swales, 11% open-rolling uplands, 83% rocky-brushy uplands	483	coarse loams	Compared livestock and herbage response to repeated seasonal, rotated seasonal, and continuous grazing on unfertilized pastures and pastures fertilized with elemental sulfur	In open-rolling uplands and rocky-brushy uplands, average peak herb standing crops were highest under blue oaks: in swales, forage was greatest on open land
Battles et al. (2008)	Sierra Nevada foothills (UC SFREC) Yuba County, 39° 15' N, 121° 17' W	Savanna with watershed-level mean of 56% canopy cover. Dominated by annual grasses.	775	fine, mixed, thermic	Determined pattern of net primary productivity (NPP); examined relationship between blue oak biomass and productivity	Canopy cover levels of around 40-60% do not suppress forage growth. Herb productivity increased with increasing canopy cover and began to decline only once this saturation point was reached. Total NPP increased linearly with increasing canopy cover until it saturated at approximately 50% cover.

-over-

Table 3.2 continued

Jackson et al. (1990)	Sacramento Valley foothills, Approx. 38°N	Savanna grasslands with 70% oak cover	n/a	various types	Analyzed nutrient content of soils and plants under canopy and on open grasslands	Soils beneath the oak canopies have approximately one-third more carbon and N, a higher cation exchange capacity and lower pH. Phosphate levels are slightly higher under the oak canopies.
Malmstrom et al. (2009)	Sierra Nevada foothills (UC SFREC) Yuba County, 39° 15'N, 121° 17'W	Grasslands and oak savannas	568	Palexeralfs, entic chro-moxererts, and typic haploxeralfs	Used remote sensing to evaluate effectiveness of native bunchgrass restoration project. Performed prescribed burns for weed control	Biomass declined in first year after native grass reseeding, followed by 1-2 years of biomass increase, and more varied responses afterwards. 3-5 years after the treatments, biomass had decreased in some fields but increased in fields where mixed approaches (flash grazing, fencing or reburning) were utilized. Native bunchgrasses did particularly well in areas of deep soil and short-duration, high-intensity grazing.
Frost et al. (1991)	San Joaquin Experimental Range, 25 mi NE of Fresno	blue oak-interior live oak/grass cover type	483	coarse loamy	Compared forage quantity and quality under blue oak canopy and in open grasslands	Forage production was significantly higher under blue oaks. Average 1987-1990 peak standing crop was 1,089 lbs/acre more under blue oaks. Forage quality under blue oaks was 54% higher in crude protein concentration and lower in acid detergent fiber (ADF) and lignins (LS). Open grasslands exceeded crude protein requirements of nursing cow only in one month (March), while forage under blue oak canopies exceeded them from mid-Dec to May.

inherently more fertile (Bartolome et al., 1994). Conversely, a number of studies have shown how oaks impact the microclimate of a site (Frost et al., 1989, 1991; Ratliff et al., 1991), which in turn would explain increases in forage production. Other studies do support a causality that runs from oaks to site fertility and not the other way around, showing that oak trees create islands of enhanced fertility through the incorporation of organic matter and enhanced nutrient cycling (Dahlgren et al., 1997; Camping et al., 2002; Jackson et al., 1990).

The empirical evidence on the relationship between the level of canopy cover and understory forage production is contradictory as well. Battles et al. (2008) found that herb productivity was increasing for relatively open sites (canopy closure less than 40 percent) and then monotonically declined as canopy closure increased further. In contrast, Connor and Willoughby (1997) found no consistent effect of canopy level on forage yield.

Oak cover has been shown to impact the timing of understory forage growth, producing much faster forage growth early in the season (March-May) compared to open areas (Frost et al., 1991; Duncan and Clawson, 1980). Forage under blue oak remained green after surrounding forage had dried, and Duncan and Clawson (1980) reported that cattle preferred forage beneath blue oak to that of open grassland, even in summer after forage in both areas has dried.

Table 3.3 summarizes the impacts of blue oak cover on rangeland forage quantity for the studies listed in Table 3.2. The general conclusion of the findings reported in studies that detected significant forage impacts is that blue oaks increase forage quantity on uplands, with impacts ranging from -0.3 to 1.3 AUMs (790 lbs) per acre, if we exclude Battles et al.'s (2008) findings for areas with 80 percent oak cover.[7] Thus, based on these studies, and depending on the particulars of a site, oak cover on upland sites is expected on average to produce a forage increase of 0.5 AUM per acre, or the equivalent of half the forage consumed by one mature 1,000 pound cow and her suckling calf.

[7] Canopy cover levels this high are not typical of most rangelands, and generally would not be the goal for oak reestablishment on grasslands.

Table 3.3: Differences in forage yield under blue oaks reported in more recent studies

Study	*Terrain type and study years*	*Blue oak cover*	*Annual forage production*				
			Blue oak	*Open grassland*	*Increase under blue oak*		
		%	*kg/ha*	*kg/ha*	*kg/ha*	*%*	*lb/ac*
Bartolome et al. (1994)	*Oak removal did not produce significant difference in forage production*						
Battles et al. (2008)	Hilly, rolling terrain	20%	2,450	2,250 *	200	9%	178
		43%	2,750		500	22%	445
		60%	2,250		0	0%	0
		80%	1,500		-750	-33%	-668
Connor and Willoughby (1997)	*No consistent effect of canopy level on forage yield across study years*						
	Uplands			1,631			
	1990-95 avg.	25%	1,478 **		-153	-9%	-136
		50%	1,408 **		-223	-14%	-199
		75%	1,424 **		-207	-13%	-184
Frost and McDougald (1989)	Open-rolling						
	1986-87	Under canopy	2,789	1,672	1,117	67%	994
	1987-88		2,667	1,622	1,045	64%	930
Frost et al. (1991)	Open-rolling						
	1986-1990	Under canopy	2,392	1,303	1,089	84%	970
Ratliff et al. (1991)	1961-1968						
	Swales (unfertilized)		3,664	4,295	-631	-15%	-562
	Open-rolling uplands	Under canopy	3,802	3,118	684	22%	609
	Rocky-brushy uplands		2,828	2,070	758	37%	675
	Overall weighted average:		3,086	2,795	291	10%	259

Notes: * 4% blue oak cover. ** Blue and Interior Live oak.

Carbon sequestration through oak reestablishment or afforestation

Due to their large spatial extent, rangelands may represent an important component in the global carbon cycle (Svejcar et al., 2008). Although grazing has not been shown to have a consistent effect on soil carbon, at least in the semiarid and Mediterranean climates of the Western United States (Jackson et al., 2007; Silver, 2009), conversion of rangelands releases large quantities of carbon (Lal, 2002; Potthoff et al., 2005; Kern, 1994). The inverse is true for conversion of cultivated lands into grasslands (Conant et al., 2001). Thus, avoided conversion of rangelands results in large avoided carbon emissions (Conant et al., 2001; Laca, 2009).

Carbon fluxes in California's Mediterranean-climate rangelands are influenced by a variety of factors including management practices (e.g., grazing pressure, application of organic amendments), vegetation type (savanna vs. grassland), species (native vs. exotic annuals), precipitation and soil type. The defining difference between grasslands and oak savannas is that the former have very little or no tree cover, while oak savannas are generally and somewhat arbitrarily defined as having between around five and 50 percent canopy coverage (Allen-Diaz et al., 1999; Henderson, 1995), with oaks being the dominant tree species. Generally, grasslands in the region are carbon neutral while savannas are moderate carbon sinks (Baldocchi, 2009). Total net carbon sequestration of California's Mediterranean grasslands and savannas (Figure 3.1) at the biome scale is estimated at 8.6 Tg/yr (avg. of 150 $gC/m^2/yr$) (Figure 3.1; Baldocchi, 2009). This amount is equivalent to 1.8 percent of California's gross greenhouse gas emissions in 2006 (California Air Resources Board, 2009), or 0.12 percent of total U.S. gross greenhouse gas emissions in 2007 (U.S. Environmental Protection Agency, 2009). Reforestation of cleared areas in current oak woodlands over a 75-year time horizon could sequester an additional 1.37 million tons of carbon per year, equivalent to another one percent of California's current annual greenhouse gas emissions (Gaman, 2008).[8] Thus, management of California's oak savannas and woodlands represents an important part in the State's climate change mitigation efforts.

Laca (2009) identifies eight practices (including rangeland conservation) that can reduce carbon emissions from the different rangeland types found in our study area (Table 3.4). These include the restoration of perennials and riparian corridors and improved grazing management, all of which are examined in this report, as well as restoration of woody species, the subject of this section. Like all of the rangeland practices included in this report, oak reestablishment and afforestation have the potential to increase revenues for ranchers. Planting of oaks can benefit ranch operations in two ways: First, by improving forage quantity and quality, which increases net revenue by increasing livestock output or reducing input costs (feed, pasture); and second, by increased net sequestration of atmospheric carbon dioxide by rangeland soils and vegetation.

Around 80 percent of oak woodlands in California are privately owned, and the primary use of these lands is livestock production (Bolsinger, 1988). In the Sacramento Region that includes three of our study area counties - Butte, Glenn and Tehama - the percentage of private ownership of oak woodlands is over 80 percent; in Shasta County, it is 73 percent (Gaman and Firman, 2006). An important question in evaluating the possibility of re-

[8] One ton of carbon is equivalent to 3.667 tons of carbon dioxide.

establishment of oaks on rangelands therefore is: Can livestock and oaks be "raised" together?

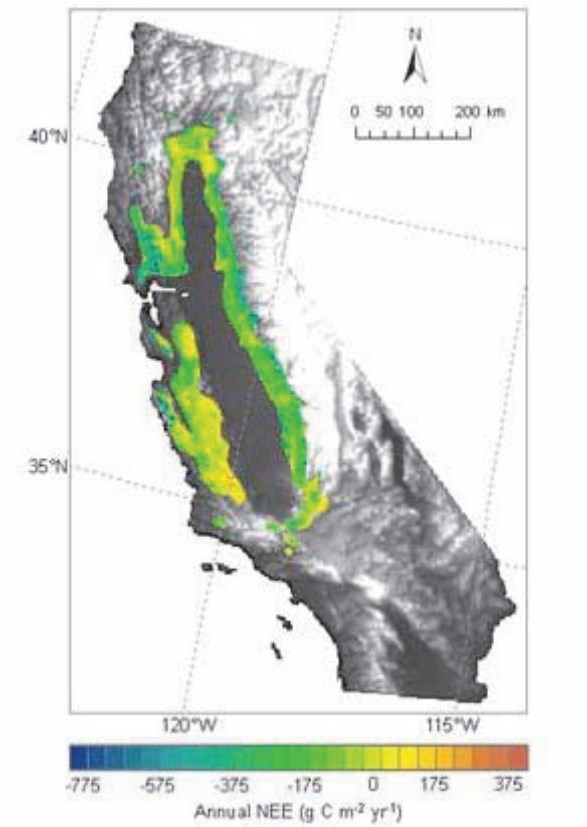

Figure 3.1: Estimated net carbon fluxes of California's Mediterranean rangelands
Source: Xiao and Baldocchi, cited in Baldocchi (2009)

Table 3.4: Relative magnitude of potential carbon benefits associated with particular management practices on California rangelands

Project type	*Rangeland Types*		
	Chaparral	*Oak-Woodland*	*Annual Grassland*
Reduction of wildfires	++	+	+
Restoration of woody species		+++	+
Restoration of riparian corridors	+	++	+++
Restoration of perennial grassland	+	++	+++
Control of invasive weeds	+++	++	+++
Management of expanding shrubs and trees	+++		
Conservation as rangeland (REDD)		++	+++
Improved grazing management	++	++	++
Addition of carbonate/black carbon	?	?	?
Area (million acres)	5.8	7.4	7.1

Source: Laca (2009).

Drawing on the results of several studies that assessed the impacts of livestock on oaks of various sizes, McCreary and Tecklin (2005) found that native California oaks can be

established in pastures grazed by cattle, as long as individual seedlings are protected from browsing and rubbing until they are approximately two meters tall. One study showed that in riparian plantings, protection of individual trees with shelters was necessary for successful oak restoration, with total survival rates of sheltered trees of 58 percent vs. 5 percent for fenced plots and less than one percent for unprotected trees (McCreary, 1999). Another study showed that exclusion of cattle from young (less than seven years old) plots with tree shelter-protected oak seedlings did not reduce tree mortality, although the damage from cattle browsing led to reduced height and basal diameter growth of the seedlings (Tecklin et al., 2002). One study that analyzed the impact of opening up plots with 13-15 year-old oaks to cattle grazing found that there appeared to be a threshold height of around two meters above which oaks generally are large enough to withstand cattle damage. Smaller trees were heavily impacted by cattle browsing, suffering substantial losses in height compared to the ungrazed control plot (McCreary and Tecklin, 2005). However, in moderately-grazed pastures, such damage from livestock clipping appears to have little long-term impacts on seedling survival or growth (McCreary and Tecklin, 2005). This is confirmed by the results reported by Koenig and Knops (2007), who found that regeneration of blue oaks, although very slow, can occur in open oak savannas in California despite significant grazing pressure. Another study (Hall et al., 1992) found that cattle damage to unprotected oak seedlings was significantly less in winter, when the oaks did not have foliage and were apparently less appetizing to cattle, compared to spring, when clover patches near seedlings seemed to attract cattle and to lead to incidental browsing of oaks. The largest damage occurred in summer, when the young oaks often were the only green vegetation in the grazed pastures and were more palatable than dry annual grasses.

A study analyzing the impacts of herbaceous interference and small mammal and insect depredation on oak seedling establishment and survival at seven sites in California found that herb exclusion was the most important factor for seedling establishment and survival (Adams et al., 1991). The study found that establishment and survival of (blue and valley) oak seedlings in California Oak-grass savannas can be significantly increased through both exclusion of herbs and screens against insect depredation (Adams et al., 1991).

The evidence of the existing literature thus shows that reestablishment or afforestation of oaks on rangelands is indeed compatible with livestock production. It also shows that the growth of planted trees, and the concomitant ecosystem service and ranch financial benefits, can be improved through implementation of comparatively simple practices.

Reestablishment and afforestation of oaks will need to take into account expected changes in oak habitat that may result from climate change. For example, a recent study suggests that portions of blue oak habitat in our study area may shift over the next 100 years, with habitat expansions in some areas and contractions in others (Kueppers et al., 2005).

The potential carbon impact of oak restoration and afforestation

To assess potential carbon sequestration from afforestation and reforestation of native oaks on rangelands in the study area, we construct low and high estimates of the changes in net carbon sequestration from oak planting. These estimates are developed by comparing the annual carbon balance of grasslands with that of oak savannas. The difference between the

carbon balances of the two systems is the quantity of carbon uptake that could be achieved by planting oaks on current range grasslands.

There are very few published studies that estimate net carbon uptake, or net ecosystem exchange (NEE) of grasslands and oak savannas in California (Table 3.5). NEE represents the total net flow of carbon between the atmosphere and the biosphere, including all changes in above- and belowground carbon. If there is carbon accumulation in the biosphere, the particular ecosystem is acting as a net carbon sink. If there is a net flow from biosphere to atmosphere, the ecosystem represents a net carbon source. By meteorological convention, net carbon flows into the biosphere carry a negative sign, indicating a loss in atmospheric carbon; net flows from the biosphere carry a positive sign, indicating an increase in atmospheric carbon. With the exception of Valentini et al. (1995), grassland studies in California are for sites with exotic annual grasses. The study results indicate that the carbon balance of annual grasses ranges from moderate carbon source (126 $gC/m^2/yr$, or 0.51 tC/ac/yr) to small sink (-51.1 $gC/m^2/yr$, or -0.21 tC/ac/yr).[9] By contrast, the one study that examined a native perennial grassland found that the site was a moderate carbon sink (-133 $gC/m^2/yr$, or -0.54 tC/ac/yr). These values fall within the range of grassland NEE values reported in the literature (Novick et al., 2004), but are an order of magnitude lower than those reported for most forests (Curtis et al., 2002; Scott et al., 2006; Schmid et al., 2000; Turner et al., 2007; Xiao et al., 2008), indicating the generally low productivity of California's Mediterranean climate grasslands (Battles et al., 2008).

There is only one study (Ma et al., 2007) that examines net carbon fluxes of a Mediterranean-climate oak savanna in California, with the results indicating that the savanna is a small to moderate carbon sink (-98 $gC/m^2/yr$ or -0.40 tC/ac/yr). Another study in the same climate zone (Battles et al., 2008) develops estimates of the net primary productivity (NPP) of three adjacent watersheds covered by oak savannas. To convert NPP estimates into net ecosystem exchange (NEE) estimates, the former need to be reduced by the carbon released through heterotrophic respiration (R_h). Battles et al. do not provide information on R_h in their systems. However, Ma et al. (2007) develop a system of equations describing their savanna system, located in the same area of the Sierra Nevada foothills that allows estimation of R_h. Based on these equations and their reported measurements, R_h of Ma et al.'s savanna is estimated as an average of 346 $gC/m^2/yr$ during 2000-2006. Subtracting this amount from Battles et al.'s average NPP for their three savanna systems yields an average estimated NEE of -176 $gC/m^2/yr$ (Table 3.5). This value is almost twice that reported by Ma et al., a discrepancy that could be attributable to the higher oak cover (56 percent vs. Ma et al.'s 40 percent) and higher average annual precipitation (77.5 cm vs. Ma et al.'s 56.2 cm) of Battles et al.'s site. These results indicate that grasslands are carbon neutral while savannas are moderate carbon sinks (Baldocchi, 2009).

In developing our estimates of the potential carbon sequestration that may result from oak afforestation of grasslands, we use the following scenarios:

[9] We use Chou et al.'s (2008) value from their high root contribution scenario (RC= 70%), 126 $gC/m^2/yr$, as that RC value is based on a study of an annual grassland in southern California; their low RC estimate (RC= 35%) is based on studies of perennial grasslands.

