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The Effects of Spatial and Temporal Scale on Conservation Planning and Ecological Networks in the Central Valley, California

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The Effects of Spatial and Temporal Scale on Conservation Planning and Ecological Networks in the Central Valley, California

By

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ABSTRACT

Conservation planning and priority setting requires selection of both a planning area boundary and temporal baseline, or reference condition. This dissertation uses the Central Valley ecoregion of California as the context within which to examine the results of conservation planning decisions that are made using different spatial and temporal baselines.

The first chapter investigates the effects of ecoregional boundary choice on gap analysis, a tool used to identify shortfalls in protection of biological resources. A gap analysis for California was conducted using five statewide sets of ecoregional boundaries to identify and compare existing conservation shortfalls in major land cover type representation per ecoregion. Another gap analysis was run for the Central Valley using two temporal baselines. We found that the boundaries of different ecoregional schemes affected both the total area needed to meet the per-ecoregion land cover conservation goals and the spatial location of underprotected land cover types. Choice of temporal baseline also had a significant effect on the establishment of conservation targets.

The second chapter compares potential ecological networks identified at both the regional and local scales in order to illustrate the impact of spatial scale effects on conservation planning. We identified a potential regional conservation network for the Central Valley from which we extracted those portions found within five individual counties. We then conducted the same analysis for each of those individual counties. An overlay of the results from the two scales shows large differences in the identified networks. The results suggest that planning efforts at any one scale neglect to include biodiversity patterns and ecological processes that are important at other scales.

The final chapter integrates future land use scenarios with contemporary conservation planning in a subregion of the Central Valley in order to identify future patterns of vulnerability and degradation of an ecological network. We assessed current and projected future impacts from modeled urban growth. The modeled urban growth forecasts were overlaid on the ecological network to identify expected impacts. A threat index was calculated for individual network components and for component clusters, and revealed significant impact differences between the various urban growth scenarios.

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CHAPTER 1

INTRODUCTION: THE EFFECTS OF SPATIAL AND TEMPORAL SCALE ON CONSERVATION PLANNING AND ECOLOGICAL NETWORKS IN THE CENTRAL VALLEY, CALIFORNIA

Successful conservation planning encompasses a multitude of scales, both spatial and temporal. Any given place is located within numerous land units, each embedded within the next when one moves between spatial scales. Further, the ecological patterns of each location are context-dependent and are influenced by past land use history and will be influenced by future human actions. Therefore it is imperative to look beyond the spatial boundaries of any given planning unit and in both the past and future directions rather than merely the present when conservation decisions are being made.

Over the past several decades, we have come to the understanding that conservation planning should take place within an ecological rather than political framework (Bailey 1996). Ecological patterns do not follow political boundaries (unless those political boundaries themselves are based on physical geography or if humans have changed the ecological patterns based on the political boundaries). This understanding has led to numerous efforts to classify the land into coherent ecological units, or ecoregions, within which are found repeating patterns of physical attributes, such as climate, landform, or vegetation (Omernik 1987, Welsh 1994, Bailey 1998, Hargrove & Hoffman 2005). Management actions appropriate to these geographical entities can then be prescribed in order to guide human activity in a more sustainable direction than could otherwise be achieved (Thayer 2003).

A fundamental question arises at this point however: what is the physical boundary of an ecoregion? The planet's surface is composed of environmental gradients but land classification for management purposes implies establishment of binary boundaries (Bailey 1996). Each ecoregional classification scheme answers the boundary question in different ways with differing resulting sets of ecological units and boundaries.

While these schema might be seen as intellectual exercises framed to increase scientists' understanding of physical and biological processes, when they are applied to tangible management practices they can have substantial effects on conservation actions. Chapter 2 addresses this issue in California, a state with complex geographic patterns that have been subjected to a number of ecoregional classification efforts. As various government agencies and conservation organizations attempt to incorporate ecoregional boundaries into the planning process, the choice of classification scheme is hypothesized to have a direct effect on potential conservation targets.

Incorporation of ecoregional boundaries in management plans of various sorts is far from ubiquitous however. In the United States, most land use authority is vested in local jurisdictions, such as counties and municipalities (Theobald et al. 2005). These legal entities are generally substantially smaller in area than are their ecoregional counterparts. Further, many of these entities are not merely subunits within ecoregions but rather spill across ecoregional boundaries. They could thus have several fundamentally different ecological patterns to consider when planning conservation activities.

Chapter 3 investigates the effects of this shift in spatial scale on conservation planning results. The specific ecoregion in question, the Central Valley of California, is comprised of portions of 29 counties. Assuming that most land use authority adheres to these counties, a comprehensive ecoregional conservation plan would in essence be the amalgamation of 29 separate county-based land use plans. This chapter looks at potential differences between a systematic regional conservation plan and its overlap with local conservation plans derived using the same methods.

Effective conservation planning not only involves questions of space but of time as well. Not only do ecosystems change over time, but this change has often been accelerated by human land use patterns. Working landscapes, such as the Central Valley with its large amount of agricultural land, have generally seen much land cover change over time. These land use histories have been major drivers of the current ecological patterns and processes in these landscapes (Foster et al. 2003). For example, over the past 150 years European influx and resulting activity has converted most of the Central Valley to "non-natural" land cover types with remaining natural vegetation found in highly fragmented patterns (Ricketts et al. 1999). If conservation planning efforts ignore this geographic history then there is the possibility that potential ecological patterns and processes that have been lost in this landscape will remain permanently absent, i.e. too little restoration will take place to ensure future ecological viability. Further, the time lag associated with "extinction debt" (Tilman et al. 1994) could be placing current habitat fragments and their resident species at risk even if those remnants are placed under conservation management.

In addition to past land use history, potential future human land use could play a strong role in determining the effectiveness of current conservation planning efforts. Working landscapes such as the Central Valley are generally the location of new urbanization pressures. With human population expected to dramatically increase in the Central Valley in the coming decades (PPIC 2006), the location of new developments will potentially contribute to the vulnerability of components of ecological networks in the region.

The latter section of Chapter 2 focuses on the first of these temporal considerations. Here we investigate the effects of choice of temporal baseline on the setting of conservation target levels. Target levels based on pre-1900 land cover patterns in the Central Valley and their resulting determination of needed conservation actions are compared to those based on current conditions.

The second temporal component is addressed in Chapter 4. This chapter posits a methodology for assessing the vulnerability of a conservation network to land use change (i.e. urbanization) under a number of possible urban growth scenarios in the San Joaquin Valley, the southern portion of the Central Valley. This methodology can be used to help determine appropriate management regimes for conservation network components and prioritize them based on expected future conditions.

The structure of this dissertation is designed to elucidate aspects of both the spatial and temporal components of conservation planning processes. The chapters are sequentially ordered to move from larger to smaller spatial scale while remaining centered on the region as a unit of analysis (Figure 1.1). In the three chapters we move from the state of California as a whole to the Central Valley ecoregion to a subregion within the Central Valley. The first two chapters also include transitions to the scale addressed in the subsequent chapters, with the Central Valley being a particular unit of analysis in Chapter 2 and Central Valley subregions (i.e. counties) being addressed in Chapter 3.

The temporal aspect also is structured sequentially (Figure 1.1). Chapter 2 includes an analysis of the setting of conservation targets based on reference to past conditions. Chapter 3 is concerned with present conditions (however these too are

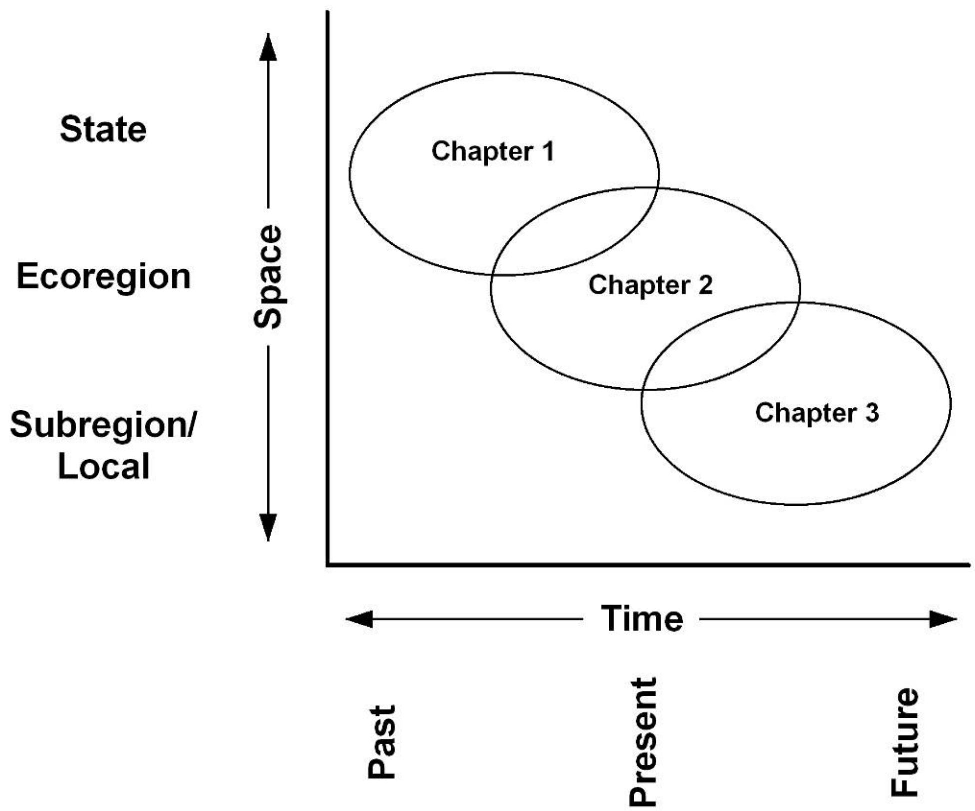


Figure 1.1. Schematic showing the spatial and temporal frameworks of the three dissertation chapters. The spatial scale constricts as the chapters progress while the time component of the chapters moves from the past to the future.

predicated on past human activities). Finally the future condition of the San Joaquin Valley is addressed in Chapter 4 with the urban growth scenario modeling as a central theme.

One caveat should be noted here however. While the spatial and temporal patterns centered in the individual chapters move from larger to smaller and past to future respectively, the sequence of data analysis and drafting of the chapters did not follow this same pattern. Specifically the first chapter written chronologically was in fact Chapter 4. In this chapter, a more basic methodology was used to identify the ecological network that resulted from the analysis than was used while compiling Chapter 3. Thus there are differences in both identified core reserves and the linkages that connect them. Ideally the more robust methods described in Chapter 3 would have been used in Chapter 4, but timing precluded this from being the case.