Table 3.5: Net carbon sequestration by grasslands and oak savannas in central western California

Rangeland type	*NEE* $gC/m^2/yr$	*Study period*	*Location*	*Vegetation*
Grassland				
Chou et al. (2008) * - Field - Adjusted **	80-387 *** 126-433 ***	2003-06 water years	Sierra Foothill Research and Extension Center (39.15°N, 121.17°W)	Annual grassland, historically grazed but cattle removed for experiment
Ma et al. (2007)	38 ±52	2000-06 avg.	Vaira Ranch (38.41° N, 120.95° W) (Sierra Nevada foothills)	Open C_3 annual grassland
Valentini et al. (1995)	-133	1990-91	Jasper Ridge (37.27°N, 122.13°W) (Eastern foothills of Santa Cruz Mountains)	Low-productivity, serpentine grassland with mostly **native** C_3 annual and perennial fortes and perennial bunchgrasses
Xu and Baldocchi (2004)	-51.5	avg., 2000-01 and 2001-02 growing seasons	35km southeast of Sacramento (38° 24.4 N, 120°57 W) (Sierra Nevada Foothills)	Grazed grassland opening, cool-season C_3 annuals
Savanna				
Ma et al. (2007)	-98±51	2001-06 avg.	Tonzi Ranch (38.43° N, 120.96° W) (Sierra Nevada foothills)	Oak/C_3 annual grass savanna; Blue oak, 144 stems/acre, 9 m ± 4.33 m avg. height; ~40% oak cover
Battles et al. (2008)	-176 #	2001-02	Sierra Foothill Research and Extension center (39.15°N, 121.17°W)	Blue oak savanna (some Interior live oak and foothill pine) with introduced annual grasses and forbs; watershed-level mean canopy cover: 56 percent

Notes: NEE – Net ecosystem Exchange. Following meteorological convention, negative values indicate a flow from atmosphere to biosphere; positive, a net C flow from biosphere to atmosphere. * Data are for control plots only, not for irrigated plots. ** Upward adjustment of belowground NPP by 50% based on Higgins et al. (2002). *** Estimated values assume root contributions of 70% (low values) and 35% (high values), respectively. # Avg. of NPP of three watersheds, converted to NEE by subtracting estimated heterotrophic respiration, derived using equations and data from Ma et al. (2007); see text for explanation.

Scenario 1: Low Carbon Gain: Change from a small carbon sink, -51.5 $gC/m^2/yr$ (Xu and Baldocchi, 2004) to a slightly larger carbon sink, -98 $gC/m^2/yr$ (Ma et al., 2007)

Scenario 2: High Carbon Gain: Change from a moderate carbon source, 126 $gC/m^2/yr$ (Chou et al., 2008) to a small to moderate carbon sink, -98.0 $gC/m^2/yr$ (Ma et al., 2007)

Our Low and High Carbon Gain scenarios thus assume that planting oaks on grasslands results in a net carbon uptake of 47 and 224 $gC/m^2/yr$, respectively, or 0.19-0.91 tC/ac/yr.

The Low Carbon Gain scenario results in a very conservative estimate as it relies on the only study that reports a net carbon uptake by an exotic annual grassland (Xu and Baldocchi, 2004). In both scenarios, we rely on the lower (Ma et al., 2007) of the two estimates of the net carbon balance of an oak savanna. Our adjusted NEE estimates for the Battles et al. (2008) study are higher but may be less reliable due to the fact that our adjustment of their NPP estimate is based on heterotrophic respiration values from a site with lower oak cover and precipitation (Ma et al., 2007).

The above scenarios are based on carbon flux values from mature oak savannas, not for young savannas characterized by recently planted trees. The actual net carbon balance (NEE) of young savannas depends on the rate of carbon accumulation (i.e., biomass growth) of the planted oaks and on the associated impacts of oaks on the understory grassland carbon balance.[10] Our carbon gain scenarios thus should be corrected for the trend in NEE of oak savannas with tree age. Unfortunately, the literature does not provide these estimates. As a second-best method for incorporating tree age into carbon gain estimates, we assume that the change in NEE of planted oak savannas is approximately proportional to tree growth. We incorporate this assumption by scaling our savanna NEE estimates to the basal area increments (BAI) of oak trees, which measure changes in tree stem diameter and whose trajectory over time serves as a general indicator of tree growth (Kertis et al., 1993).

Oak growth depends on a variety of factors, including species, precipitation, browsing pressure (from livestock and wildlife), length of the growing season and exposure to direct sunlight (vs. growing in the understory). Blue oak in particular are very slow-growing. One study (Koening and Knops, 2007) of blue oak seedlings in Monterrey County, California found that even seedlings protected from grazing by large herbivores and, to some extent, rodents as well, needed two decades to reach a mean height of 66 cm. Unprotected seedlings grew much slower still, reaching a mean height of only 77 cm after over four decades. The average growth rate of the latter oaks increased markedly near the end of the study period, presumably because more of the oaks had achieved a height or width that provided some protection against grazing.

Growth rates in much of our study area likely are higher than those reported by Koening and Knops because mean precipitation in the four counties is substantially higher. The overall range of mean annual precipitation across our study area spans 38.4-50.8 cm to 203.5-254 cm. However, annual means in most of the oak woodlands in the area (Gaman

[10] Understory grasslands in Mediterranean climate oak savannas in California are higher net emitters of carbon than open grasslands. However, this effect is overcompensated through the carbon sink created by the oaks themselves (Ma et al., 2007).

and Firman, 2006) ranged from 63.5-76.2 cm to 152.4-177.8 cm during 1961-1990 (Daly and Taylor, 2000), compared to a mean annual precipitation (1939-2006) of 53.3 cm for Koenig and Knops' site (Koenig and Knops, 2007). The average precipitation in much of the oak rangelands in our study area thus is substantially higher than at Koenig and Knops' site. Since oak growth is strongly correlated with precipitation (Kertis et al., 1993), blue oak growth rates in our study area are expected to be higher than those observed by Koenig and Knops. This assumption is supported by the few studies measuring oak growth closer to our study area. For example, in a study of blue oaks planted at the University of California's Sierra Foothills Research and Extension Center (SFREC), located in northern Yuba County, just south of Butte County, McCreary and Tecklin (2005) report that seven year-old oak seedlings protected from grazing through tree shelters were exceeding 1.3 m in height. Another study at SFREC (Bartolome and McClaran, 1985) recorded vertical growth rates from 0-60 cm and 0-135 cm (browse line) of 34 cm/yr and 16.5 cm/yr, respectively for sprouts and 18 cm/yr and 11.3 cm/yr, respectively for seedlings. The long-term average mean annual precipitation at SFREC was 72.4 cm (Connor and Willoughby, 1996), 19 cm (36 percent) higher than at Koenig and Knops' site (2007). Since the average annual mean precipitation in most of the oak woodlands in our study area is higher than at SFREC, it is likely that oak seedlings in much of our study area would grow faster. Nevertheless, the absolute increments in total aboveground and root biomass of new oak trees will be small for several decades, and as a result, "planted oaks don't begin to register appreciable CO_2 storage for at least 20 years, longer for very slow-growing blue oak" (California Oak Foundation, 2008).

Kertis et al. (1993) analyzed long-term growth trends of oaks at five sites in California, including sites in Soeth (Glenn County) and Butte (Butte County). Their reported BAI for the Soeth and Butte sites are replicated in Figure 3.2. Trees at the Butte site show rapid growth until approximately year twenty, from which time on growth is slower but continues for the next sixty years (there were no trees older than 80 years at the site).

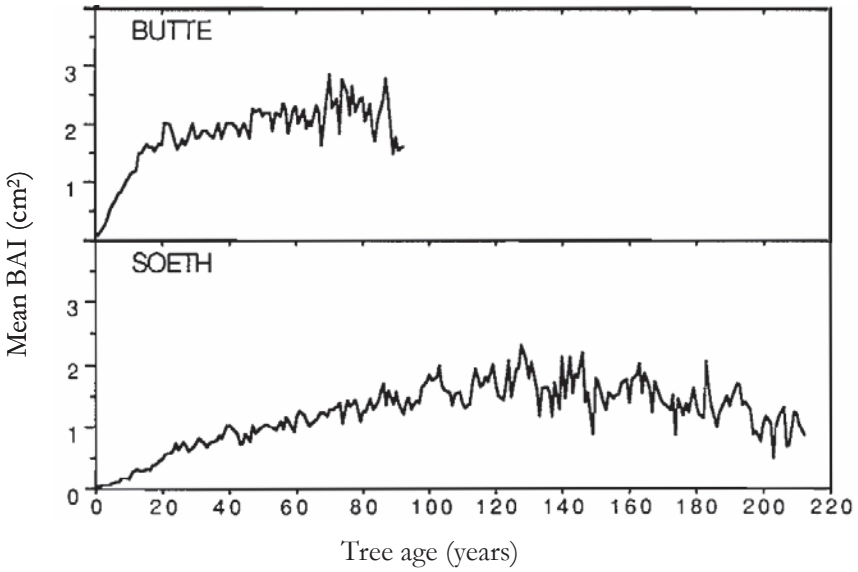

Figure 3.2: Mean annual BAI curves for Kertis et al.'s (1993) Butte and Soeth study sides, from Kertis et al.'s Fig. 2

The overall trend is approximated by a logarithmic curve. Trees at the Soeth site grow much slower as indicated by the smaller slope of the BAI curve, possibly as a result of the steep slopes (70%) (Kertis et al., 1993). Although a change in the growth trend is less obvious than at the Butte site, the data show a slightly faster growth until about year 25, with a somewhat smaller but overall relatively constant BAIs during the following 100 years. Key characteristics of the Butte and Soeth sites are reported in Table 3.6.

Table 3.6: Key characteristics of Kertis et al.'s (1993) Butte and Soeth and Ma et al.'s (2007) Tonzi Ranch blue oak savanna sites

	Butte (Butte Co.)	*Soeth (Glenn Co.)*	*Tonzi Ranch (Amador Co.)*
Slope (%)	20	70	-
Density (stems/ha)	464	518	144
Basal Area (m^2/ha)	15	9	11
Mean annual precipitation (cm)	52	55	56
Soil depth	45	70	75

Sources: Kertis et al. (1993); Ma et al. (2007).

We use the historic growth trends of blue oaks at these two sites to scale our carbon gain estimates to the age of the hypothetically planted oaks. We employ the information on mean basal area (BA) per stem and total number of stems per hectare from Ma et al.'s site to calculate the mean stem BA per hectare for their site. Similarly, we use the annual BAI reported in Kertis et al. to calculate historic mean BA per stem during the first 90 years at their two sites (see dark green lines in Figure 3.3) and then multiply the values by the stem density for their Butte and Glenn County sites to calculate the history of total stem BA per hectare for the oaks. We estimate equations that describe the historic trends in total mean BA per stem at the two sites (see polynomial lines and equations in Figure 3.3). We then scale our low and high carbon gain estimates to the oak growth at the Butte and Soeth sites by multiplying for each of the first 90 years of oak growth the scenario values by the ratio of the total BA per hectare of the respective site in a given year and the total BA per hectare at Ma et al.'s site (10.656 m^2 per hectare).[11] These estimates of annual NEE are constructed using the estimated curves that describe the total BA/stem as a result of tree age (see the polynomial functions in Figure 3.3), not the actually-observed mean BA/stem. This slightly smoothes the estimated annual NEE (compare the polynomial to the Total BA/stem line in Figure 3.3). It also allows us to generate estimates of average annual projected NEE for oak restoration that are not based on the year-to-year historic variability in annual BAI at the particular sites.

The resulting scaled carbon gain scenario estimates show the expected increase in annual net carbon gains from the planting of blue oaks. Gains are close to zero in the beginning ten to 20 years and then steadily increase over the 90-year period analyzed here (Figure 3.3).

[11] Ma et al. report an average of 144 stems per hectare for their site and a mean BA per stem of 0.074 m^2 ($\pm$0.0869 m^2).

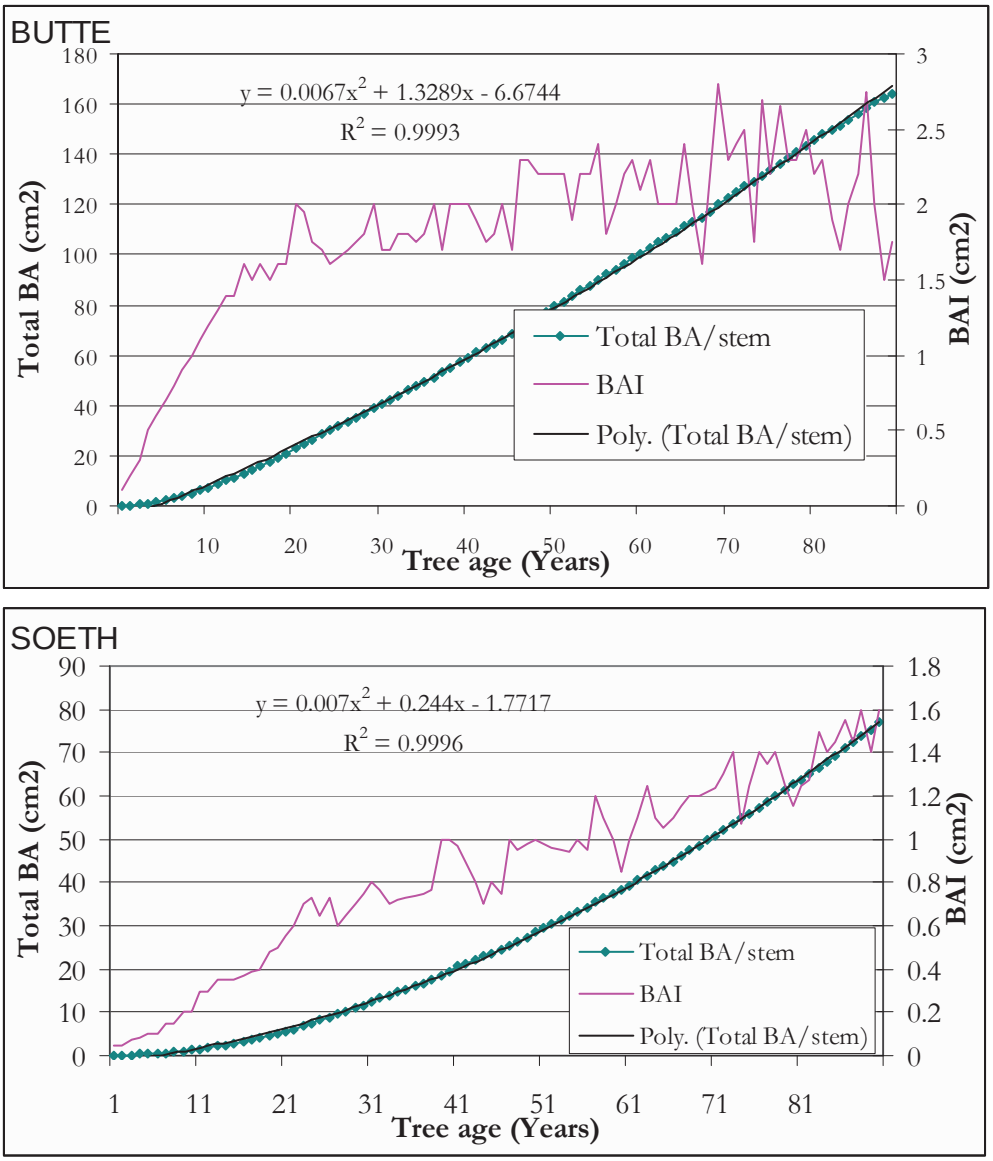

Figure 3.3: Actual BAI and mean BA/stem at Butte and Soeth sites studied by Kertis et al. (1993) and polynomial equation best describing growth in mean BA/stem at each site

Combining our low and high carbon gain scenarios with our two sites for which we have tree growth information (Butte and Soeth) yields four possible carbon gain trajectories (Figure 3.4). Of these, we select the "Butte-High" and "Butte-Low" trajectories for use in our carbon gain estimates for oak planting in our study area. The "Soeth-Low" trajectory – the one in Figure 3.4 that falls outside the carbon gain range covered by the "Butte-High" and "Butte-Low" trajectories – is the result of combining the very conservative low carbon gain scenario – based on the only study in Table 3.5 reporting a net carbon uptake by an exotic annual grassland (Xu and Baldocchi, 2004) – with oak growth estimates from a particularly unproductive site. For this reason, we exclude the "Soeth-Low" trajectory from our estimates of potential carbon gain from oak planting in our four counties.

Even after eliminating the lowest of the four carbon gain trajectories, the spread between the high and low carbon estimates for reestablishment of oaks on grasslands remains very large, due to the almost five-fold difference between the high and low carbon gain scenarios for oak planting.

Carbon in the California Blue oak savanna is cycled much more slowly than typical of savanna ecosystems, with the low productivity of the California Blue oak savannas being more typical of arid tropical tree-grass sites (Battles et al., 2008). As a result, the expected total above- and belowground net carbon uptake associated with the oak planting on grasslands in our study area is rather low, with values that reach only around 0.12 tons per-acre per-year (low scenario) to 0.55 tons per-acre per-year (high scenario) (Figure 3.4).

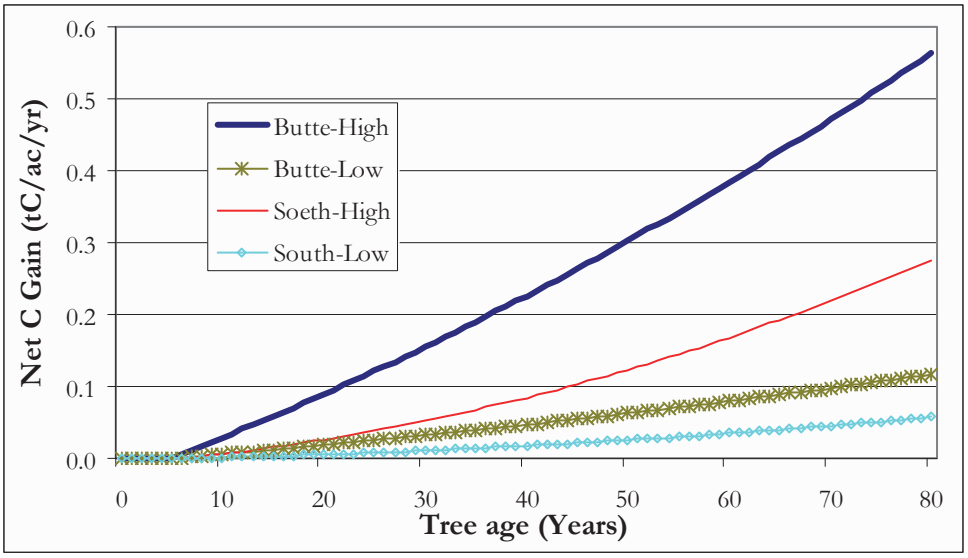

Figure 3.4: Projected carbon gain trajectories based on high and low net gain scenarios for oak planting on grasslands and high (Butte) and low (Soeth) tree growth sites

Nevertheless, over decadal time spans, the total amount of carbon that could be taken as a result of oak reestablishment is substantial, with an acre of savanna accumulating an estimated seven to 33 tC over the first 100 years after oak establishment (Table 3.7). By comparison, current blue and interior live oak (*Quercus agrifolia*) woodlands in our study region are estimated to contain on average 31 and 46 tC per hectare, respectively in tree biomass alone (Gaman, 2008). These numbers do not include downed woody debris or duff or litter layers, which together are estimated to account for another 42 and 37 tC per hectare in blue and interior live oak woodlands, respectively (Gaman, 2008). These tree and duff and litter layer carbon pools together are similar in size to our high carbon gain estimate (82 tC/ha) from oak planting on grasslands. However, they still do not take into account the increase in soil organic carbon associated with oak establishment. For example, in their study of live oak encroachment on grasslands in central Texas, Jessup et al. (2003) found that concentrations and densities of soil organic carbon were generally greater in woody patches than in grasslands. Similarly, Dahlgren et al., (1997) found that compared to adjacent grasslands, soils beneath oak canopy have greater concentrations of organic carbon. Jackson et al. (2002) found that the proportional change in soil carbon after woody plant invasion of

grasslands was negatively related to precipitation, with wetter sites losing and drier sites gaining soil carbon after invasion.