Results from the analyses presented in the following three chapters include findings of relevance to the field of conservation planning. First, boundaries make a significant difference. The choice of conservation planning area and the borders of that area can play an important role in the setting of conservation target levels. Next, the spatial scale at which planning areas are determined also makes a difference. Ecological networks identified at a regional scale are not the same networks identified in the same locations at the county-scale. If regional conservation goals are to be achieved, it will likely take more than merely summing the conservation achievements attained via local government-driven planning efforts. This implies that additional legal structures may be needed for effective regional conservation planning.

Finally, temporal considerations should be a part of regional conservation planning efforts. Using historic baselines in conservation planning in working landscapes rather than contemporary baselines can lead to substantial differences in not only levels of conservation effort that are needed to meet regional goals but the kinds of management actions called for. It might not be enough to confer protected status on all remaining natural areas to ensure future functioning of ecosystems but instead large restoration efforts might be required. Further, looking to potential futures can lead to a better understanding of the vulnerabilities of an ecological network to human-driven land use change. Modeling these expected future conditions can also help guide management strategies for specific network components in order to address the combination of present and future land use patterns.

As numerous authors have noted, effective conservation planning requires an understanding and incorporation of regional ecological realities into the politics of the planning process (Sale 1991, Woodward 2000, Thayer 2003). Biological patterns and processes occur at multiple spatial scales but in circumscribed places. This dissertation project aims to advance our knowledge of how those bounding decisions might impact the results of conservation planning and design processes. Past and future human land use patterns and their effects on conservation planning are also important elements that should be considered in planning efforts. While there is no possibility of return to a "pre-human" pristine landscape (Mann 2005), successful conservation planning can lead to systems with less conflict between the human and non-human realms. It is hoped that this dissertation will help provide a means for placing conservation planning more fully in a meaningful spatial and temporal context that can help achieve that goal.

CHAPTER 2

BOUNDARIES MAKE A DIFFERENCE: THE EFFECTS OF SPATIAL AND TEMPORAL PARAMETERS ON CONSERVATION PLANNING

INTRODUCTION

Regional conservation planning by definition takes place in discrete locations on the landscape. Conservation plan boundaries can be natural (e.g. watersheds) or administrative (e.g. national boundaries) but in either case are used to demarcate the study area. Once the planning area has been defined, numerous approaches are available to determine important locations of ecological features that are to be the targets of conservation efforts. Some of these approaches include representation (Margules & Pressey 2000), focal species (Lambeck 1997), and connectivity (Bennett 2003). All of these approaches seek to conserve the locations and features that contribute most to that defined region's ecological integrity.

While administrative boundaries are easier to integrate within a regulatory framework, ecologically it is generally seen to be preferable to use natural boundaries where possible in order to more effectively address natural processes and patterns that are constrained by these boundaries (Groves 2003). As a result, many current conservation planning processes use ecoregions as their spatial template. The ecoregion is a unit of land classification popularized by Bailey (1996, 1998, 2002) and others (e.g. Omernik 1987, Hargrove & Hoffman 2005). It is a large area that is characterized by such patterns as similar climate, landforms, or vegetation. However, because these patterns are often not strictly co-incidental (Wright et al. 1998) as well as the continuous nature of landscapes, operational definitions for the ecoregion vary widely, especially in locations displaying a complex mosaic of physical geographic characteristics.

One of the popular approaches to conservation planning is gap analysis (Scott et al. 1993). Gap analysis is a planning tool that is designed to test whether the established regional conservation targets are being met by the existing reserve network. This technique overlays reserve boundaries on spatial datasets representing the ecological components in order to assess the amount of each falling within the reserve boundaries. If there is a shortfall, the analysis will output how much more of a given component must be conserved to achieve the conservation target level. This globally utilized technique (Gleason et al. 2003, Maiorano et al. 2006, Trisurat 2007, Tognelli et al. 2008) assesses the representation, or portion, of all of the regional ecological components (such as vegetation types or species) that fall into various land management classes, minimally reserve networks and other lands. Conservation actions to attain target extents of habitats, as informed by a gap analysis of the amount missing in protected lands for each habitat type ensures that no ecological pieces are lost and that potential biotic integrity can be maintained into the future. A certain minimal threshold of each component type is generally considered necessary to be protected in order to maintain ecological function and viability (Svancara et al. 2005). Thus conservation targets (e.g. percentage or numbers) must be set for each component within the planning region.

These conservation targets have ranged widely since they first appeared in the scientific literature in the early 1970s. Some of the earliest identified conservation target levels were 40% (Odum 1970) and 50% (Odum and Odum 1972) of the respective study areas. A common current target level of 10% has been used to evaluate conservation needs in many studies (Wright et al. 1994, Scott et al. 2001, Andelman and Willig 2003). This figure was suggested as being the minimum necessary to preserve tropical forests

(Myers 1979) and was subsequently included as a policy-driven target several years later (Miller 1984). While policy needs drive many of the efforts that use this 10% figure, Svancara et al. (2005) show that science-based targets are generally greater, averaging roughly 30% of a given ecosystem type, although these figures vary in response to ecological context.

However, because selected reserve boundaries and ecological component occurrences are defined by the study area boundary, the results of the gap analysis could potentially change dramatically given the location of the demarcated boundary. Such a change was documented by Diamond et al. (2005), whose gap analysis examined conservation needs at different nested spatial scales. However, the effect of altering subdivision boundaries within a given planning area on the resulting identification of conservation needs remains untested.

In addition to these spatial planning parameters, temporal baselines for conservation assessments can potentially exert a heavy influence over the results (Jennings 2000). While some of the world's ecoregions are relatively intact, many have experienced substantial conversion to human-dominated land cover types (Ricketts et al. 1999). The analysis of conservation needs and levels of protection for these altered landscapes may well lead to varying results depending on which temporal parameter is selected.

In this study, we examined the effect that planning boundary choice has on identification of conservation needs in terms of ecological representation. We conducted gap analyses on an ecoregional basis, using five different sets of ecoregional boundaries to determine how study area boundaries affect assessment of conservation shortfalls across the state of California. Shortfalls were assessed through the use of representational thresholds for each vegetation type found in each ecoregion of each of the five schemes and results totaled by ecoregion and then summed for the entire state. Finally we compared both the quantitative and spatial results from each of the five schemes to illustrate the potential impact that planning area boundary selection can have in the analysis of conservation needs.

The effects of temporal baseline were analyzed for one heavily human-converted ecoregion within California, the Central Valley. We conducted a gap analysis on the major vegetation components of the Valley floor using current land cover and conservation areas data. Then we conducted the same analysis using historic land cover data coupled with the contemporary protected lands data. The resulting conservation needs were compared between these resulting assessments.

BACKGROUND

California Ecoregions

The state of California consists of a tremendous variety of physical geographies. Five major climate types - Mediterranean, highland, steppe, desert, and cool interior (California Resources Agency 2003) – interact with topographic and latitudinal gradients to create a complex mosaic of ecological and ecosystem patterns. This complexity is illustrated by the many attempts to categorize the California landscape into constituent ecoregions for both research and management purposes as illustrated below. Many of the resultant schemes have widely divergent boundary locations or even disparate regions altogether. This paper analyzes five classification schemes (Figure 2.1).

- **Interagency Natural Areas Coordinating Committee (INACC)** – Here, 10 bioregions are delineated (tied for the least of the five schemes). They are based on the physiographic features of the state, although some are modified to include land management boundaries (INACC 1992). The distinctive feature of this scheme is the splitting of California's Central Valley into smaller independent regions: the Sacramento Valley, San Joaquin Valley, and Bay Area/Delta.

- **Jepson** – This scheme also consists of 10 ecoregions (Hickman 1993). While physical characteristics are also considered, the purpose of the regional classifications is meant to aid in the prediction of ranges of plant species and communities in California.

- **The Nature Conservancy (TNC)** – The 12 ecoregions in the TNC scheme were delineated using a variety of physical characteristics including climate, landforms, vegetation, and ecological processes. A U.S. Forest Service ecoregional assessment was used as the base for the scheme and then updated to reflect those processes that would most influence regional conservation planning, a focus of this organization (TNC 2000).

- **U.S. Forest Service (USFS)** – These 18 "sections" (USFS terminology) have the most ecoregional sub-divisions of the five analysis schemes. The boundaries identified were meant to capture the differences in moisture, energy, and nutrient gradients (driven largely by climate and elevation) which in turn influence the composition of ecological communities (McNab 1996). Distinctive identified units include the Sierra Nevada Foothills, Northern Inner Coast Range, and narrow coastal sections.

- **World Wildlife Fund (WWF)** – This scheme is a modified version of the ecoregion boundaries as identified by Omernik (1995). The 12 ecoregions identified here are based not only on environmental features but species and communities as well (Olson et al. 2001). This is the only scheme of the five that explicitly considers animal species in delineation efforts. Two ecoregions unique to this scheme include the Inner and Montane Chaparral and Woodlands.