Table 3.7: Estimated cumulative average net carbon uptake per acre from oak planting on grasslands in the study area

Scenario:	*High C Gain*	*Low C Gain*
Time span	*tC/ac, cumulative*	
10 yrs	0.1	0.0
20 yrs	0.7	0.1
30 yrs	2.0	0.4
40 yrs	3.9	0.8
50 yrs	6.6	1.4
60 yrs	10.0	2.1
70 yrs	14.3	3.0
80 yrs	19.6	4.1
90 yrs	25.7	5.3
100 yrs	32.9	6.8

With mean annual precipitation of their driest sites approximately four times that of the sites on which our carbon gain estimates are based, the relative size of increases in soil organic carbon would be expected to be even higher in our study sites than those observed by Jackson et al. (2002). These findings have been confirmed in central California by the Marin Carbon Project, which found that woody plants increased rangeland soil carbon by about 30 percent (Silver, 2009).

Thus, our high average per-acre carbon gain estimate is likely to be a more realistic representation than our low estimate of the carbon accumulation that would be expected to result from oak planting.

Total potential carbon sequestration through oak reestablishment and afforestation on study area grasslands

To develop first approximations of the total amount of carbon that could be sequestered through oak reestablishment and afforestation in Butte, Glenn, Shasta and Tehama counties, we first calculated total grassland acreage in each county using California Land Cover Mapping & Monitoring Program (LCMMP) GIS data (California Department of Forestry and Fire Protection, 2009). The current spatial distribution of oak woodlands and grasslands is shown in Figure 3.5, which is an excerpt from a Fire and Resource Assessment Program (FRAP) land cover map (California Department of Forestry and Fire Protection, 2003). Most of the oaks in the study area are blue oaks followed by black (*Quercus kelloggii*) and live (Interior and Canyon) oaks (Gaman and Firman, 2006).

Next, we assumed that all grasslands located within the potential modern distribution of blue oak (Kueppers et al., 2005) are suitable for oak planting. This includes all grasslands in Butte and Shasta Counties, and an estimated 90 percent of grasslands in Glenn and Tehama Counties (Table 3.8). However, these estimates do not take into account projections of climate change-induced shifts in oak habitat. A visual assessment of the climate change-induced projected changes in oak habitat by 2080-2099 from Kueppers et al.'s (2005)

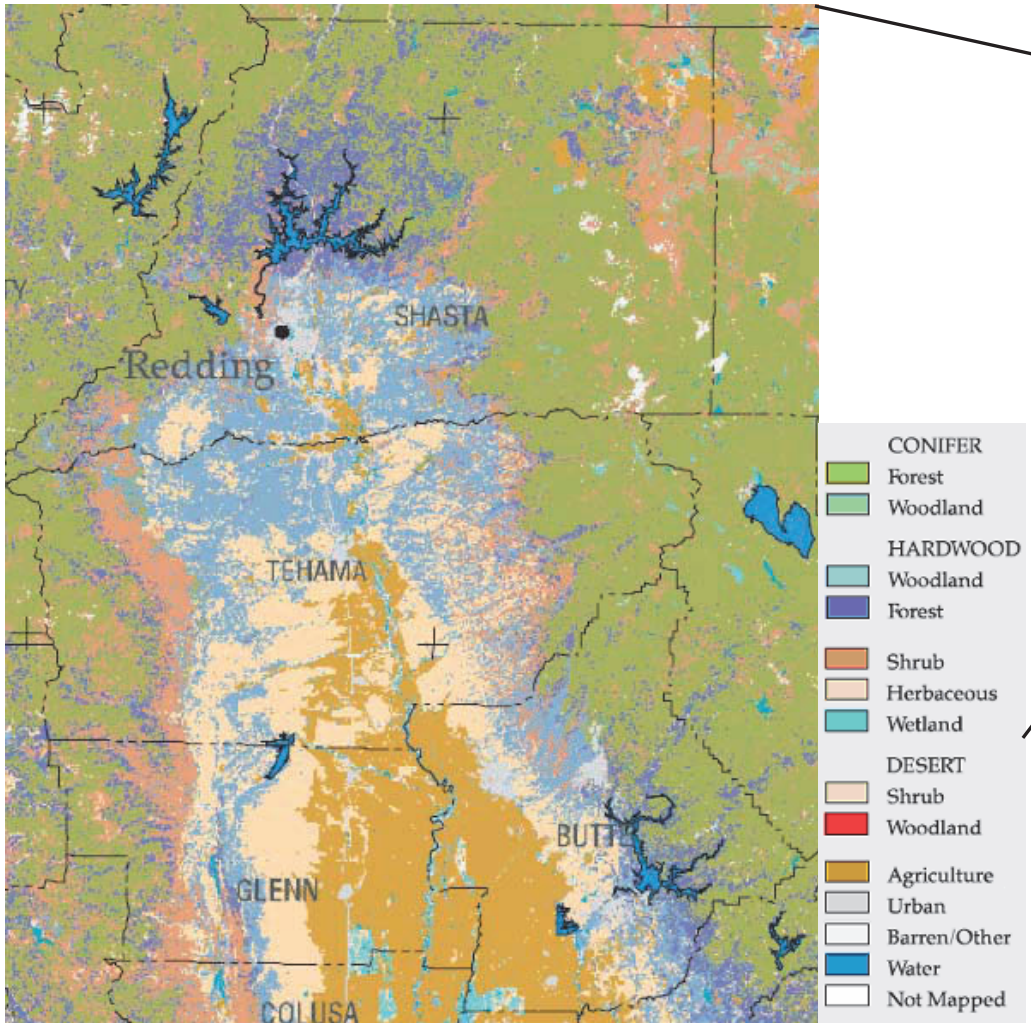

Figure 3.5: Land cover in the four study area counties
Source: California Department of Forestry and Fire Protection (2003)

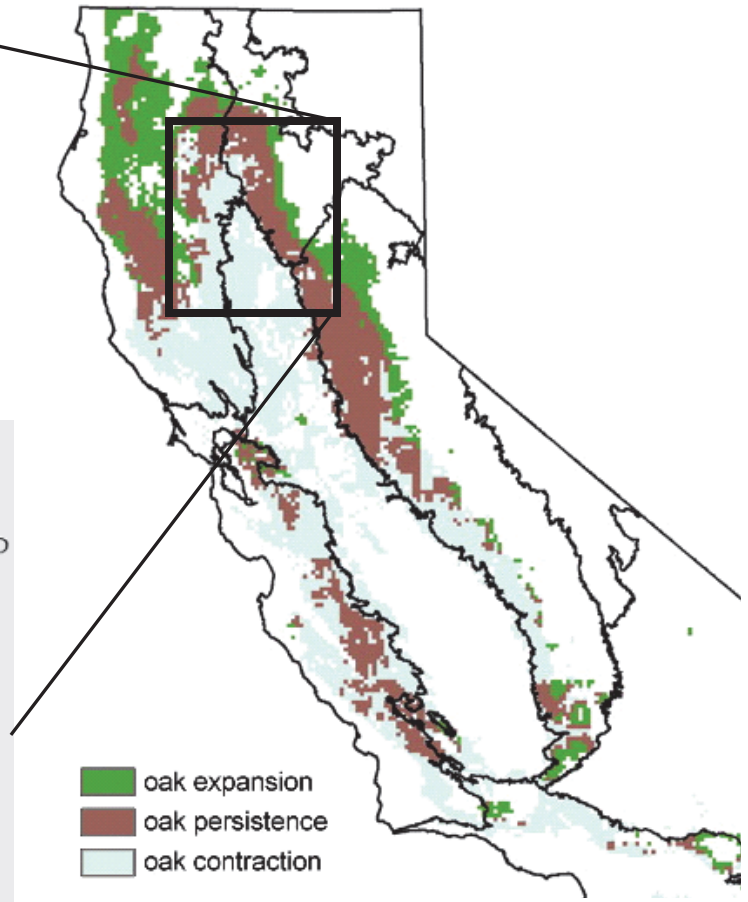

Figure 3.6: Kueppers et al.'s (2005) projections of changes in blue oak habitat in California based on regional climate change model
Source: Fig. 2 A in Kueppers et al. (2005)

regional climate model-based analysis suggests that estimated acreage suitable for oak reestablishment and afforestation may be reduced by around 50 percent in Butte and Shasta Counties, 75 percent in Tehama County and 100 percent in Glenn County (Figure 3.6). These two sets of estimates form the basis for our High and Low scenarios, respectively for oak plantings (Table 3.8).

Table 3.8: Grassland acreage in the study area by county, and percentages and acres potentially suitable for oak planting

County	*Acres*	*Potentially suitable for oak planting*			
		Scenario 1 (High)		*Scenario 2 (Low)*	
		Percent	*Acres*	*Percent*	*Acres*
Butte	77,768	100%	77,800	50%	38,900
Glenn	170,570	90%	153,500	0%	0
Shasta	127,333	100%	127,300	50%	63,700
Tehama	438,757	90%	394,900	23%	98,700
Total	814,428		753,500		201,300

Note: Includes both wet herbaceous (wet meadows and Tule-Cattail-Sedge) and dry herbaceous grasslands (annual grasses and forbs). See text for scenario details.

Some current shrub lands may convert to grasslands under the projected temperature and precipitation changes, and may become potential candidates for oak plantings (see green areas in Figure 3.6). Thus, our Kueppers et al.-based afforestation estimate, which does not consider these potential new candidate lands for oak plantings, likely understates the acreage available for oak planting under climate change projections.

In the High scenario, an estimated 753,000 acres are suitable for oak planting; in the low scenario, this area is reduced to just over 200,000 acres.

Multiplying the potential cumulative net carbon uptake (Table 3.7) by the high and low acreage scenarios, respectively (Table 3.8) yields our estimates of total (above and belowground) potential net carbon uptake that could be achieved by planting oaks on the grasslands in the four-county study area (Table 3.9). Over the 100-yr time horizon commonly used in CO_2 offset calculations, planting of oaks on study area grasslands could sequester between 1.4 and 24.8 million tC, or 5 to 91 millions tCO_2e, respectively. The assumptions underlying the High and Low total net sequestration estimates are summarized in Figure 3.7. Both of these estimates are somewhat extreme as they assume, respectively, that all grasslands that are ecologically suitable for oak planting could in fact be planted with oaks and that net carbon accumulation on these lands will occur at the higher of the two rates reported in the literature (High scenario), or that net carbon accumulation per acre will occur at the lower of the two rates reported in the literature and that only a fraction of currently suitable lands will remain suitable due to climate change (Low scenario). The average of the sequestration values from these two scenarios, 13 million tons of carbon (48 million tCO_2e) perhaps is a more realistic figure for assessing oak planting potential. This amount is equivalent to an average annual sequestration of 480,000 tCO_2e, or 0.1 percent of California's 2006 greenhouse gas emissions of 484 million tons of CO_2e (California Air Resources Board, 2009).

Table 3.9: Total potential net carbon uptake from planting of oaks on study area grasslands

Cumulative uptake per acre:		*High*	*Low*	*High*	*Low*
Acreage suitable for oak planting:		*High (Scenario 1)*		*Low (Scenario 2)*	
			1000 tC		
		High Est.			*Low Est.*
Years:	10 yrs	66	14	18	4
	20 yrs	530	110	142	29
	30 yrs	1,472	305	393	82
	40 yrs	2,932	608	783	162
	50 yrs	4,949	1,026	1,322	274
	60 yrs	7,564	1,568	2,020	419
	70 yrs	10,816	2,243	2,888	599
	80 yrs	14,745	3,057	3,938	816
	90 yrs	19,391	4,021	5,178	1,074
	100 yrs	24,795	5,141	6,621	1,373

Notes: Based on Tables 3.7 and 3.8.

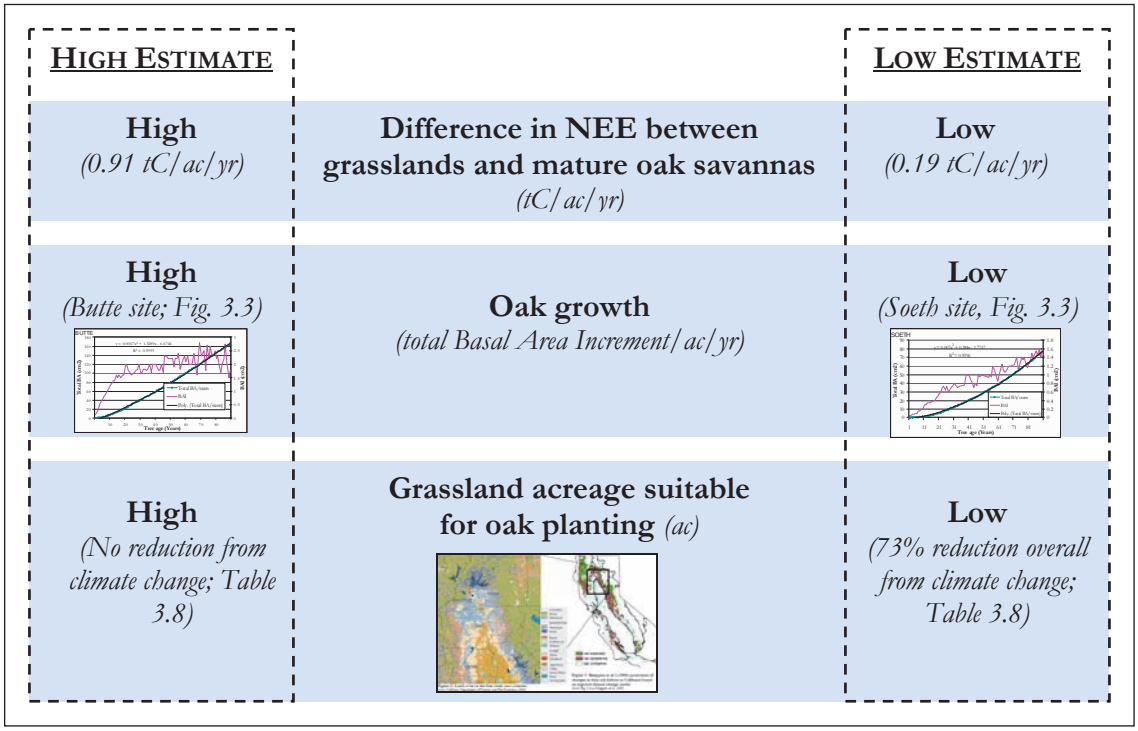

Figure 3.7: Major assumptions underlying the net carbon uptake estimates for oak planting

How do these numbers compare to the carbon bound up in current oak savannas? A recent comprehensive assessment of oaks in California (Gaman, 2008) estimates that oak woodlands in the four counties contain approximately 17.2 million tons of carbon in live and dead trees (not including downed logs, litter and soil borne carbon) (Table 3.10).

Table 3.10: Total carbon stored in oak tree biomass in study area counties

County	*Total tC*
Butte	3,283,286
Glenn	1,341,899
Shasta	4,955,757
Tehama	7,616,397
TOTAL	17,197,339

Source: Gaman (2008)

This value is about a third more than the mean of our high and low estimates of 13 million tons of carbon. Thus, based on our analysis, oak afforestation in the four counties over 100 years could absorb an amount of atmospheric carbon roughly similar to that already stored in current oak woodlands in these counties.

Our estimates also are in line with the estimated 103 million tons of carbon in tree biomass alone that reforestation of California's oak woodlands may sequester (Gaman, 2008). Thirty-eight percent of oak woodland and forest plots in Gaman's (2008) analysis fell into non-forest inclusion areas in oak woodlands that may be suitable for reforestation. Reforestation of these 4.9 million acres (38 percent of the total statewide mapped oak woodland and forest acreage of 12.9 million acres) would result in a carbon accumulation in tree biomass of an estimated 103 million tons (Gaman, 2008). This equates to 21 tC per acre, compared to our mean estimate of 27 tC per acre, with the difference between the two estimates likely due to the omission in Gaman's estimates of soil carbon increases.[12] Gaman (2008) estimates that through measures including interplanting, enhanced grazing management and conservation-based sustainable forestry, and through continued sequestration on existing oak lands, California's oak woodlands and forests combined have the potential to sequester up to a billion tons of carbon in the 21st century.

Water quality benefits of oak reestablishment

Planting oak trees on rangelands increases ground cover and root structures, both of which are beneficial to water quality of surrounding surface waters. Erosion and runoff from rangelands need to be controlled in order to limit emissions of pathogens (i.e. *E. coli, C. parvum*), sediment and nutrients (nitrogen and phosphorous) into waterways (Hubbard et al., 2004). Water bodies on rangelands provide human and livestock drinking water as well as fish habitat, all of which are uses that can be adversely impacted by runoff of pathogens, nutrients and soil particles.