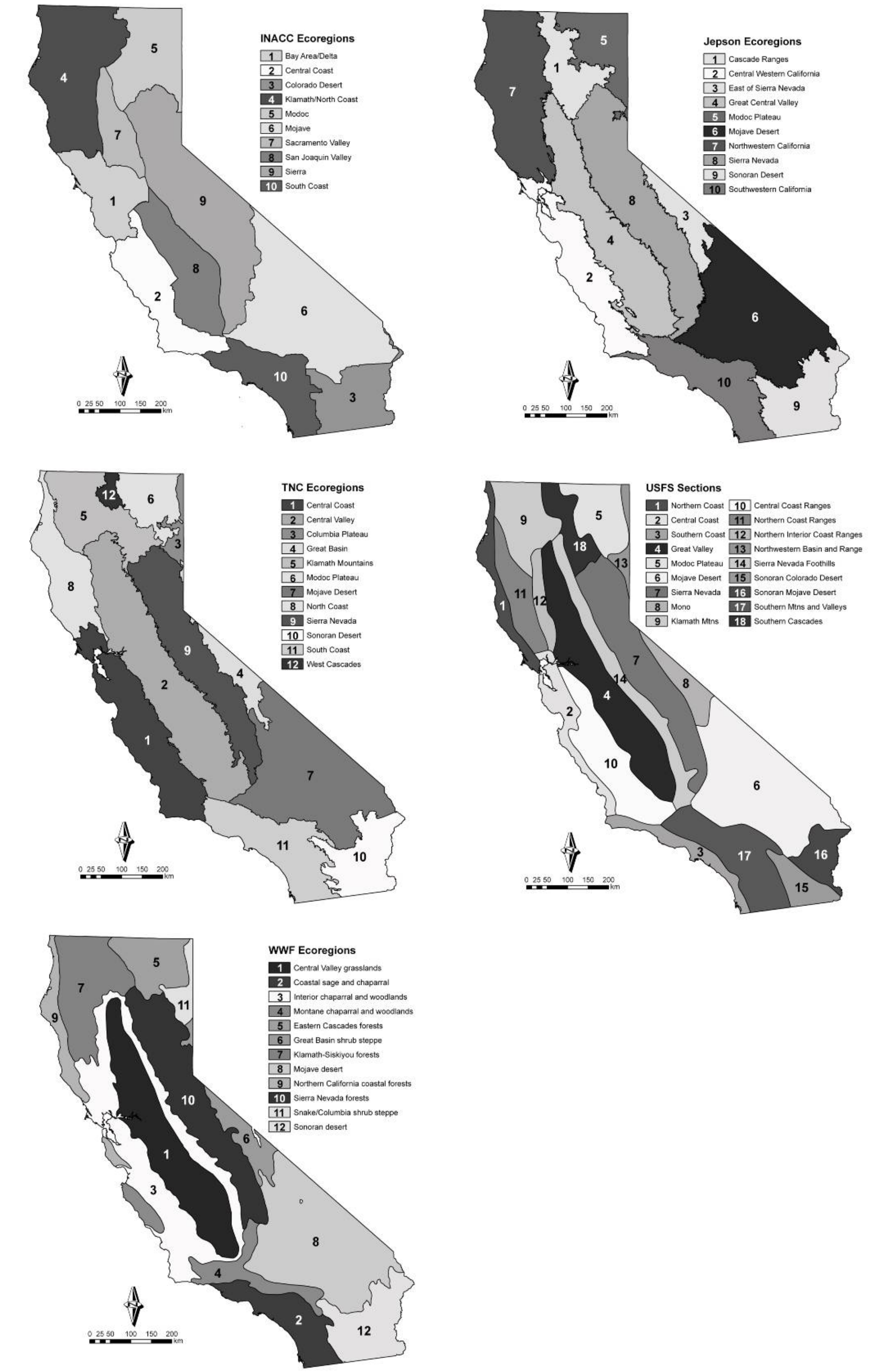


Figure 2.1. Five ecoregional schemes for the state of California.

METHODS

Data Sources

Ecoregion boundaries (in ArcGIS shapefile format) for the INACC, Jepson, and USFS schemes were downloaded from publicly available websites (INACC: http://frap.fire.ca.gov/data/frapgisdata/select.asp; Jepson: http://www.biogeog.ucsb.edu/projects/gap/gap_data_state.html; USFS: http://www.fs.fed.us/rm/analytics/publications/eco_download.html) for use in a geographical information system (GIS). The TNC and WWF shapefiles were obtained directly from the organizations for use in this project.

The land cover dataset used was the California Gap Analysis (CA Gap) Land-Cover for California ArcINFO coverage (Davis et al 1998). This dataset was converted to a shapefile. The CA Gap polygons are assigned a dominant vegetation community type based on the California Wildlife Habitat Relationships classification (WHR). There are 60 WHR types defined for California, 44 of which were used as natural communities for the purposes of this analysis.

The conservation areas dataset used was the Public, Conservation and Trust Lands (PCTL) ArcGIS shapefile (Figure 2.2) (California Resources Agency 2005). This dataset includes all publicly owned land as well as private lands that are managed for conservation purposes. However, private lands that are held under conservation easement rather than fee title by a private conservation organization are not included in this dataset. Because we did not have information on the specific management plans for each public lands unit, we included all public lands as potential conservation.