The root structure of mature blue oaks can decrease soil erosion on rangelands by binding together the soils of watersheds (Burns et al. [1990], cited in Sacramento Valley Conservancy [no date]). Similarly, the practice of oak removal has been shown to negatively impact soil stability, with clearing of oak trees from California oak savannas leading to excessive soil

[12] 13.1 million tC divided by our mean suitable acreage of 477 thousand acres (Table 3.8).

erosion (Bartolome et al., 1994; IHRMP, 1998). One study reported that removal of oaks led to a 59 percent increase in runoff (Pitt et al., 1978).

Research indicates that many rangeland owners appear to be aware of the beneficial impacts of oaks on water quality. A survey conducted in 1985 indicated that 64 percent of owners of parcels under 5,000 acres thought that blue oaks had value for erosion control on their land (Huntsinger and Fortmann, 1990).

Wildlife benefits of oak reestablishment and afforestation

Oak savannas are a crucial component of California's Mediterranean climate region and provide habitat for a large variety of species. These are described in detail in Chapter 1.

Social economic and private financial value of benefits from oak planting

This section has presented an overview of some of the benefits provided by oak afforestation, focusing specifically on increases in forage production, carbon sequestration, water quality and wildlife habitat. Where possible, we developed quantitative estimates of these benefits, as in the case of forage production and carbon sequestration. In the cases of water quality and wildlife habitat, quantitative analyses could not be completed for this study. Nevertheless, the values associated with water quality and wildlife habitat benefits are very real and are easily identifiable.

Lower soil erosion rates on oak savanna rangelands lead to a reduced silt loading of streams and downstream water bodies in the watershed. This in turn increases the attractiveness of these water bodies for recreational uses (angling, swimming, boating) and thereby the enjoyment of recreationists. Given the number of water-based recreation participants downstream of our study area and with the net value of a water-related recreation day to participants (in economics referred to as the consumer surplus) ranging from around $30 to $50 per day in the Pacific coast region (Loomis, 2005), it is likely that reductions in sediment loading carry substantial aggregate recreational economic value.

Reducing soil erosion also slows the siltation of downstream reservoirs, canals and streams and thus reduces the need for costly dredging and the loss of ecosystem services like provision of habitat for wildlife dependent on healthy river channels and visibility. As of 2006, California had a total of 87 waters classified as impaired due to sediment loading, many of which are located in or downstream of our study area (U.S. EPA, 2009b). As of March 2007, the state had a total of 56 Total Maximum Daily Load (TMDL) limits in place for sediment, siltation or a combination of the two (ibid.), although none of these are located in the four counties that comprise our study area. In other areas of the state where rangelands lie upstream of sediment TMDL-limited areas, reductions of soil erosion from rangelands into water bodies resulting from rangeland management practices such as oak conservation or afforestation or riparian buffers can have real economic benefits for sediment point sources bound by TMDLs. In those cases, rangeland conservation practices effectively reduce the severity of the sediment TMDL limits imposed on point sources, and thereby result in reduced compliance costs for those sources. In several areas of the country, owners of agricultural lands are compensated by regulated point sources for management activities on their lands that produce water quality improvements. Examples for such point-source

non-point source water quality markets are found for nitrogen in the Susquehanna River and Connestoga watershed in Pennsylvania, the Kalamazoo River in Michigan (World Resources Institute, 2009) and for water temperature in the Tualatin River in Oregon (Oregon Department of Environmental Quality, 2009), to name but a few, and further ones currently are being designed for example for the Ohio River (Electric Power Research Institute, 2009).[13] These markets have the dual benefits of potentially achieving water quality goals more efficiently and allowing land owners to reap economic benefits from activities that provide additional benefits for society – in our case, from oak afforestation and riparian restoration or planting.

Even in the absence of any TMDLs for sediment, there are a total of 43 major dams in our four-county study area (National Atlas of the United States, 2009). Sedimentation of reservoirs imposes economic costs in the form of a reduction in the quality and availability of the services provided by reservoirs, such as recreation, electricity generation, water provision and flood control (Hansen and Hellerstein, 2007). Hansen and Hellerstein (2007) estimated that across the 2,111 U.S. watersheds, a one-ton reduction in soil erosion provides reservoir benefits ranging from zero to $1.38. As the authors note, this estimate only includes a small portion of total soil conservation benefits. The cost of dredging of reservoirs is highly variable, with $2.70 per ton to $41.90 per ton reported in the literature (ibid.; all values expressed in 2000 dollars). Thus, reductions in erosion from rangelands through oak planting and other practices are likely to lead to substantial avoided costs in our study area.

Because of the importance of study area's blue oak savannas and woodlands for terrestrial wildlife (see chapter 1) in addition to aquatic species, oak conservation, reestablishment and afforestation generate economic benefits in the form of terrestrial (mammal and bird) wildlife populations that are directly used by humans for recreational purposes (hunting, wildlife viewing) or are valued simply because of their existence (passive use values).

Wildlife-associated recreation activities attract large numbers of participants both in California (U.S. Fish and Wildlife Service and U.S. Census Bureau, 2007a) and nationwide (U.S. Fish and Wildlife Service and U.S. Census Bureau, 2007b). These activities generate substantial net benefits for participants (Aiken, 2009) and as well as large economic impacts in the local and regional economies (Carver, 2009; Leonard, 2008). By providing habitat for the species supporting these activities, oak savannas and woodlands directly contribute to these values and impacts.

Forage production and carbon sequestration

Increases in forage production resulting from the presence of blue oaks on rangelands generate benefits for land owners or lessees to the extent that the increase in forage availability leads to increased stocking rates or reduced feed costs. As discussed in the section on forage impacts of blue oaks on California's rangelands, the literature suggests that in California's Mediterranean climate uplands, blue oak cover on lands with up to 40-60

[13] A complete list and program information of the water quality trading programs operating in the U.S. can be found at Environmental Trading Network (2009). See also Selman et al. (2009) for a review and analysis of these programs.

percent oak canopy cover may result in an average increase in forage of around 0.5 AUM per acre. At the average 2008 price paid per AUM on private lands in California of $17.80 (National Agricultural Statistics Service [NASS], 2009), the potential value of this increased forage production was $8.90 per acre.

The forage benefits from blue oaks reported in the literature are for mature trees. Thus, the estimated value in 2008 of this increase was $8.90 per acre on uplands is a realistic *average* estimate for mature oak savannas. In contrast, newly planted oak acorns or seedlings will not produce these benefits until their root structures are sufficiently developed to alter soil nutrient cycles in their locations, something that is likely to take several decades even under favorable conditions.

The economic value of net carbon uptake from oak restoration and afforestation is more difficult to quantify than that of forage increases. This value consists of reduced damages from climate change-related impacts. Although projections of potential impacts and their associated economic costs are available, including for California (Climate Action Team, 2009), at this time the uncertainties surrounding these cost estimates and the marginal damages caused by changes in greenhouse gas emissions are too large, and the impact of oak planting on rangelands in our study area on overall emissions too small, to allow the generation of reasonably defensible estimates of avoided climate change costs due to oak afforestation.

Even though the full, social economic value of reductions in atmospheric concentrations of greenhouse gases resulting from oak planting is difficult to estimate at this time, carbon sequestration through oak planting on rangelands has the potential to generate private financial benefits for rangeland owners. Currently, voluntary carbon markets allow landowners to sell so-called carbon offsets - financial instruments aimed at a reducing greenhouse gas emissions. The volume of carbon offsets transacted on voluntary carbon markets globally has recorded strong and continued annual growth since 2003, with the bulk of that growth in the last three years occurring on the Chicago Climate Exchange (CCX) (Hamilton et al., 2009). Voluntary over-the-counter (OTC) markets, which handle deal-by-deal transactions of more tailored offsets than those that take place in the CCX also have recorded strong growth since 2003, and as of 2008 still accounted for over half of all offset transactions globally. Both markets allow landowners to sell emission offsets resulting from carbon sequestration, with the major difference between the two being that CCX offsets satisfy well-published and consistent standards for credit calculation and verification. Currently, the CCX only accepts credits for changes in soil carbon stocks on rangelands for prescribed changes in management - sustainable stocking rates, rotational grazing and seasonal use (CCX, 2009), with standardized credit rates (0.12-0.32 metric tCO_2 per acre per year) for particular project types and locations. Currently, projects in Butte, Glenn and Tehama counties are eligible for offset generation, while those in Shasta are not (CCX, 2009). Thus, the CCX currently does not permit the generation of offsets via oak afforestation. However, OTC markets do accept afforestation and reforestation credits, and in 2008 the U.S. was home to the majority of OTC afforestation and reforestation projects worldwide (Hamilton et al., 2009). The average OTC price in 2008 for projects in the U.S. was $6.9 per ton of CO_2e (ibid.). Thus, based on our 100-yr per-acre carbon gain scenarios for rangelands in our study area (Table 3.7), rangeland oak afforestation projects would have generated credits worth between $47 and $227 per acre. Since credits generally are accrued

with actual sequestration, landowners would not have received these amounts as lump sums, but rather over time. Furthermore, due to the comparatively slow growth especially of blue oaks, annual payments would be small in the early years and even decades, and then would increase with increasing total annual carbon sequestration by maturing oaks. These payment levels probably are an underestimate of what oak afforestation could yield in land owner income from the sale of OTC carbon offsets because most analysts expect carbon prices to increase (e.g., see New Carbon Finance, 2008; U.S. EPA, 2009c). The generated credits would have accounted for a very small share of total OTC transaction volume and thus would not have affected average credit prices. Furthermore, the demand for offsets is likely to increase, as the California Global Warming Solutions Act (AB32) encourages voluntary greenhouse gas reductions, and California's Forestry Greenhouse Gas Accounting Protocols will register reforestation projects (Climate Action Reserve, 2009).

Voluntary carbon markets pale in comparison to regulated markets, with the former accounting for less than three percent of the total global volume of transactions in 2008 (measured by weight of CO_2e). In the U.S., there are several currently operating and emerging state and regional regulatory carbon markets, and a federal cap-and-trade based market is expected to develop. Development of these regulatory carbon markets will drive up demand and prices for offsets on both regulated and voluntary carbon markets. For example, under the Western Climate Initiative (WCI) that comprises 11 partner and 14 observer states and provinces from Nova Scotia to Mexico, including California, up to 49 percent of reductions may initially be achieved through offsets (California Air Resources Board, 2008). Thus, it is likely that planting of oaks on California rangelands will become more financially attractive for land owners than it currently is. Even under current carbon prices, however, such afforestation may make economic sense for land owners as long as the associated transaction costs are not too onerous. Meeting of the latter condition becomes more likely over time as familiarity of landowners with carbon markets increases and technical advances make offset verification cheaper and more efficient.

Cost of oak planting

Guides for landowners on how to artificially regenerate oaks are readily available (for example, McCreary and Nader, 2007). Unfortunately, information on the cost of this regeneration is much more difficult to obtain.

Below we present the one example of oak reestablishment costs that was available (Table 3.11). The data are from an oak planting project on California Audubon's Bobcat ranch. In this particular case, 200 acorns were planted and protected using herb control and tree shelters. The cost of this planting is substantial, at around $100 per acorn. On the other hand, the project includes all the measures recommended in the literature to ensure high acorn and seedling survival and growth rates. Private landowners may be able to reduce planting costs compared to those listed in Table 3.11 if they are able to reduce labor costs to below the $50 per hour used for landowner, Audubon and FWS labor in the example. For example, at $15 per hour for these labor inputs, costs per planted acorn drop by half, to $51.

Per-acre costs from this example can be extrapolated to other sites by incorporating particular desired target tree densities and acorn survival rates from artificial regeneration projects reported in the literature.

Table 3.11: Budget of oak restoration project on Bob Cat Ranch, California

Description	*Quantity*	*Unit*	*Cost/Unit*	*Total*
Services and Labor				
Labor – Landowner, Audubon, US FWS	280	hours	50	$14,000
Hired crews (cage installation)	80	hours	10	$800
Equipment rental	40	hours	40	$1,600
Subtotal services and labor				$16,400
Supplies and Expendables				
Acorns	200	unit	0.25	$50
Tubex tree protectors and stakes	100	units	3.65	$365
Native grass straw	20	bales	9	$180
Round-up herbicide	128	oz	0.5	$64
Woven wire 12 gauge 4x2 4ft tall	9	rolls	128	$1,152
T-Post 6ft	200	post	4.15	$830
Electrical wire	1	rolls	15	$15
Irrigation hose	4	rolls	100	$400
2 inch PVC	200	feet	0.6	$120
Pressure compensating emitters	100	units	0.45	$45
Miscellaneous irrigation supplies	1	lump	200	$200
Bird boxes (4 blue bird and 1 owl)	6	lump	40	$240
Subtotal Supplies and Expendables				$3,661
TOTAL				$20,061

Source: Personal Communication. Chris Rose. California Audubon Society. May, 2008.

Table 3.12 shows the survival rates of a number of artificial regeneration projects, for both acorn-based and seedling-based plantings. Since survival rates differ for the two approaches, we use only the rates reported for acorn planting projects - 33 percent (Kraetsch, 2001) and 75 percent (Tecklin et al., 1997).

Table 3.12: Survival rates for artificial blue oak regeneration

	Survival rate	*Treatment*	*Source:*
Planted blue oak acorns that become strong saplings after 4-5 years	33%	Screens, tree shelters	Kraetsch (2001)
Three-year survival rate of seedlings	45%	Screens, herb control	Adams et al. (1991)
Three-year survival rate of acorns	75%	Treeshelters, herb control	Tecklin et al. (1997)
Three-year survival rate of seedlings	88%	Treeshelters, herb control	Tecklin et al. (1997)
Four-year survival rate of seedlings	58%	Treeshelters, fencing	McCreary (1999)

Multiplying the 200 planted acorns from the example presented in Table 3.11 by the average of these two survival rates (54 percent) yields an expected 108 acorns that mature into robust seedlings. These can be planted at desired densities. Table 3.13 presents per-acre cost examples for acorn-based oak plantings at densities ranging from 27 to 108 stems per acre.

Table 3.13: Cost per acre of oak plantings for different tree densities

Oak density (Stems/acre)	*Project acreage at given seedling density*	*Cost/acre*	
		*High **	*Low **
108	1	$20,061	$10,261
54	2	$10,031	$5,131
27	4	$5,015	$2,565

* High costs are based on labor costs shown in Table 3.11; low costs are based on reduced costs of $15 per hour of landowner labor input.

The corresponding costs range from around $2,600 to $20,000 per acre, depending, in addition to tree density, on labor cost. On rangeland sites in California, typical oak planting costs (at the labor cost rates shown in Table 3.11) are $3,000 to $6,000 per acre.[14]

The foregoing numbers indicate that the cost of planting oaks on grasslands in our study area is likely to be one to two orders of magnitude higher than the revenues, even without taking into account that the landowner would not receive a large portion of those revenues for many years, whereas all of the costs are incurred upfront. This result is partly due to the rather low productivity of the Mediterranean climate grasslands in the study area, the low growth rates of blue oaks, the relatively low present prices on carbon markets accessible to grassland owners, and the absence of sediment and nutrient water quality trading programs in the area. Thus, from a private financial perspective, at this point in time oak afforestation of these grasslands does not make economic sense for private landowners. Even under cost share programs that cover 50-75% of the total cost of oak planting, most such planting projects likely would not be financially viable at this time.

From the perspective of society at large, the economic rationale for oak planting may be much more favorable given that such planting can provide both offsite (e.g., water quality) and onsite (e.g., wildlife habitat) public benefits. Oak planting thus exemplifies the fairly widespread problem of a divergence of privately- from socially-desirable land management actions. If socially-desirable actions are to be achieved – oak planting in this case – it is necessary to align private incentives with social objectives. This could be achieved through several different approaches, including higher cost shares, the creation of missing markets (for example, water quality trading), or payment programs that compensate landowners for implementing desired management actions. Examples of the latter already exist in the form of federal and state conservation payment programs. However, there may be an economic case for increasing the payment levels in situations where the payments do not reflect the full value of the third-party benefits generated by private management actions and where they are insufficient to bring about the socially-desirable management types or levels of conservation actions.

[14] Personal communication. Vance Russell, California Audubon Landowner Stewardship Program. August, 2009.

C. Restoration of Native Perennial Grasses

In the past two hundred years the grasslands in California's Central Valley have gradually changed from predominantly native perennial species to a landscape dominated by non-native annual species. Only two percent of California grasslands contain native perennials (USDA, ARS) and these areas most likely are not "pristine" native grasslands, but contain some non-native species (Stromberg et al., 2007). There are many questions regarding the feasibility of restoring native perennials to California grasslands. Many argue that restoring native perennials to the landscape will increase forage value (Bartolome, 2007; Dyer, 2002), wildlife habitat and biodiversity, and improve soil ecosystems and functions (Menke, 1992; D'Antonio and Myerson, 2002). But, what constitutes a restored grassland ecosystem varies by location (Stromberg et al. 2007), and restoration requires long-term management with variable results (Brown and Rice, 2000; Bartolome et al., 2007; Stromberg et al., 2007). This section will briefly discuss the historical landscape changes within California rangelands, successful restoration case studies within Shasta, Butte, and Tehama counties, and the effectiveness of three selected restoration practices along with their associated costs. These practices include prescribed burning, managed grazing, and re-seeding native grass species.

Historical landscape

Before the 1800s, large areas of grasslands were burned regularly by Native Americans (Anderson, 2005) and grazed by large herbivores like elk and antelope (Stromberg et al., 2007) defining the surrounding landscape. After the early 1800s, European settlement began changing the landscape and populations of native perennial grasses started declining. There are many plausible reasons for the shift from native perennials to non-native annuals and most likely it is a combination of many factors (Bartolome et al., 2007). With European settlement, fires were suppressed, roaming wild megafauna were replaced with intense livestock grazing, and rangelands were converted to agriculture and urban uses (DiTomaso, 2000, 2006). These changes in land use benefited non-native annual grass species, which have higher rates of growth and reproduction, enabling them to invade disturbed sites much more quickly and efficiently than native perennials (D'Antonio and Myerson, 2002).

Another theory about what may have contributed to the decline of native perennials is the introduction of viruses from European agriculture. These viruses target native perennials and in some locations where they are prevalent native perennials may only be able to exist in small populations within larger populations of non-native grasses (Malmstrom, 2005).