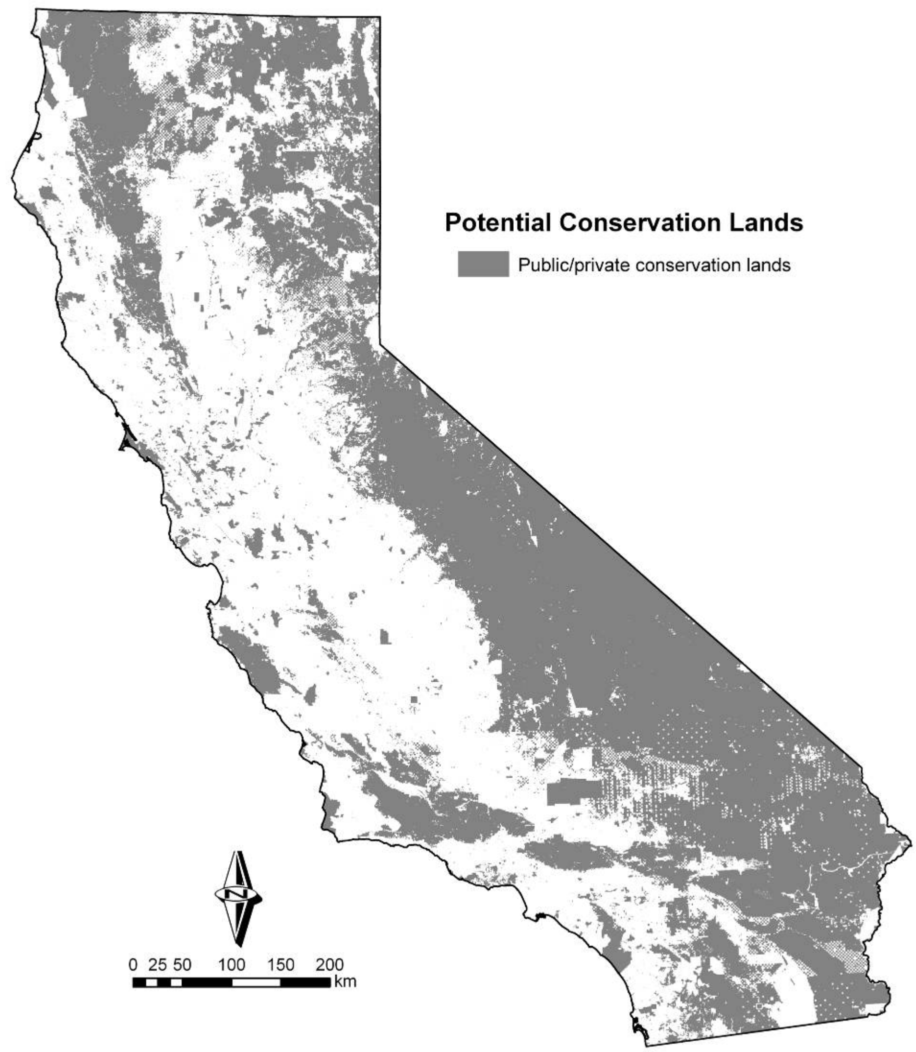


Figure 2.2. Public and conservation trust lands (PCTL) of California.

The Central Valley historic land cover dataset was produced by the Geographical Information Center (GIC 2003). This effort combined a number of analyses of historic vegetation in the Central Valley to produce one synthetic dataset.

All analyses were conducted using ArcGIS version 9.2 (ESRI 2005).

Data Analysis

To determine and demonstrate the amount of congruence between the ecoregional schemes, we overlaid the five ecoregional datasets and examined their rate of overlap. For each ecoregion in the Jepson classification scheme, we identified the ecoregion in each of the other four schemes that overlapped the greatest amount of area and calculated the proportion of the Jepson ecoregion that the ecoregion in the other scheme covered. Means were calculated both for the amount of overlap for each of the Jepson ecoregions as well as across ecoregion within each of the other four schemes. The Jepson scheme was chosen for this analysis because it is a commonly used scheme in California floristic ecology and conservation analysis.

For the gap analysis, we overlaid the boundaries from each of the five ecoregional schemes on the land cover dataset in order to identify the amount of each WHR vegetation type occurring in the ecoregions of each ecoregional classification of the state. The total area of each WHR vegetation type was calculated for each ecoregion in each scheme.

Next, the conservation areas dataset was overlaid on both the land cover and the ecoregion datasets. The amount of each vegetation type falling within the conservation area boundaries of each ecoregion was calculated. A comparison with the total extent of each vegetation type in the respective ecoregion enabled the calculation of the percentage of each vegetation type that is currently protected on a per-ecoregion basis. This analysis was conducted using the five separate ecoregion classifications as well as the entire state of California as one analytical unit.

Gap analysis permits use of a threshold target for protection of each ecological feature. We chose a conservation target of 30% protection of each vegetation type falling within a given ecoregion. A robust region-specific conservation plan should assign target levels based on relevant ecological data on minimal amounts of area needed for a given ecosystem to maintain ecological function (Wiersma & Nudds 2006). However, without these data being available, we chose to use the approximate mean of conservation targets derived by Svancara et al. (2005) from the conservation assessment literature.

Using this 30% threshold, we calculated the shortfall, if any, in the amount of protected area of each vegetation type within each ecoregion. We refer to this shortfall for a particular vegetation type in a specific ecoregion as “conservation focus”, i.e. the greater the short fall, the greater the amount of conservation activity needed to make up that short fall for that vegetation type in that ecoregion. For each classification scheme, the results from the ecoregion analyses were summed to derive statewide conservation shortfalls for each vegetation type as well as a summed total across vegetation types.

We used the same gap analysis methodology for the historical analysis. Here, we assessed the conservation needs for one ecoregion in the state, California’s Central Valley, as defined by Geographical Information Center (GIC) classification. First, we calculated the area of the five major valley floor natural vegetation types (grassland, freshwater emergent wetland, alkali desert scrub, valley oak woodland, and valley riparian forest; Figure 2.3) in both the current and historic time periods (Figure 2.4). We chose to exclude from analysis the blue oak woodlands located on the perimeter of the valley floor in order to focus our efforts on the highly impacted valley floor, because we had questions concerning seeming discrepancies between the historic and current extents of this vegetation type, and because it was a relatively minor component of the Central Valley as defined by the GIC dataset (only comprising 2.3% of the total area of the GIC-defined Central Valley). Next, we conducted a gap analysis using the current land cover data and a 30% conservation threshold. Finally, we compared these results with those obtained by using the 30% threshold on the historic land cover dataset in order to identify the protection afforded the pre-industrial “natural” ecoregion as opposed to the highly fragmented current conditions.

a)

b)

c)

Figure 2.3. Examples of three remnant ecosystems of the Central Valley: a) grasslands near the Sutter Buttes, b) created freshwater wetlands at Stone Lakes National Wildlife Refuge, and c) riparian forest along Elk Slough in Yolo County. (Photos: P. Huber).

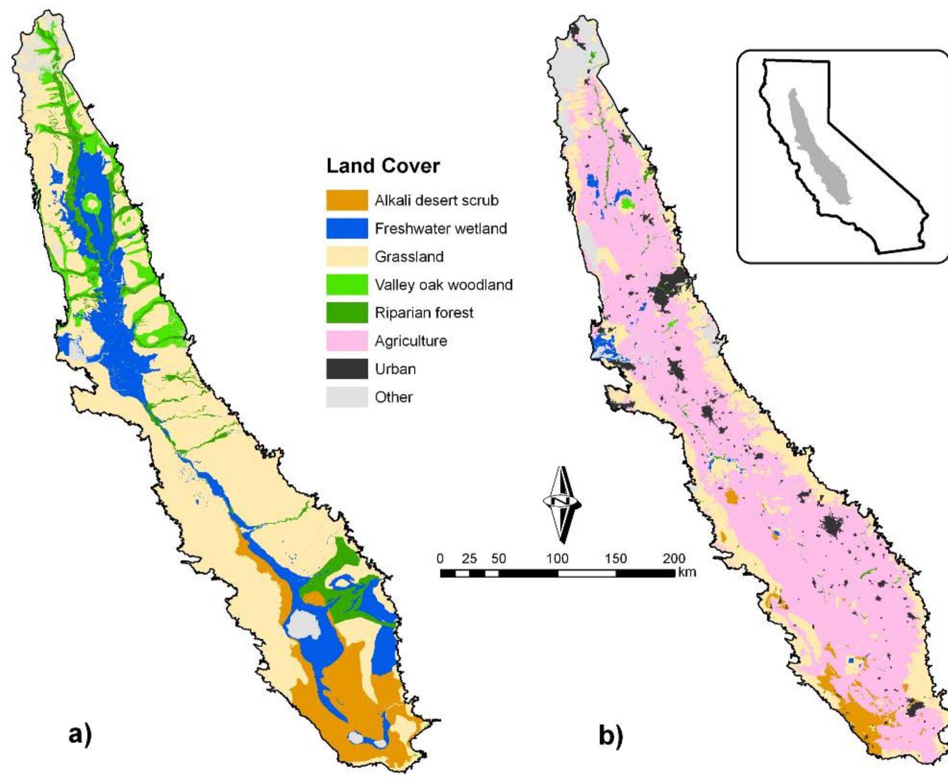


Figure 2.4. Historic (a) and current (b) land cover in the Central Valley of California.

RESULTS

Ecoregions

The number of ecoregions in the classification schemes ranged from 10 (INACC and Jepson) to 18 (USFS). The delineated ecoregions varied by area and boundary location, and also by definition. For the Jepson classification scheme, the ecoregion with the highest rate of overlap across the other schemes is the Mojave Desert, with a mean overlap of 95.5% (Table 2.1). No other Jepson ecoregion had a mean overlap rate greater than 79.5% (East of Sierra Nevada). The INACC Mojave ecoregion covered the largest proportion of the Jepson Mojave Desert ecoregion (97.7%) while the USFS Mojave Desert section covered the least (90.9%). The Jepson ecoregion with the lowest rate of overlap was the Cascade Ranges ecoregion (in the northern central portion of California), with a mean overlap of 48.5%. The Jepson Cascade Ranges ecoregion also had the lowest single overlap rate (35.7%), this with the WWF Sierra Nevada Forests ecoregion. The other Jepson ecoregions with the lowest overlap rate were Northwestern California (57.7%) and Central Western California (69.9%).

Land cover

Of California's 410,000 km² area, 331,000 km² (80.7%) was classified as natural land cover (one of the 44 WHR land cover classes). The areas for each vegetation type statewide ranged from 12.3 km² (palm oasis) and 57.3 km² (saline emergent wetland) to 28,888 km² (annual grassland) and 75,353 km² (desert scrub).

Table 2.1. The ten Jepson ecoregions (Figure 2.1) and the ecoregion in each of the other four classification schemes with which it has the greatest overlap. “Overlap” is the % of the Jepson ecoregion that overlaps with the ecoregion in the respective classification scheme. Mean overlap is calculated for both individual Jepson ecoregions and across ecoregions within each classification scheme.

Jepson	INACC	Overlap	TNC	Overlap	USFS	Overlap	WWF	Overlap	Mean
Cascade Ranges	Modoc	53.6	Klamath Mtns.	50.7	Southern Cascades	54.1	Sierra Nevada Forests	35.7	48.5
Central W California	Central Coast	67.6	Central Coast	92.2	Cent. CA Coast Ran.	50.9	CA Int. Chap. & Wood.	68.8	69.9
East of Sierra Nevada	Sierra	83.5	Great Basin	77.1	Mono	75.9	Gr. Basin Shrub Steppe	81.5	79.5
Great Central Valley	San Joaquin Valley	54.2	Central Valley	89.6	Great Valley	78.4	CA Cent. Vall. Grass.	86.2	77.1
Modoc Plateau	Modoc	93.6	Modoc Plateau	66.9	Modoc Plateau	75.3	E Cascades Forests	70.3	76.5
Mojave Desert	Mojave	97.7	Mojave Desert	95.7	Mojave Desert	90.9	Mojave Desert	97.6	95.5
NW California	Klam./North Coast	88.2	North Coast	51	Klamath Mountains	37.3	Klam.-Sisk. Forests	54.3	57.7
Sierra Nevada	Sierra	93.9	Sierra Nevada	73.5	Sierra Nevada	68.7	Sierra Nevada Forests	68.7	76.2
Sonoran Desert	Colorado Desert	85.5	Sonoran Desert	86.1	Sonoran Mojave Des.	38.5	Sonoran Desert	93.5	75.9
SW California	South Coast	83	South Coast	94.9	So. CA Mtns. and Vall.	64.4	CA Coast. Sage & Chap.	67.3	77.4
Mean		80.1		77.8		63.4		72.4	

Representation

The PCTL dataset covers an area of approximately 207,000 km² statewide (50.6% of California). The amount of protection afforded the vegetation types ranges from 10.9% (valley oak woodland) and 12.4% (annual grassland) to 99.7% (palm oasis) and >99.9% (alpine-dwarf shrub).