The proliferation of suburban development, road building (Gelbard and Belnap, 2003; Gelbard and Harrison, 2005), and the expansion of the ranching industry (DiTomaso, 2006) over time slowly changed soil structure, biodiversity, and forage value of California rangelands (Menke, 1992; D'Antonio et al., 2007). Presently, many rangelands contain invasive non-native annuals like yellowstar thistle (*Centaurea solstitialis*), medusahead (*Taeniatherum caput-medusae*), and barbed goatgrass (*Aegilops triuncialis*), which are not palatable to livestock after maturation and require intensive management practices to control spreading. Also, they tend to be fierce competitors with native perennials, monopolizing resources and decreasing the biodiversity and habitat for wildlife (DiTomaso, 2000). There are some native perennial grasses that compete well with non-native annuals when management practices such as burning and grazing are introduced. The native, purple

needlegrass (*Nassella pulchra*) in many restoration cases rebound successfully (Dyer and Rice, 1997). However, purple needlegrass does very well in disturbed sites and careful management is required to not create a monoculture at the expense of a diverse stand of native grasses and forbs.[15]

Native grass restoration practices

Successfully restoring native perennial grasses and/or increasing biodiversity of native species including annual species to Central Valley grasslands requires consideration of many factors such as current site conditions (soil, slope, water availability, species diversity), historical use of the site, and future land managers objectives (DiTomaso, 2000; D'Antonio, 2002; Stromberg, 2007; Lulow, 2007). All require significant investment of time and money from a landowner. Given that these landscapes are dynamic and constantly changing, a landowner must be flexible and adaptive to various circumstances and conditions (Bartolome, 2004).

Within Butte, Shasta, and Tehama there are some restoration projects that represent successful examples of restoration practices (Stromberg et al., 2007). Llano Seco Tract 1 owned by the USFWS in Butte County was burned in 1999 and disking and herbicide were applied to the site before it was drill seeded in 2002. In 2006 the USFWS reported close to 90 percent cover of six native grasses. Turtle Bay Discovery Park, owned by the City of Redding in Shasta County, was treated with herbicide before it was plug planted in 2004 and achieved a good establishment of two native perennials valley: wild rye (*Leymus triticoides*) and valley sedge *Carex barbarae*) in 2006. Dye Creek, owned by the Nature Conservancy in Tehama County, was rotated between burning and grazing before it was seeded with purple needlegrass in 1997 using hay designed to reduce medusahead and improve native grass establishment. In 2006, purple needlegrass cover was close to that of remnant local stands. This project illustrates the concern of restoring native grassland species that could result in monoculture and potentially reducing the biodiversity of the site.

Other restoration projects have reported re-establishment of some non-native species and potentially needing long-term management to maintain native perennials. Sunset Ranch, owned by the Nature Conservancy in Butte County, had a cover crop of legumes before it was mowed twice in 2003 and 2004. Herbicide was applied in 2004 before it was drill seeded ten months later and twice again after the planting. In 2006, there were still problems with noxious weeds fluevellin (*Kickxia elatine*), Johnson grass (*Sorghum halepense*) and Russian thistle (*Salsola spp.*). It was recommended to avoid legume cover crops and leave land barren two years prior to planting (Stromberg et al., 2007).

Sulphur Creek, owned by the City of Redding in Shasta County, planted seven species of native perennials along one mile of stream bank[16]. The seed was hand-broadcasted in 1997 and 2005 along areas where extensive erosion control, soil stabilization, and stream bank restoration were also conducted. In 2006, yellowstar thistle was still a problem, but an excellent stand of seven native perennials was reported. Many other restoration projects have

[15] Personal Communication. Rodd Kelsey, California Audubon. February, 2009.

[16] The Sulpher Creek project does not involve grazing, but it does provide valuable cost information and an example of successful re-establishment of native perennials.

seen the re-establishment of non-native species after a few years, requiring long-term management and up-keep of restored areas. Some site examples are Audubon's Corral Pasture, Yolo County; Ranchette Private, Yolo County; Citrona Farms, Yolo County; Hedgerow Farms, Yolo County; and Russian Ridge, San Mateo County projects (Stromberg et al., 2007).

It may be more beneficial to introduce a mixture of management practices such as prescribed burnings where appropriate, grazing, and re-seeding of native perennials. These practices have mixed results and vary between site locations, so it is important to understand the ecosystem in which the restoration practice is taking place (Bartolome et al., 2007).

Prescribed Burning

The purpose of prescribed burning is to decrease populations of invasive plants such as medusahead, barbed goatgrass, and yellowstar thistle before they can produce viable seeds. The results are variable because it can be very difficult to ensure proper timing and the effectiveness of burning for the re-establishment of native perennial species depends on site location and timing of precipitation (Marty, 2005). For example, a study by Bartolome (2004) showed that prescribed burning in the coastal range grasslands had no effect on increasing native perennial species. However, DiTomaso (1999, 2006a, 2006b) argues that prescribed burning can decrease the seed banks of yellowstar thistle and effectively control their proliferation, while at the same time increasing native perennial grass cover.

Several studies have demonstrated success of burning with the re-introduction of purple needlegrass. A study by Dyer (2002) found that the seed size of purple needlegrass increased with prescribed burning, improved the likelihood of germination, and increased its competitiveness against invasive species. Lulow (2007) also recorded an increase in seed size of native perennial grasses with prescribed burning and that the increase was correlated with increased cover.

Generally, the best timing for prescribed burns is late spring when undesirable non-native annual grasses have not yet seeded (Wirka, 1999). However, burning can frequently result in a flush of extensive, dense populations of filaree (*Erodium botrys*) (DiTomaso et al., 2006a). The exception was purple needlegrass, which was able to thrive within the filaree (DiTomaso, 2006a; Lulow, 2007). Another study (Marty, 2005b) found that prescribed burning actually decreased the population of purple needlegrass after two years. Bartolome (2007) and DiTomaso (2006b) indicate that it takes 3-5 years to see viable results.

Although burning can be a very good range management tool, it is not widely used in California because of the difficulties in getting permits and associated problems related to air quality concerns and the risk of wildfires. Also, burning can be quite expensive. Out-of-pocket expenses for labor, fuel, minor equipment repairs, permits, and seed and fertilizer for firebreaks are estimated at $23 per burned acre (Connor [2003] cited in DiTomaso [2006b]).

Managed grazing

Using grazing as a management tool for restoring native plant species in California rangelands is a controversial and complex subject. There is little substantive research that

replicates and supports the conclusions of studies done in the Central Valley and elsewhere. For example, studies done within different California ecosystems like vernal pools (Marty, 2005a) and coastal grasslands (Bartolome, 2004) have shown an increase in native perennial plant communities resulting from managed grazing. A study by Hayes and Holl (2003) in the coastal grasslands showed that managed grazing increased native annual forbs, but *decreased* native perennial grass species.

Many ranches in the Central Valley practice some form of managed grazing and certain grazing practices have proven to be an effective tool in the restoration of native perennials when combined with prescribed burning (Menke, 1992). Light to moderate managed grazing for eradicating some invasive species have shown to increase aboveground biomass of native grasses, especially purple needlegrass (Marty, 2005b). Lulow (2006) argues that maintaining diversity and increasing the aboveground biomass of targeted native grass species enables them to resist invasions by non-native annuals. In general, established native grasses have shown to respond favorably to long-term managed grazing when site specific characteristics are taken into account (Jackson and Bartolome, 2007). Managing for native perennial diversity will require a mixture of grazing and non-grazing practices that will vary by location (Hayes and Holl, 2003).

The absence of livestock grazing has been shown to actually increase invasives like yellowstar thistle, medusahead and barbed goatgrass. These species are not palatable to livestock after maturation and decrease the carrying capacity of rangelands over time (Bartolome, 2007). They should be grazed while still young and before seeds are established, thereby reducing their abundance (Thomsen, 1993, 1996). Managed grazing is a relatively cheap management practice and may be a good alternative to burning for invasives control.

According to DiTomaso (2006b) the costs associated with controlling yellow star thistle with managed grazing include purchasing or leasing of the animals, maintaining them in proper health, and monitoring their grazing activity to minimize harm to desirable forage. Other expenses may include leasing land, spraying molasses to increase palatability, herding dogs, fencing, and sometimes supplemental feeding.

Seeding Native Grass Species

Seeding native grass species is expensive and requires thorough knowledge of a site, including species presence, soil type, water availability, and species life cycles (Lulow, 2007). Re-seeding with native perennials can be successful if these factors are taken into account (Corbin and D'Antonio, 2004). Two types of seeding techniques are used: plug planting and drill seeding. In general "plugs" are more expensive, but the survival rate is high at about 90 percent (Stromberg et al., 2007; Cunliffe and Meyer, 2002; Corbin and D'Antonio, 2004; Huddleston and Young, 2004). Plugs are most cost effective if used on small plots. A restoration project in Shasta County (Turtle Bay Discovery Park) used plug plantings on 28 acres in 2004. After two years there was a good establishment of two native perennials (Stromberg et al., 2007).

Drill seeding can be used for larger areas with a tractor operated seed drill. Four years after drill seeding, a restoration project in Llano Seco (Butte County) reported 90 percent cover of native grasses on 65 acres. The land was treated a few years prior to seeding with burning,

disking, and herbicides. Drill seeding is expensive for large tracts of land and in cases where there are still natives mixed-in with the exotic grasses the use of prescribed burning and managed grazing is a cheaper option. There are many cases where direct seeding has not established a good healthy stand of native perennials (see Stromberg, 2007). For example, non-natives can re-invade a site, indicating that long-term management is necessary. Jetter et al. (2003) argues that establishing native perennials will effectively control the proliferation of yellowstar thistle because they have similar water intake and root structure. However, getting an established native perennial stand takes many years and requires repeated burning and grazing of a targeted invasive species.

The costs associated with re-seeding native species include seedbed prep, seeding, and follow-up management. According to the Agricultural Research Service (USDA), a pound of native grass seed costs $40. Approximately 15 pounds of native grass seed are needed to effectively sow one acre of land, thereby costing about $600/acre.[17] Re-seeding native grass species varies by location and size of tract. For example, The Cottonwood Slough Restoration Project conducted by California Audubon is a 17-acre tract with estimated costs of $4200/acre the first year and $880/acre the second year. These costs include management, site preparation, seeding, irrigation, erosion control, habitat enhancements, and weed control. In contrast Bobcat Ranch, also a restoration project conducted by Audubon, is a 6,800-acre tract with estimated costs of $9.50/acre. Costs include staff time, fence installation, equipment rental, seeding, and weed control. According to DiTomaso (2006b), costs recorded for re-seeding in a native legume and perennial grass restoration effort at Fort Hunter Liggett were between $500 and $2000/acre. In this trial, the native species represented 5 to 30 percent of the total vegetative cover two years after seeding.

California grasslands now have a mixture of European annual species and native perennial and annual species. Non-native annual species are an integral part of the landscape and how they interact with native species and change the ecosystems is still not widely understood. Restoring native perennial grassland species to vernal pool sites and coastal grassland areas has had more success than in drier areas of the Central Valley. These site locations should be seriously considered and management practices accommodated appropriately. Range manager objectives must be seriously considered. For example, if biodiversity is an objective, purple needlegrass must be carefully managed so as not to create a monoculture in the site area. If establishing perennial grasses for forage is the objective, then the life cycles of invasives such as yellowstar thistle, medusahead, and barbed goatgrass should be well understood so as to graze and burn them before they have seeded. More research is needed on how these non-native annuals affect wildlife, soils and the pathogenic microorganisms that positively affect native perennial species (D'Antonio et al., 2007).

Forage impacts of restoring native perennials

Restoration of native perennials in California's Mediterranean climate rangelands is expected to increase forage value (Bartolome, 2007; Dyer, 2002). In one recent study, restoration of perennials led to moderate forage gains after several years, following a short-term decrease in forage immediately after the establishment of native grasses. This short-term decrease was due to the strong control measures applied to suppress annual forage grasses and noxious

[17] Personal Communication. John Anderson, Hedgerow Farms. August, 2009.

annual weeds (Malmstrom et al., 2008). Despite encouraging signs, there is a lack of sufficient quantitative information on forage impacts of perennials restoration. Thus, in this section we focus on the impact of perennials on soil carbon stocks.

Carbon impacts of restoring native perennials

California rangelands exhibit a wide range in soil carbon pools, with studies showing ranges between 20-140 t/ha in the top 50 cm, ~80-180 t/ha in the top 1 m, and 210-250 t/ha in the top 1.5 m for woody lands, and 20-65 t/ha in the top 30 cm depth, 40-100 t/ha in the top 50 cm, and 80-100 t/ha in the top 60 cm for non-woody lands (Figure 3.8; based on data in Silver [2009]). Because of this variation in soil carbon among sites, it is not possible to directly compare the soil carbon content of different sites with native perennial grasses with others covered by non-native annual grasses. Rather, what is needed are studies that measure net carbon fluxes (NEE) or soil carbon content of sites characterized by patches of both grass types in close proximity.

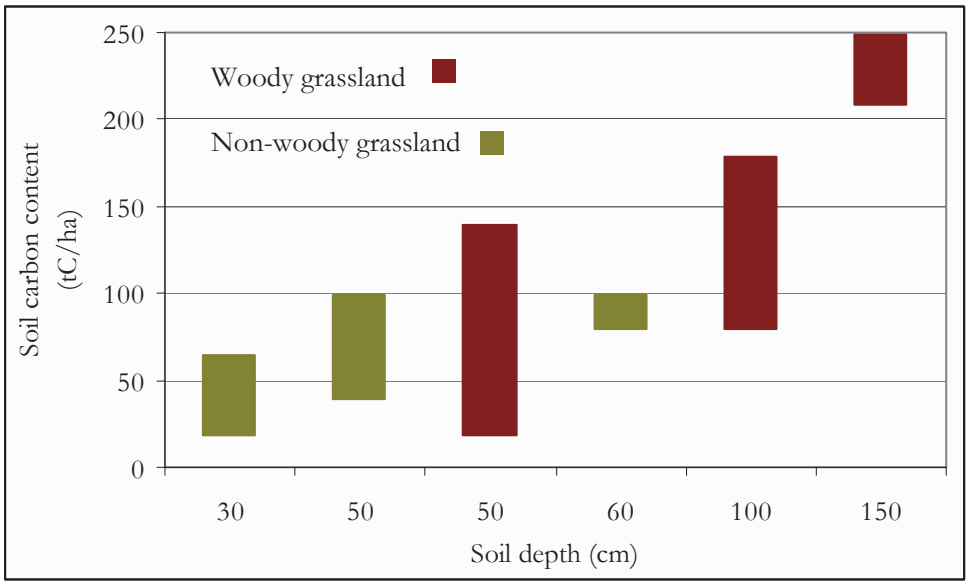

Figure 3.8: Soil carbon content of woody and non-woody California grasslands

Of the few available grasslands NEE studies, none examine NEE for patches of native perennials and exotic annuals on the same site. Although potential productivity differences between sites as well as interannual variability in climatic variables limit the validity of comparisons of the findings of those studies, it should be noted that the only study examining a native grassland (Valentini et al., 1995) reported by far the highest net carbon uptake of all grassland studies (Table 3.5).

While suitable NEE studies are not available, there are three soil carbon content studies of sites that contain patches of both native perennial and exotic annual grasses (Table 3.14). Koteen (2007) and Koteen et al. (2005) analyzed differences in total soil carbon between exotic annual and native perennial grasses at two sites in Marin County. They found that the soil carbon content in the top 50 cm of patches of native grasses was between eleven and 57

percent higher than in patches of exotic grasses. Thus, the displacement of native perennial by exotic annual grasses at those sites is estimated to cause soil carbon losses of between 7.1 to 21.2 metric tons per acre in the top 50 cm of soil, depending on species type and site. Total losses are likely to be even higher, since perennials have deeper roots than exotic annual grasses.

Conversely, restoration of native perennials can increase soil carbon stores. For example, Potthoff et al. (2005) (see also Jackson et al., 2007) examined total soil carbon of plots of old field annual grassland (fallow for 65 years) and restored perennial grassland (4 yrs old). They found that total soil carbon was little affected by plant species composition at this early stage of restoration (4 yrs). However, as the authors note, and despite the fact that tilling strongly reduces soil carbon due to carbon mineralization (the restored patches had been tilled several times before seeding), total soil carbon of the perennial sites already had recovered to levels similar to those of the untilled old-field annual site, indicating rapid carbon stock recovery. In addition, the authors note that the deeper (15-80 cm) distribution of total soil carbon and soil microbial carbon pools in perennial grasslands may increase soil carbon stocks over time based on the trend for more roots below 15 cm in the perennial compared to the annual grassland. As a result, the authors expect that restoration of perennial grasses will lead to net gains in soil carbon.

The fact that Valentini et al. (1995) (Table 3.5) report moderate net uptake for a native grassland, compared to the on-average neutral or weakly negative carbon balance of California grasslands (Baldocchi, 2009), supports the hypothesis that reestablishment of native grasses can improve the carbon balance of rangelands.

The potential carbon sequestration that could be achieved through large-scale restoration of native perennials on California rangelands can only be tentatively gauged, due to the very limited available empirical data. Koteen et al.'s (2005) and Koteen's (2007) estimates of the difference between soil carbon contents in the top 50 cm of patches of native and exotic grasses could be used as the amounts by which soil carbon stores could be increased through restoration of perennials. This would yield estimates of carbon sequestration of 21.2 tC/ac and 7.1 tC/ac, respectively. These are estimates of the total amount of soil carbon that would be restored through perennials. The annual increases in carbon stocks would be much smaller, depending on the time it takes to restore soil carbon pools to pre-invasion levels.

Due to the large expanse of grasslands found in our study area (Table 3.8), restoration of native grasses - even on a relatively modest scale - can generate substantial total quantities of net carbon uptake. For example, if five percent of grasslands in the four-county area are restored, soil carbon stocks would increase by an estimated 288 thousand to 865 thousand tons (1.06 to 3.17 million tons of CO_2e), depending on the carbon gain scenario (Table 3.15).

Table 3.14: Comparisons of total soil carbon in exotic annual and native perennial grasslands in California

Study	*Grassland type*		*Total soil C* kgC/m^2	*Total soil C* gC/kg	*Location*
Koteen et al. (2005)	Mixed exotic **annual**		9.3 *		Bolinas Lagoon Preserve, Marin County (37°56'40"N, 122°41'W)
	Native **perennial** (*Bromus carinatus* and *Elymus glaucus*)		14.5 *		
Koteen (2007)	Mixed exotic **annual**		15.3 *		Tennessee Valley, Golden Gate National Recreational Area, Marin County (37°52'N, 122°31'W)
	Native **perennial** (*Agrostis halli* and *Elymus glaucus*)		17.5 *		
	Native **perennial** (*Festuca rubra*)		17.0 *		
Potthoff et al. (2005)	Old field **annual** grassland (65 yrs fallow)			26.0 **	UC Hastings Natural History Reservation, Upper Carmel Valley, Monterey County (121°0'31"W, 36°30'12"N)
	Restored native **perennial** grassland	- near bunches		21.3 **	
	(*Nassella pulchra*, *Elymus glaucus* and *Hordeum brachyantherum californicum* (4 yrs after seeding)	- between bunches		21.8 **	

Notes: * Top 50 cm. ** Top 80 cm.

Table 3.15: Examples of potential soil carbon sequestration from restoration of native perennial grasslands in the four county study area

% Grassland acreage restored	*Low C gain (7.1 tC/ac)*	*High C gain (21.2 tC/ac)*
	1000 tC	*1000 tC*
5%	288.4	865.2
15%	865.2	2,595.6

If 15 percent of grasslands were restored, total soil carbon stocks would increase by 865 thousand to 2.60 million tons (3.17 to 9.52 million tons of CO_2e).

Carbon accumulation from restoration of native perennials could be achieved in savanna soils, too. However, the studies listed in Table 3.14 looked at carbon differential in grasslands only. In savannas and woodlands, trees are likely to increase soil carbon stores, so the incremental increase in soil carbon from restoration of perennials may be smaller than in grasslands. Nevertheless, due to the large acreage of woodlands in the study area (1.4 million acres; Gaman and Firman, 2006), the total carbon sequestration potential in savannas in our study area is very large.

Social economic and private financial value of carbon sequestration from restoration of native perennials

Restoration of native perennial grasses on rangelands can generate a variety of benefits that carry economic value. These include the provision of habitat for wildlife people value for direct (hunting, fishing, wildlife viewing) or passive (existence values) uses. In addition, the longer period of green cover provided by perennials extends the foraging season and reduces the risk and spread of catastrophic wildfires and the economic costs associated with fire control and loss of life, health and property.

Perennials restoration also is likely to increase soil carbon stocks, which in turn increases site productivity and may generate carbon offsets that can be sold on carbon markets. (For a general discussion of carbon markets, please refer to the section *Social economic and private financial value of benefits from oak planting*.) At 2008 average OTC prices of $6.9 per ton of CO_2e (Hamilton et al., 2009), the quantity of carbon sequestered through restoration of perennials on rangelands (7.1 - 21.2 tons/acre) could generate an estimated $49 to $147 per acre over the multi-decade period over which the carbon uptake would occur. These potential revenues, even without the transaction costs associated with project accreditation and verification, are substantially lower than the costs of perennials restoration. However, from a socio-economic perspective, restoration of perennials may nevertheless be a worthwhile undertaking because of the other, public on- and offsite benefits such restoration creates (reductions in fire-related costs, wildlife habitat, conservation of native habitats and species). In the absence of compensation for the provision of these benefits as a result of perennials restoration, the landowner has no economic incentive to undertake such restoration projects. This presents another example of why achieving socially desirable outcomes may require economic incentives for private land owners.

D. Grazing Management, Riparian Fencing and Restoration, and Water Development

In this section we examine *public* ecosystem service benefits that are derived from adopting resource conservation practices associated with improved grazing management. These practices were discussed in Chapter 2 and include prescribed grazing, water development, fencing of riparian areas and restoration of riparian buffers. These practices are frequently adopted by ranchers using NRCS conservation programs and recommended as a package that can improve overall range health. For example, prescribed grazing, in combination with water development to disperse livestock from riparian areas, can help control invasive species. Thus, although we do not address the ecosystem service impacts of water development *per se*, we do consider this practice as contributing to the overall ecological health of a particular rangeland landscape. The ecosystem services we address are carbon sequestration, water quality and wildlife habitat/species conservation.

Generally, improved grazing management can have beneficial impacts on biodiversity on California's rangelands. Improved grazing management mimics "natural" systems through manipulation of animal grazing patterns, typically using a rotation-based system. To date, direct scientific evidence of the benefits of many grazing practices is lacking for California grasslands (Huntsinger et al., 2007). Some environmentalists maintain that livestock grazing is incompatible with native biodiversity preservation. DiTomaso (2000), Stromberg et al. (2007), and Barry et al. (2006) have argued that *intensive* grazing is unsustainable, decreases biodiversity, and is partly responsible for the decrease in native perennial plants and an increase in non-native species. Other researchers have concluded that managed grazing can have a neutral or positive effect on grassland health. Managed grazing practices are endorsed as a tool for promoting biodiversity in native grassland remnants and for restoration projects (Menke, 1982; Edwards, 1995, 1996; Reeves and Morris, 2001; Hayes and Holl, 2003; Stromberg et al., 2007, cited in Huntsinger et al., 2007). In a review of the impacts of grazing in riparian areas, Skovlin (1984) reported that the literature on wildlife response to moderate and seasonally controlled grazing is encouraging.

The success of managed grazing to achieve biodiversity conservation varies between locations and is dependent on external factors such as climate, particularly precipitation levels. The interaction between climate and grazing in relation to native plants is likely important but has not been rigorously examined (Huntsinger et al., 2007). Although studies of California grasslands are numerous, most suffer from design flaws and results cannot always be unambiguously assigned to grazing treatments or generalized across sites (Huntsinger et al., 2007).

*Under*grazing California rangelands can be detrimental to native biodiversity. In a study by Bartolome (2007), ungrazed plots showed an increase in invasive plant species such as medusa head, barbed goat grass and yellowstar thistle. Many invasive plants may alter ecosystem structure and functional processes, including hydrologic, fire, and nutrient cycles. Structural changes in invaded plant communities typically cause reduced native species richness and diversity and changes in canopy structure (Belcher and Wilson, 1989; Parmenter and MacMahon, 1983; Rikard and Cline, 1980; Wallace et al., 1992, cited in DiTomaso et al.,

2007). Marty (2005) has shown that some grazing is essential for maintaining the hydrology and species diversity of vernal pool grasslands.

Allen-Diaz et al. (2004) found that light cattle grazing near natural springs on California rangelands had desirable impacts on ecological functions. Carroll et al. (2007) concluded that rotational grazing was successful in providing summer nesting habitat to dabbling ducks and Aleutian Goose in the Central Valley study area.

Carbon

Silver (2009) notes that an unanswered question is whether grazing management *alone* increases soil carbon storage. Grazing by itself has not been shown to have a consistent effect on soil carbon, at least in the semiarid Mediterranean climates of the western U.S. (Jackson et al., 2007). Conant et al. (2001) completed a comprehensive literature review on grasslands and found that improved grazing increased total soil carbon on average by 0.35 tC/ha/yr. Many of the studies reviewed, however, were from outside the U.S., including Australia, the United Kingdom, New Zealand, Canada and Brazil. Their findings therefore are likely not quantitatively applicable to California due to the wide range in soil types and grazing treatments contained in the data, and the fact that none of the studies were conducted in California.

Passive and active restoration of riparian areas through the installation of buffers likely has beneficial impacts on carbon sequestration. However, there are as yet no research findings that indicate the extent to which the restoration of riparian areas does increase carbon storage.

Ranchers can market carbon offsets from prescribed grazing on the Chicago Climate Exchange (CCX). The CCX grants credits for changes in soil carbon stocks on rangelands for prescribed changes in management - sustainable stocking rates, rotational grazing and seasonal use, with a standardized offset credit rate of 0.16 metric tCO_2 per acre per year in California (CCX, 2009). Currently, projects in Butte, Glenn and Tehama counties are eligible for offset generation, while those in Shasta are not (CCX, 2009).

Water Quality

Restoring vegetated riparian buffers can be an affordable and easy-to-maintain tool for rangeland managers to reduce runoff into local watersheds. Vegetated buffers not only can attenuate water temperature, but also reduce sediment, phosphorus, and nitrogen discharge to drainage water in agricultural areas (Osborne and Kovacic, 1993).

The primary relationship between soil erosion and water quality is that soil erosion increases runoff into surface waters, which in turn increases turbidity and nutrient content. Here, runoff refers to nutrients (nitrogen, phosphorous), pathogens (*E. coli, C. parvum*), and sediment/soils (Hubbard et al., 2004). A vegetative buffer strip acts as a sponge that filters and can reduce the amount of runoff (Hubbard et al., 2004). Generally, the water quality of upland rangelands is very important for an entire watershed because these areas are often the headwater tributaries for larger rivers (Lewis et al., 2002).

Fencing riparian areas prevents livestock from trampling riparian areas and causing increased erosion of sediment and direct deposits of fecal matter into streams. However, the trade-off for the rancher is that fencing will exclude cattle from forage that may be higher in nutritional value compared to other grazed areas. There is a potential for up to 6 times more forage in riparian areas, which may also be higher in crude protein concentrations (Bailey, 2005). The opportunity cost for a rancher is that although exclusion is important for water quality and rangeland sustainability in the long run, fencing riparian areas cuts cattle off from valuable forage and can decrease ranch income. However, impacts on soil stability and water quality are difficult to reverse and mitigate once they have taken effect, and riparian forage is likely to decrease anyway as erosion claims stream banks.

Several factors contribute to the rate and amount of runoff received by a water body. These factors include but are not limited to slope, soil type, rainfall intensity and duration, type of pollutant, and the total size of the drainage area (Castelle et al., 1994; Schmitt et al., 1999; Bharati et al., 2002; Bedard-Haughn et al., 2004 cited in Tate et al., 2006). Generally, the effectiveness of a vegetated buffer strip depends heavily on the site-specific micro topography (Landry and Throw, 1997, cited in McEldowney et al., 2002). Sites with larger slopes had greater concentrations of *C. parvum* runoff (Tate et al., 2000). In addressing the impacts of riparian buffers on water quality, we examine issues related to microbial contamination, nutrient pollution, erosion and sediment pollution, and water temperature.

Prevention of microbial/protozoan contamination

Vegetated riparian buffers on rangelands have been shown to filter pathogens. Pathogenic contaminants are the most common surface water impairment in California, and pose a significant public health concern (Knox et al., 2007). They are not only dangerous to human end-users, but also to downstream wildlife and livestock (Tate et al., 2004). Vegetated buffers can minimize water contamination of pathogenic materials (Tate et al., 2006; Tate et al., 2004). Studies have shown that buffers ranging in size from 0.1 to 1.8 meters can filter pathogens by 90-100 percent (Knox et al., 2007; Tate et al., 2006).

Buffer width is an important factor for maximizing filtration and uptake effectiveness. One study (Tate et al., 2006) found that a 1-meter buffer had a 95-100 percent effectiveness rate on preventing *E. coli* runoff into a waterway. Another study (Fleming et al., 2001) proposed that twenty yards is an appropriate width for vegetative buffers under otherwise healthy range conditions, and more width would be necessary for areas of large slope or heavy fertilization. A third study found that a 1-2 yard buffer reduced pathogen contamination by 90-99 percent under heavy rainfall conditions (Knox et al., 2007).

Decreased nutrient runoff

Another concern is the runoff of nutrients into rangeland surface waters. Nutrient runoff can be detrimental to both humans and aquatic species (Hubbard et al., 2004). Vegetative buffers reduce nitrogen runoff through the process of denitrification, infiltration, and plant uptake (Hill, 1996 cited in Berard-Haughn et al., 2004). According to Berard-Haughn, et al. (2004), runoff was effectively cut by the use of buffers. In an 8-meter buffer, for example, nitrogen loads decreased by 28 percent, ammonium by 34 percent, and dissolved organic nitrogen by 21 percent.

Erosion prevention

A physical component of water quality is the runoff of sediment into surface waters caused by erosion. Severe erosion has the potential to claim land that could be used for grazing. Additionally, the runoff of sediment into water bodies can fill small streams and/or increase the turbidity of the water, which is harmful to aquatic species and overall water quality (Lewis et al., 2002). Vegetated buffers can reduce stream bank erosion and increase water infiltration by providing a root structure to hold soils together (Beschta, 1997).

Water temperature improvements

Riparian vegetation can lower water temperatures of streams and rivers, which benefit aquatic species such as salmon and trout, both of which are listed as endangered species and found in the study area. However, Larson and Larson (1996) argue that although stream shading may have some value for in-stream water temperature attenuation, there are too many factors that can diminish the effectiveness of buffers. For instance, vegetation must be tall and abundant enough to cover the stream during peak direct sunlight hours. Additionally, the stream would need to be fairly narrow in order to be significantly covered. Using riparian buffers for stream temperature attenuation is either too unrealistic or will yield limited benefits (Larson and Larson, 1996). This argument is countered by Beschta (1997) who suggests that riparian vegetation is like a 'hat' that prevents light and heat from impacting the surface waters. Additionally, there is a proportional increase in stream temperature when solar radiation reaches water, so it is critical to take advantage of all the shading possible.

Wildlife habitat

Even though riparian zones only represent 1-2 percent of western forest and rangeland landscapes, they are considered hot-spots for the provision of ecosystem services such as biodiversity, wildlife habitat and water quality (Kauffman and Kruger, 1984; Kauffman et al., 2004; Gregory et al., 1991; Naiman and Decamps, 1997). It is well known that riparian vegetation plays an essential role in the provision of ecosystem services by regulating light and temperature regimes, providing nutrients and energy and maintaining biodiversity (Naiman and Decamps, 1990). Research suggests that plant community structure and composition determines the density and composition of the wildlife community (Johnston and Anthony, 2008; Nur et al., 2008). Riparian buffer zones can provide valuable refuge areas for wildlife in otherwise homogeneous agricultural landscapes (Triquet et al., 1990).

Riparian buffers offer generally undisturbed land for nest sites, den locations, and bedding areas in habitats exposed to periodic disturbance by farming machinery (Best et al., 1995). Buffers harbor a variety of foods including plant seeds, vegetative material, and arthropods. Finally, buffers can serve as travel corridors between fragmented habitats, thus facilitating gene flow among otherwise isolated wildlife communities (Dickson et al., 1995; Haas, 1997; Jobin et al., 2001). These corridors will increase wildlife's ability to adapt to climate change impacts.

In California, over 225 species of birds, mammals, reptiles and amphibians depend on riparian habitat (Knopf et al., 1988; Dobking, 1994 citing Vaghti and Greco, p. 426). California riparian ecosystems provide habitat for 83 percent of the amphibians and 40 percent of the reptiles known in that state (Brode and Bury, 1984).

Riparian habitat in our Central Valley study area provides for the needs of more California mammals than any other habitat in the state (Williams and Kilburn, 1984). Great Valley riparian habitats are also important for native fish such as winter-run Chinook (Sommer et al., 2001). Some native fish such as the delta smelt and the Sacramento splittail that are threatened by extinction (Moyle, 2002) can also benefit from fencing riparian areas.

Riparian habitats in the Central valley of California also provide important habitat for invertebrate species. The Valley elderberry longhorn beetle (elderberry beetle) is endemic to California's Central Valley (Barr, 1991), and was listed as federally threatened in 1980 (USFWS, 1980). The elderberry shrub (*Sambucus spp.*) serves as the sole host for the elderberry beetle (USFWS, 1980). Previous studies reveal that the size or maturity, density, and connectivity of elderberry shrubs strongly affect the beetle's presence (Collinge et al., 2001; Talley et al., 2007) and stress the importance of riparian areas for this species' recovery (Holyoak and Koch-Munz, 2008).

Riparian areas in rangelands are particularly vulnerable to disturbance. Uncontrolled livestock grazing can be detrimental to wildlife by altering vegetation through defoliation and trampling, reducing water quality through fecal contamination, and increased erosion (Knopf et al., 1988; Fleischner, 1994; Belsky et al., 1999). Due to the potential negative impacts of livestock grazing, fencing has been advocated as a protective measure to minimize impacts of livestock on riparian and aquatic habitats (Knopf et al., 1988; Elmore and Kauffman, 1994; Platts and Rinne, 1985). Other practices include off-site water development, herding, mineral supplement placement, and animal breeding. Riparian vegetation experiences a rapid recovery after the removal of livestock (Kondolf, 1993; Platts and Wagstaff, 1984; Popotnik and Giuliano, 2000). Willow (*Salix spp*) or cottonwood (*Populus spp*) densities and/or cover increases with livestock exclusion (Case and Kauffman, 1997; Green and Kauffman, 1995; Sarr, 1995). In turn, water quality improves as stream banks stabilize, excess nutrients are trapped by riparian vegetation, and the stream is shaded (Kauffman and Kruger, 1984; Belding et al., 2000).

The recovery of riparian vegetation after livestock exclusion often results in an increase in the abundance and diversity of wildlife populations. Research has shown that birds (Popotnik and Giuliano, 2000), fish (Platts and Wagstaff, 1984) and small mammals (Hayward et al., 1997) benefit from the exclusion of livestock from stream zones. Warren and Schwalbe (1985) found that improved vegetation structure in riparian plant communities supported larger insect fauna and greater lizard density. Some herpetofauna (northern queen snake, eastern garter snake, and tadpoles) exhibit positive responses to the improved conditions provided by stream bank fencing (Kauffman et al., 2004). Fenced areas, with increased vegetative diversity and structure, can support a more abundant and diverse reptile and amphibian community, as suggested by Busack and Bury (1974), Szaro et al. (1985), and Bock et al. (1990).

Different species respond differently to livestock fencing in riparian areas. Homyack and Giuliano (2002) reported no differences in the abundance of reptile and amphibian communities after livestock exclusion. In a similar study, Rinne (1988a) also found few differences in the macro-invertebrate community between grazed and un-grazed stream sections. Fenced riparian areas may attract predators, thereby reducing reptile and amphibian numbers. For example, great blue herons, green herons, and belted kingfishers occur more commonly in areas with stream bank fencing than in unfenced areas (Popotnik and Giuliano, 2000). More research is needed to determine the different responses of species to livestock exclusion (Homyack and Giuliano, 2002).

Differences in the responses of native fishes and their habitat to livestock fencing have been noted as well. Some studies indicate a rapid recovery of aquatic and fisheries habitat, such as decreased stream bank angles, increases in shading, water column depth, and substrate quality for salmonids (Rinne, 1988a, b; Knapp and Matthews, 1996). Numerous studies have documented greater biomass and abundance of trout in livestock ex-closures (Keller and Burnham, 1982; Knapp and Matthews, 1996), but others have shown little or no difference (Rinne 1988b; Rinne and LaFayette, 1991). In California, fencing riparian areas can improve fish habitat for species like steelhead by shading, improving large woody debris and creating narrower, deeper and more complex channel morphology (Opperman and Merlender, 2004).

Restoration of Riparian Vegetation

In general, the literature on the wildlife impacts from riparian vegetative restoration practices is scarce. Both passive and active restoration methods have resulted in the establishment of woody vegetation and improved plant community structure and composition.[18] A recent survey of restoration sites on private ranches in Marin, Mendocino, and Sonoma Counties showed that both passive restoration methods (grazing management and fencing) as well as active re-vegetation techniques (planting and/or bioengineering) are beneficial to plant communities and wildlife (Lennox et al., 2007).

Few studies have compared long-term results from active and passive re-vegetation (Thayer et al., 2005). Active restoration methods have been shown to accelerate the benefits associated with canopy cover and bank stability in the first ten years after project implementation. In general, however, the magnitude of the benefits from active and passive restoration methods converges after approximately 10 to 15 years (Lennox et al., 2007).

In a survey of restoration project managers, 59 percent cited significant improvements in wildlife habitat and populations, including more diverse fish and avian species, from riparian restoration efforts. There have also been documented sightings of threatened and endangered species in restored project reaches (Kondolf et al., 2007).

A restoration site in the Carmel River in California was successful in establishing vegetation but after two years no differences between abundance of reptilian and amphibians and

[18] Active restoration entails physical modifications to or treatments of the riparian zone (e.g., disking, application of herbicides, burning, planting), generally with the intent of re-establishing native species. Passive restoration does not involve such interventions, but simply allows already-present vegetation to grow or allows colonization of the site by plants from neighboring areas.

species richness and occurrence of mammals were detected. However, greater bird species richness was observed on restored sites in the summer (Queheillalt and Morrison, 2006). Even though vegetation may recover rapidly, sometimes restoration sites take decades to provide habitat for wildlife species (Homyack and Giuliano, 2002). Riparian re-vegetation in the Sacramento Valley has been largely successful in terms of providing habitat for a diverse community of breeding birds (Gardali et al., 2006).

E. Oak reestablishment and afforestation, restoration of native perennials and riparian restoration measures – some concluding remarks

Our review in this chapter of the ecosystem services and associated benefits provided by California rangelands and by rangeland conservation practices has yielded several general observations. First, rangelands generate a wide range of services that carry considerable total economic value. These services support benefits like livestock production, wildlife- and water-based recreation activities, drinking and irrigation water, species conservation, aesthetic benefits in the form of scenic views, and avoided damages to health, private property and public infrastructure (Table 3.1).

The second finding is that while some of the value supported by the services provided by rangelands can be captured by landowners – for example, improvements in forage quantity, quality or timing and carbon sequestration – a substantial portion of the overall benefits accrues off-site. The latter include avoided water treatment and dredging costs, avoided health costs and property damages, passive use values for threatened, endangered or rare species, and aesthetic benefits associated with scenic views.

Rangeland conservation thus provides an illustrative example of what economists refer to as incompletely specified property rights (Freeman, 2003; Randall, 1987). In such cases, landowners are unable to prevent others from enjoying the benefits their land conservation produces and thus fail to reap the full value of these benefits. They therefore do not have an incentive to take the full value of these benefits into account when making land management decisions (Kroeger and Casey, 2007). If, even in the presence of such positive externalities or third-party effects, rangeland conservation actions would be economically beneficial for landowners, divergence between private and public benefits would not be a major concern as most landowners likely would adopt these practices anyway. However, for the rangeland practices examined here – reestablishment or afforestation of blue oaks on grasslands; restoration of native perennial grasses; and riparian conservation and restoration – this is not the case. Rather, for all three practices, their private costs exceed their private benefits, in some cases by a considerable margin. Cost share programs in some cases can reverse this result, as is true for some riparian fencing or restoration measures (see Chapter 2). However, for oak reestablishment or afforestation and for restoration of native perennials, current cost share levels generally are not sufficient to make these practices economically attractive for landowners. To the extent that the public benefits from these conservation practices exceed the increases in cost share levels needed to make these practices attractive to landowners,

increased public funding for these practices would yield positive net benefits for society as a whole.[19]

[19] This does not automatically imply that such funding increases for these practices would be efficient. Given limited public conservation funds, there may be other conservation practices that yield even higher net benefits for given investments.

4. Economic Policies and Incentives to Promote Habitat Conservation on California Ranchlands

California's rangelands are a valuable natural resource that produces ecosystem services that support a wide range of benefits for ranchers and society at large. These benefits include forage production for livestock and wildlife, pollination of nearby crop lands, erosion control and water quality, outdoor recreation, scenic views, carbon sequestration, and the provision of habitat for threatened, endangered and rare species, to name but a few. As noted in the Introduction to this report, conservation of California's rangelands is essential to maintaining the State's biodiversity heritage.

Rangeland conservation practices such as prescribed grazing, restoration of riparian areas and native perennial grasses, and oak reestablishment and afforestation increase the flow of ecosystem services on and from rangelands, and thus increase the economic benefits supported by rangelands.

There are a number of rangeland conservation practices that currently generate benefits for society at large whose adoption also can be financially beneficial for ranchers. Thus, many ranchers should have an economic incentive to adopt these practices. However, as our analysis has shown, these practices often are financially attractive to ranchers only because a substantial portion of their total implementation costs are absorbed by public agencies through cost share programs. Public cost sharing thus is vital for the implementation of these conservation practices and for securing the large public benefits they generate. Nevertheless, due to a number of constraints that we will discuss below, these practices are not adopted by ranchers on as widespread a basis as one would expect based on financial grounds alone.

Our analysis also has demonstrated that, as illustrated by the examples of oak reestablishment or afforestation and restoration of native perennial grasses, that there are rangeland conservation practices that generate substantial public benefits but that currently are not profitable or at least financially neutral for ranchers. There are two reasons for this. First, these practices are relatively more costly to implement than those that currently are profitable or revenue neutral for ranchers. Second, they produce benefits that disproportionately accrue to people other than the ranchers, and for many of which there are no established markets that would allow ranchers to require compensation from beneficiaries for the services provided. As a result, these practices are not adopted on a widespread basis, even though from the perspective of the public at large there may be an economic case for adopting them because their total benefits for society exceed their total costs.

This presents a problem common to the management of natural resources that generate large non-market benefits, and where an important share of these benefits accrue to people other than the landowner. Ranchers have no incentive to adopt management practices whose adoption would be beneficial for society as a whole, because they absorb all the costs but receive few of the benefits. Thus, society at large forgoes the net benefits these natural resource management practices would produce.

As a result, adoption levels are too low both for those rangeland conservation practices that generally are financially profitable for ranchers and for those that are not but that would be

economically beneficial for society as a whole. What is needed are approaches that can overcome the constraints currently limiting the more widespread adoption of rangeland conservation practices that are desirable on economic grounds. In identifying such approaches, it is helpful to distinguish between what economists refer to as "private good" ecosystem services and those that have "public good" character.

Increasing the provision of public-good ecosystem services from rangelands

Public good ecosystem services are those services that are non-exclusive – meaning that once provided, it is infeasible to prevent anyone from enjoying them – and that are non-rival – meaning that enjoyment by one individual does not reduce their enjoyment by others. Examples of such public good ecosystem services are clean air or the conservation of threatened, endangered or rare species. Whenever an ecosystem service is a public good, landowners cannot demand compensation for its provision to others and thus do not have an incentive to provide this service at the level that would be economically efficient (that is, the level that would maximize net benefits for society). To overcome this problem, approaches are needed that would allow landowners to be compensated for the economic value of the benefits generated by the ecosystem services their lands provide. This would provide rangeland owners with an incentive to take into account these benefits in their land management decisions and thus could increase the flow of ecosystem services from these lands.

One approach frequently chosen is to have a government agency act on behalf of the public at large (who is the recipient of the benefits) and pay the landowner for the provision of the services. Examples of this are found in the form of environmental payment programs such as USDA's Conservation Stewardship Program (CSP), the Environmental Quality Incentives Program (EQIP), the Conservation Reserve Enhancement Program (CREP), the Wildlife Habitat Incentives Program (WHIP) or the Wetlands Reserve Program (WRP). However, the budgets of these programs are not sufficient to fund a truly comprehensive system of payments to private rangeland owners, especially if payment or cost share levels need to be increased for those practices that carry high costs but also produce large public benefits. As a case in point, our analysis shows that current incentives in the form of NRCS cost share programs are insufficient to make conservation practices such as oak reestablishment and afforestation or restoration of native perennial grasslands economically viable propositions for private landowners in our study area.

However, the social benefits of these practices may make them economically viable (i.e., generating positive net benefits) for society at large. Thus, it may make sense to invest more public resources to encourage adoption of these practices. To assess how public funds could be used efficiently to achieve more widespread adoption of rangeland conservation practices, what is needed is a comprehensive economic assessment – including both costs and benefits – of a larger variety of rangeland conservation measures than are examined in this study, in order to determine the net-benefit ranking of these measures and their per-acre costs. Furthermore, because of the spatial heterogeneity of both the costs and benefits of the practices, this analysis should be sufficiently detailed spatially. To yield credible results, it also should be based on sound valuation approaches, relying on ecosystem service production functions and ecosystem service demand analysis. To carry out such a valuation for all or a significant portion of the state's rangelands is a challenging task that will require a well-

planned and -coordinated research effort. One promising new ecosystem services valuation tool, InVEST (Tallis and Polasky, 2009), fulfills the requirement of incorporation of spatial heterogeneity of services and service values. This tool, which relies on outputs of existing biophysical and economic models and can accommodate different model complexities, has been applied to several case studies. Ideally, its application to California rangelands would involve a collaborative effort between ranchers, state natural resource and environmental agencies and university researchers to assure the most accurate and comprehensive assessment.

Once a net-benefit ranking of rangeland conservation practices has been carried out by comparing estimates of the ecosystem service benefits generated by rangeland conservation practices with the costs of these practices, this ranking then could be used to inform the allocation of scarce public conservation funds to practices for maximum economic benefit, by identifying, across all rangelands, the optimal conservation practices for a given location, and the locations where those practices would generate maximum economic net benefits. If desired, this allocation of conservation resources based on net benefits could be modified subject to particular conditions to fulfill goals other than the generation of maximum net benefits, such as spatial or distributional equity concerns. For example, allocation of funds could be divided among counties based on rangeland acreage, county population, ranch size, or some other variable of concern for distributional equity, and then within each county the ecosystem services net benefit ranking would be used to allocate conservation payments to ranchers. It bears noting that subjecting conservation fund allocation to any such conditions will reduce the cost-effectiveness of the public investment, so these concerns would be better addressed directly via other measures like reductions in income or property taxes.

Once the highest net benefit generating practices and properties are identified, conservation funds could be allocated cost-effectively using reverse auctions (Eigenraam et al., 2006; Greenhalgh, 2007). The NRCS would appear the logical choice for overseeing and coordinating the research effort to produce a net benefit ranking of rangeland conservation practices as well as the auctions and implementation and oversight of the contracts with local ranchers.

Increasing the provision of private-good ecosystem services from rangelands

Ecosystem services that are not intrinsically of public good character (i.e., non-rival and non-exclusive) can in principle be traded. Thus, markets can develop or be developed for these services. However, even for many of these private good-type services generated by rangelands and rangeland conservation practices, markets currently are missing. This means that, just as in the case of public-good type services, landowners cannot reap the full value of the services their lands provide, which in turn reduces their incentive to manage their lands for these services. This is due to a number of factors, including a lack of a sufficiently quantitative link between cause and effect (e.g., By how much does upstream rangeland management practice x reduce service provision and value at downstream facility y?); the difficulty for potential buyers of monitoring practice implementation by potential sellers or a lack of awareness of available mechanisms for ensuring seller compliance (such as third-party verification of compliance); and the large numbers of potential beneficiaries of some services, which increases transaction costs and may lead some potential buyers to refuse to

participate in coordinated action in the hope to benefit for free from the actions taken by others.

These problems are more difficult to overcome for some rangeland ecosystem services than for others. In some cases, information about the biophysical flow of services is available to academic experts but is not easily available to potential sellers (ranchers) or buyers. Examples of this are the impact of specific conservation practices on nutrient concentrations at particular points downstream. In other cases, ranchers may be unaware of market opportunities or lack information on potential payment levels or how to access a particular market. This appears to be the case for rangeland conservation-based carbon credits in much of California, as it is unknown if any ranchers in the state are participating in the Chicago Climate Exchange's Sustainably Managed Rangeland Soil Carbon Sequestration Offset Project Protocol (CCX, 2009). Both of these limitations could be overcome through increased outreach efforts by extension services or aggregators.

In some cases, markets for a particular service may exist and ranchers may be aware of those markets, but the prices paid for the service are too low to provide an economic incentive to adopt appropriate conservation practices. One example may be oak reestablishment and afforestation of California's Mediterranean climate rangelands. At current prices on the CCX and over-the-counter (OTC) markets, planting of oaks on rangelands does not generate sufficient income from carbon credits to make this practice financially viable for landowners. Oaks do generate additional private benefits for ranchers such as increased forage production and aesthetic attractiveness, but these, just like significant carbon sequestration, only occur once the trees have reached a certain age. Thus, carbon credits would not necessarily have to cover the full costs of oak planting, but current credit prices are too low to make up the difference.

What is needed for each of the ecosystem services generated by rangeland conservation practices is an assessment of the key constraints that currently prevent this service from being traded in markets. This would allow the identification of possible approaches for overcoming these constraints, and of those services that simply are not suited to markets due to their characteristics. The basis for this analysis is well-developed (Kroeger and Casey, 2007; Brown et al., 2007), but it has not been applied rigorously and comprehensively to rangeland conservation practices.

Markets for ecosystem services can be created through regulation. Domestic examples of such markets are the many wetland mitigation banking and water quality markets that exist in the U.S., as well as the state or regional carbon markets that already are in operation (e.g., the Regional Greenhouse Gas Initiative, or RGGI in the northeast) or entering their operative phase (California's Climate Action Registry, or CAR; Western Climate Initiative, or WCI). In these cases, regulation is the driver that creates a demand for ecosystem services, which then in turn stimulates a supply. In California, there are a few water quality markets, but they cover a small area and none of them is found in the four-county area studied in this report. However, the potential for the development of these markets exists for California rangelands, many of which are located upstream of waters classified as impaired due to sediment loading. The creation of total maximum daily load (TMDL) restrictions for sediment discharges in the respective waters, pursuant to the Clean Water Act, would be expected to stimulate exchanges between regulated point sources and ranches in which the

former pay the latter to adopt management practices that reduce sediment loading from ranchlands. However, this issue requires further investigation to assess the likelihood of setting sediment TMDLs in the area, who the participating entities would be and at what level load reductions would be set.

Many rangeland conservation practices do benefit wildlife even though wildlife conservation may not be their stated or primary objective. However, wildlife and their habitat are ecosystem services whose provision can be incentivized not only directly through government payment programs such as CSP or WHIP, or indirectly through programs targeted at other natural resources such as CREP, EQIP or WRP. Their provision also can be promoted via markets for other ecosystem services such as carbon and water quality that are joint products of wildlife habitat. Thus, if wildlife can be "bundled" with other ecosystem services whose conservation can be achieved through some form of incentive system (government payment programs or markets), it could be protected indirectly through incentives that lead to the conservation of those other services. However, wildlife habitat and other ecosystem services often are not perfect co-products. As a result, there generally are trade-offs when managing a site for more than one ecosystem service, and attempts to maximize output of one service can lead to a reduction in some or all of the other services (Chan et al., 2006). This true for managing single species as well. Carbon sequestration and wildlife habitat provide an illustrative example. A native forest sequesters carbon and provides habitat for native wildlife. Therefore, conservation of such a forest or afforestation of grasslands or agricultural lands using native tree species generates both carbon and wildlife benefits. However, if sequestration of carbon is sought to be maximized, then plantations of fast-growing species, often non-natives, may be preferred over afforestation to a native forest. Such plantations are far less suited to providing habitat for native species. To ensure wildlife conservation is achieved through incentives for other ecosystem services, those incentives need to be designed explicitly such that they are conditional upon specific management practices that benefit wildlife species of concern. In the carbon sequestration example, credit protocols could stipulate that credits could be earned only through projects that utilize native vegetation, not exotic plant species.

In regulation-driven markets (e.g., Clean Water Act TMDL-based trading areas; climate legislation-based regional or national carbon markets), this could be achieved through the regulatory agency applying appropriate credit definitions or through the setting of conditions that eligible practices need to fulfill to earn credits. In the case of California, this would be the California Air Resources Board for carbon credits (until such time when a federal carbon market supersedes state carbon regulations), or the U.S. EPA for water quality trading credits. At the federal level, the newly created Office of Ecosystem Services and Markets, housed in the U.S. Department of Agriculture, could assist with appropriate credit definitions and standards.

In voluntary markets, credit definition is up to the exchange that manages credit transactions (e.g., Chicago Climate Exchange) or individual agreements between buyers and sellers. Thus, ensuring appropriate credit definitions and verification is more difficult than in regulated markets. Nevertheless, eco-labeling initiatives and public awareness campaigns are some of the tools that are available to influence the adoption of credit definitions that fulfill wildlife conservation goals.

Policy suggestions for identifying effective and efficient incentive mechanisms for promoting increased restoration of rangelands to induce increased ecosystem services

Rangeland conservation practices provide many benefits for society at large. Many of these practices are costly to ranchers and are not viable in purely private financial terms. Hence, their adoption often is contingent upon ranchers' receiving some form of payment. This compensation can take two forms: Public payment programs or income from markets for ecosystem services.

Based on our study, we offer the following recommendations for distribution of our findings and increasing the adoption rate of rangeland conservation practices that will benefit California's native biodiversity on rangelands:

- In many cases, there exists a lack of sufficient quantification of the ecosystem service impacts of some practices (erosion from native perennial grass sites vs. exotic annual sites; habitat quality improvements from perennials restoration). Currently NRCS is carrying out an evaluation of the bio-physical impacts of rangeland conservation practices. This project, called the Conservation Effects Assessment Program (CEAP) will improve the information about the quantity of services and benefits that are generated through adoption of specific practices. This information needs to be widely communicated to ranchers and its usefulness for developing ecosystem service credits assessed.

- The California Rangeland Conservation Coalition (Rangeland Coalition) and its partner organizations should make the results of this study widely known to policy makers, ranchers, state and federal technical agents, conservation organizations, foundations and the general public for the purpose of developing and funding an institutional framework and increased incentive mechanisms for conserving California rangeland ecosystems and native biodiversity.

- There is a need to expand farm bill program funding for conservation practices and technical assistance. Such an expansion could take place through the current Conservation Stewardship Program, which is based on public payments for environmental outcomes, and could be a precursor for the development of private markets for some resources. To encourage cost-effective investment of public funds, conservation payments should take into account the actual economic value generated by the ecosystem services provided by a particular practice in a particular location. This will require the increased deployment of emerging ecosystem service value assessment models such as InVEST, coupled with results from the Conservation Effects Assessment Program. There also needs to be better coordination and targeting of farm bill conservation programs through the Cooperative Conservation Partnership Initiative in California to simplify access by landowners and provide landscape-level resource conservation. Increased availability of public (NRCS) technical assistance for rangeland ecosystem conservation is essential.

- To increase rates of adoption of rangeland conservation practices that generate large public benefits but are costly to implement (e.g., oak reestablishment or afforestation, restoration of native perennials), higher cost shares are needed to make these practices economically viable for land owners.

- The U.S. Environmental Protection Agency should aggressively pursue the development of the legislatively mandated TMDL limits for those impaired waters that still are missing such limits. The design of watershed plans should assess the potential of water quality trading to aid in achieving compliance with TMDLs. In California's Central Valley and its respective watersheds, the design of trading schemes should specifically explore the role rangeland conservation practices can play in generating water quality credits. These efforts should draw on the experiences gained in other parts of the country with the generation of water quality credits from agricultural lands. Specifically, outreach efforts to ranchers are needed to increase awareness of trading opportunities and reduce information constraints and transaction costs that may depress rancher participation. Rancher organizations, extension services and NCRS all could assist in these efforts.

- Carbon markets also would increase the financial attractiveness of several rangeland conservation measures such as oak afforestation or reestablishment or restoration of native perennial grasses. Awareness of existing market opportunities such as the Chicago Climate Exchange's Sustainably Managed Rangeland Soil Carbon Sequestration Offset Project Protocol still appears to be low in California, based on the expected low participation rates of California ranchers compared to those in other states. Increased outreach efforts by aggregators, rancher organizations and extension services are needed to overcome this deficiency. Perhaps even more importantly, market access for rangeland-based credits should be increased. For example, afforestation-based sequestration on rangelands, through oak planting or riparian restoration and soil-based sequestration from restoration of native perennial grasses, currently do not qualify for participation in the Chicago Climate Exchange's rangeland sequestration offset protocol. These are severe shortcomings that should be corrected, given that the per-acre carbon sequestration potential of these practices may far exceed that of prescribed grazing. Likewise, rangeland sequestration credits should be made eligible for all emerging regional and federal carbon markets. Whether or not credit prices would make rangeland-based sequestration efforts competitive is a question that should be left to the market to decide, not to arbitrary ex-ante decisions. It is important that verification of actual sequestration be credible. Currently, this is not always the case, with verification in many cases not even requiring site visits. To this effect, verification protocols in exiting carbon markets should be strengthened, and those of emerging and future markets should be designed to be credible from the outset.

  Higher prices of sequestration credits would increase revenues from sequestration-enhancing practices for participating ranchers and thus would provide incentives for the implementation of such practices. Since prices form in response to credit supply and demand, stricter CO_2 reduction levels would increase ranch revenue from sequestration projects.

- The California Safe Harbor legislation has been adopted. This will encourage ranch owners to conserve and manage lands for endangered species and biodiversity conservation because it removes the threat of financial penalties for violations. Outreach efforts to ranchers to provide assurances and incentives to create and manage for wildlife habitat should be expanded.

- California should promote a diverse array of incentive mechanisms to protect wildlife habitat, of which ecosystem markets are just one. For example, eco-tourism, eco-labeling initiatives can complement public investment in conservation and private ecosystem service markets. Mitigation banking and ranchland easements also should have expanded resources for increased technical assistance and monitoring and enforcement of contracts.

- There are efforts throughout the US to develop a "habitat metric" that would be the basis for defining the biological and physical characteristics of a credit suitable for trading in a voluntary biodiversity market. Under the NRCS Conservation Innovations Grant that the Rangeland Coalition has recently obtained, these efforts should be tracked for their suitability for developing a similar metric for California rangelands.

- California should identify a stable funding stream to support the continuation of the Williamson Act. This is a broad based state program that provides tax relief to landowners who forgo conversion of agricultural lands, including ranchlands. Without this tax relief, it is expected that many ranchers will sell their lands, thus leading to conversion and fragmentation.

- It is recommended that Congress adopt targeted and permanent reforms of estate tax laws that discourage the preservation of working ranches. In California, the estate tax is one of the leading causes of the break-up and loss of family-owned ranching operations.

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6. Appendices

Appendix 1

California Rangeland Resolution

The undersigned recognize the critical importance of California's privately owned rangelands, particularly that significant portion that encircles the Central Valley and includes the adjacent grasslands and oak woodlands, including the Sierra foothills and the interior coast ranges. These lands support important ecosystems and are the foundation for the ranching industry that owns them.

WHEREAS, these rangelands include a rich and varied landscape of grasslands, oak woodlands, vernal pools, riparian areas and wetlands, which support numerous imperiled species, many native plants once common in the Central Valley, and are home to the highest diversity and density of wintering raptors anywhere in North America;

WHEREAS, these rangelands are often located in California's fastest-growing counties and are at significant risk of conversion to development and other uses;

WHEREAS, these rangelands, and the species that rely on these habitats, largely persist today due to the positive and experienced grazing and other land stewardship practices of the ranchers that have owned and managed these lands and are committed to a healthy future for their working landscapes;

WHEREAS, these rangelands are a critical foundation of the economic and social fabric of California's ranching industry and rural communities, and will only continue to provide this important working landscape for California's plants, fish and wildlife if private rangelands remain in ranching;

THEREFORE, we declare that it is our goal to collaboratively work together to protect and enhance the rangeland landscape that encircles California's Central Valley and includes adjacent grasslands and oak woodlands by:

- Keeping common species common on private working landscapes;
- Working to recover imperiled species and enhancing habitat on rangelands while seeking to minimize regulations on private lands and streamline processes;
- Supporting the long-term viability of the ranching industry and its culture by providing economic, social and other incentives and by reducing burdens to proactive stewardship on private ranchlands;
- Increasing private, state and federal funding, technical expertise and other assistance to continue and expand the ranching community's beneficial land stewardship practices that benefit sensitive species and are fully compatible with normal ranching practices;
- Encouraging voluntary, collaborative and locally-led conservation that has proven to be very effective in maintaining and enhancing working landscapes;
- Educating the public about the benefits of grazing and ranching in these rangelands.

California Rangeland Resolution Signatories

Agricultural - Natural Resources Trust of Contra Costa County
Alameda County Board of Supervisors
Alameda Co. Resource Conservation District
Amador Resource Conservation District
American Farmland Trust
American Land Conservancy
Audubon California
Bureau of Land Management
Butte County Resource Conservation District
Butte Environmental Council
Calaveras Co. Resource Conservation District
California Association of Resource Conservation Districts
California Cattlemen's Association
California CattleWomen's Association
California Chapter of the International Soil and Water Conservation Society
California Deer Association
California Department of Conservation
California Dept of Fish and Game
California Dept of Food and Agriculture
California Dept of Forestry and Fire Protection
California Farm Bureau Federation
California Grazing Lands Coalition
California Invasive Plant Council
California Native Grasslands Association
California Native Plant Society
California Oak Foundation
California Open Lands
California Rangeland Trust
California Resources Agency
California Wildlife Foundation
California Wool Growers Association
Cal-Pac Section Society For Range Management
Central Coast Rangeland Coalition
Central Sierra Region of Resource Conservation Districts
Central Valley Bird Club
Central Valley Land Trust Council
Chimineas Ranch Foundation
City of Livermore
Coarsegold Resource Conservation District
Committee for Green Foothills
Community Alliance with Family Farmers
Contra Costa Resource Conservation District
Contra Costa County Board of Supervisors
Defenders of Wildlife
Ducks Unlimited
East Bay Regional Park District
El Dorado Resource Conservation District
Endangered Species Coalition
Environmental Defense
Friends of Swainson's Hawk
Glenn County Resource Conservation District
Hollister Ranch
Institute for Ecological Health
Jumping Frog Research Institute
Land Conservancy of San Luis Obispo
Land Trust for Santa Barbara County
Marin Agricultural Land Trust
Marin County Resource Conservation District
Mariposa Co. Resource Conservation District
Middle Mountain Foundation
Napa County Board of Supervisors
National Wild Turkey Federation
National Cattlemen's Beef Association
Natural Resources Conservation Service
Nevada County Board of Supervisors
Nevada Co. Resource Conservation District
Nevada County Land Trust
Northern California Regional Land Trust
Placer County Resource Conservation District
Placer Land Trust
Rocky Mountain Elk Foundation
Sacramento River Watershed Program
Sacramento Valley Conservancy
Santa Barbara County Farm Bureau
San Joaquin Et Al
San Joaquin Raptor/Wildlife Rescue Center
San Joaquin Valley Conservancy
San Luis Obispo County Board of Supervisors
Sierra Foothills Audubon Society
Sierra Nevada Conservancy
Sonoma County Board of Supervisors
State Water Resources Control Board
Sustainable Conservation
Sutter County Resource Conservation District
Tehama County Resource Conservation District
The Nature Conservancy
Trust for Public Land
Tuleyome
Tuolumne Co. Resource Conservation District
University of California
UC California Rangeland Research and Information Center
Upper Salinas-Las Tablas Resource Conservation District
US Fish and Wildlife Service
US Forest Service
VernalPools.org
Western Shasta Resource Conservation District
Wildlife Conservation Board
WildPlaces
Yolo County Board of Supervisors
Xerces Society for Invertebrate Conservation

July 8, 2009

Appendix 2

Table A2.1: Model input and output for Grazing Economic Analysis Scenario I: Fencing, Riparian fencing and water development

Grazing Economic Analysis: Scenario I: Riparian Fencing and Water Development

SITE AND LOCATION CHARACTERISTICS		Year	Percent Change in Carrying Capacity: Before Treatment	After Treatment	Other Costs/Ac
Cooperator/Ranch Name:		1	0.0%	0.0%	$0.39
County and State:	CA Butte/Tehama/Yolo/Shasta	2	0.0%	10.0%	$0.39
Soil/Site Description:		3	0.0%	20.0%	$0.39
Present Condition (good/fair/poor):	Good	4	0.0%	20.0%	$0.39
Apparent Trend (up, down):	Stable	5	0.0%	20.0%	$0.39
		6	0.0%	20.0%	$0.39
EVALUATION CRITERIA		7	0.0%	20.0%	$0.39
Grazing Land Treatment:	Rip Fence and Water dev	8	0.0%	20.0%	$0.39
Initial Treatment Cost ($/Acre) 50% cost share	$23.60	9	0.0%	20.0%	$0.39
Life of Treatment (Years):	20	10	0.0%	20.0%	$0.39
Interest Rate (%):	7%	11	0.0%	20.0%	$0.39
Value of one AUM ($):	$15.00	12	0.0%	20.0%	$0.39
Total Acres Treated:	900	13	0.0%	20.0%	$0.39
		14	0.0%	20.0%	$0.39
FORAGE UTILIZATION		15	0.0%	20.0%	$0.39
Current Forage Availability (Lbs/Ac)	2,000	16	0.0%	20.0%	$0.39
Pounds/AUM:	790	17	0.0%	20.0%	$0.39
Harvest Efficiency Without Treatment:	20%	18	0.0%	20.0%	$0.39
Harvest Efficiency With Treatment:	25%	19	0.0%	20.0%	$0.39
Current Forage Availability (AUMs/Ac):	0.51	20	0.0%	20.0%	$0.39
Maximum Carrying Capacity (AUMs/Ac):	1.75				
Months Grazed/Year	6				

RESULTS OF ANALYSIS			
Years to Break-Even on Investment:		0	
Increase in Carrying Capacity (AUMs/Ac):		0.24	
Increase in Stocking Rate, (#Head/Total Ac):		31	
Total Installed Treatment Cost ($/Total Ac):		$21,236	under 50% cost share
Amortized Installed Treatment Cost ($/Ac/yr):		$2.23	
Internal Rate of Return:		99%	
Breakeven $/AUM		$9.14	Hal Gordon
Benefit/Cost Ratio:		1.36	USDA - NRCS
Net Present Value ($/Ac) under 50% cost share	:	$9.91	Portland, Oregon

Table A2.2: Model input and output for Grazing Economic Analysis Scenario II: Fencing, active restoration of riparian zones and water development

Grazing Economic Analysis: Scenario II Fencing, Active Restoration of Riparian Zones and Water Development

SITE AND LOCATION CHARACTERISTICS	
Cooperator/Ranch Name:	
County and State:	CA Butte/Tehama/Yolo/Shasta
Soil/Site Description:	
Present Condition (good/fair/poor):	Good
Apparent Trend (up, down):	Stable
EVALUATION CRITERIA	
Grazing Land Treatment:	Rip Fence+Planting+Water dev
Initial Treatment Cost ($/Acre) 50% cost share:	$26.84
Life of Treatment (Years):	20
Interest Rate (%):	7%
Value of one AUM ($):	$15.00
Total Acres Treated:	900
FORAGE UTILIZATION	
Current Forage Availability (Lbs/Ac)	2,000
Pounds/AUM:	790
Harvest Efficiency Without Treatment:	20%
Harvest Efficiency With Treatment:	25%
Current Forage Availability (AUMs/Ac):	0.51
Maximum Carrying Capacity (AUMs/Ac):	1.75
Months Grazed/Year	6

	Percent Change in Carrying Capacity		
Year	Before Treatment	After Treatment	Other Costs/Ac
1	0.0%	0.0%	$0.39
2	0.0%	10.0%	$0.39
3	0.0%	20.0%	$0.39
4	0.0%	20.0%	$0.39
5	0.0%	20.0%	$0.39
6	0.0%	20.0%	$0.39
7	0.0%	20.0%	$0.39
8	0.0%	20.0%	$0.39
9	0.0%	20.0%	$0.39
10	0.0%	20.0%	$0.39
11	0.0%	20.0%	$0.39
12	0.0%	20.0%	$0.39
13	0.0%	20.0%	$0.39
14	0.0%	20.0%	$0.39
15	0.0%	20.0%	$0.39
16	0.0%	20.0%	$0.39
17	0.0%	20.0%	$0.39
18	0.0%	20.0%	$0.39
19	0.0%	20.0%	$0.39
20	0.0%	20.0%	$0.39

RESULTS OF ANALYSIS	
Years to Break-Even on Investment:	0
Increase in Carrying Capacity (AUMs/Ac):	0.24
Increase in Stocking Rate, (#Head/Total Ac):	31
Total Installed Treatment Cost ($/Total Ac):	$24,153
Amortized Installed Treatment Cost ($/Ac/yr):	$2.53
Internal Rate of Return:	53%
Breakeven $/AUM	$10.40
Benefit/Cost Ratio:	1.22
Net Present Value ($/Ac) under 50% cost share:	$6.67

assuming 50% cost share

Hal Gordon
USDA - NRCS
Portland, Oregon

Table A2.3: Model input and output for Grazing Economic Analysis Scenario III: Water Development and Prescribed Grazing

Grazing Economic Analysis: Scenario III Water Development and Prescribed Grazing Management

SITE AND LOCATION CHARACTERISTICS	
Cooperator/Ranch Name:	
County and State:	CA Butte/Tehama/Yolo/Shasta
Soil/Site Description:	
Present Condition (good/fair/poor):	Good
Apparent Trend (up, down):	Stable
EVALUATION CRITERIA	
Grazing Land Treatment:	Water develp. & Prescribed grazing
Initial Treatment Cost ($/Acre) 50% cost share:	$17.33
Life of Treatment (Years):	20
Interest Rate (%):	7%
Value of one AUM ($):	$15.00
Total Acres Treated:	900
FORAGE UTILIZATION	
Current Forage Availability (Lbs/Ac)	2,000
Pounds/AUM:	790
Harvest Efficiency Without Treatment:	20%
Harvest Efficiency With Treatment:	25%
Current Forage Availability (AUMs/Ac):	0.51
Maximum Carrying Capacity (AUMs/Ac):	1.75
Months Grazed/Year	6

	Percent Change in Carrying Capacity		
Year	Before Treatment	After Treatment	Other Costs/Ac
1	0.0%	0.0%	$0.52
2	0.0%	5.0%	$0.52
3	0.0%	15.0%	$0.52
4	0.0%	20.0%	$0.52
5	0.0%	20.0%	$0.52
6	0.0%	20.0%	$0.52
7	0.0%	20.0%	$0.52
8	0.0%	20.0%	$0.52
9	0.0%	20.0%	$0.52
10	0.0%	20.0%	$0.52
11	0.0%	20.0%	$0.52
12	0.0%	20.0%	$0.52
13	0.0%	20.0%	$0.52
14	0.0%	20.0%	$0.52
15	0.0%	20.0%	$0.52
16	0.0%	20.0%	$0.52
17	0.0%	20.0%	$0.52
18	0.0%	20.0%	$0.52
19	0.0%	20.0%	$0.52
20	0.0%	20.0%	$0.52

RESULTS OF ANALYSIS	
Years to Break-Even on Investment:	0
Increase in Carrying Capacity (AUMs/Ac):	0.24
Increase in Stocking Rate, (#Head/Total Ac):	30
Total Installed Treatment Cost ($/Total Ac):	$15,600
Amortized Installed Treatment Cost ($/Ac/yr):	$1.64
Internal Rate of Return:	216%
Breakeven $/AUM	$6.80
Benefit/Cost Ratio:	1.61
Net Present Value ($/Ac) under 50% cost share:	$13.95

under 50% cost share

Hal Gordon
USDA - NRCS
Portland, Oregon