Gap analysis

Total area required to make up shortfalls in conservation targets for California vegetation types varied greatly across the five classification schemes (Figure 2.5). The Jepson ecoregional scheme required the least amount of additional conservation area (10,726 km²) while the TNC scheme required the most (13,237 km²). These totals appear to track with the number of ecoregions into which each scheme divides California: schemes with more ecoregions tended to require more additional conservation activity. The two schemes with the fewest number of constituent ecoregions (INACC and Jepson, with 10 each) had the lowest total area needed to achieve conservation targets statewide while the scheme with the most constituents (USFS, with 18) had the second highest total needed. The TNC scheme was somewhat anomalous in that it required the most additional conservation land statewide while only having 12 constituent ecoregions, the middle value of the five schemes.

Many vegetation types showed great disparity between ecoregional schemes in the amount of shortfall on a statewide basis (Table 2.2). Table 2.3 shows the differences between the five schemes for five selected vegetation types. Montane hardwood, the

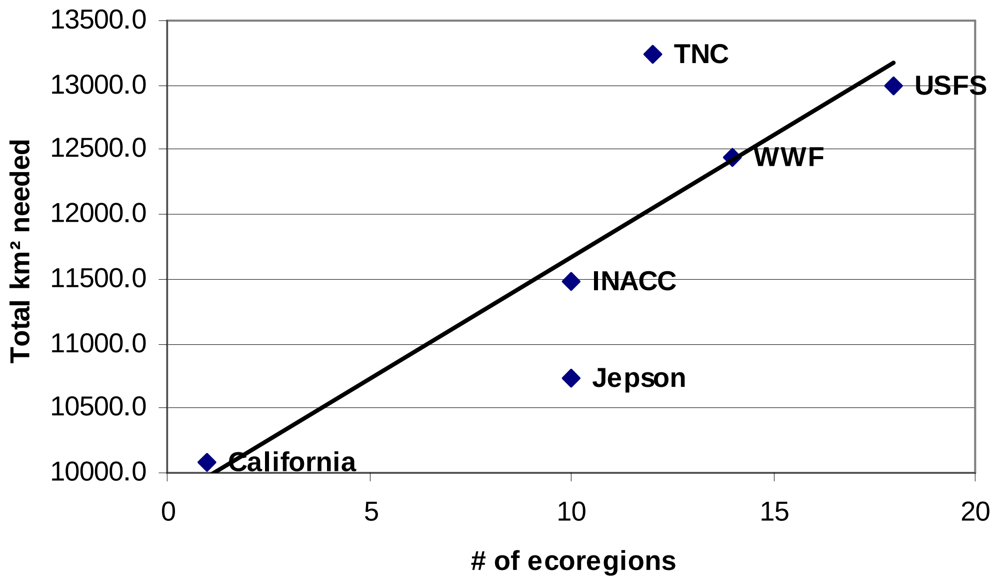


Figure 2.5. The total area of conservation shortfalls calculated statewide ("Total km² needed") compared to the number of constituent ecoregions for each of the five ecoregional classification schemes. Included is the result from using the state of California as a single planning unit.

Table 2.2. The shortfall of each WHR vegetation type and total statewide for each ecoregional scheme (in hectares). Included ("California") is the analysis for the state as one planning unit.

Land Cover	INACC	Jepson	TNC	USFS	WWF	California
Alpine dwarf-shrub	0.0	0.0	0.0	0.0	0.0	0.0
Annual grassland	534,024.8	534,513.3	541,634.8	543,485.9	539,884.6	509,637.8
Alkali desert scrub	2,977.4	2,492.7	22,731.3	22,860.5	19,520.8	0.0
Aspen	0.0	0.0	0.0	0.0	0.0	0.0
Bitterbrush	0.0	2.7	0.0	241.6	0.0	0.0
Blue oak – foothill pine	144,230.5	144,231.1	171,008.8	176,524.5	165,100.3	144,230.3
Blue oak woodland	161,195.9	157,597.9	188,506.5	164,880.1	163,429.3	156,049.9
Coastal oak woodland	56,175.5	56,007.3	56,024.6	56,037.4	56,188.1	56,007.2
Closed-cone pine – cypress	0.0	0.0	0.0	2,098.6	2,681.9	0.0
Chamise – redshank chaparral	9,300.0	569.6	0.0	4,426.2	2,270.7	0.0
Coastal scrub	28,273.9	26,229.8	25,825.5	25,178.9	26,663.9	18,797.7
Douglas-fir	41.6	0.0	18,984.3	13,765.9	10,573.4	0.0
Desert riparian	402.7	292.4	123.7	315.0	284.7	0.0
Desert scrub	0.0	0.0	0.0	309.4	1,606.6	0.0
Desert succulent shrub	0.0	0.0	0.0	0.0	0.0	0.0
Desert wash	0.0	0.0	0.0	0.0	0.0	0.0
Eastside pine	0.0	0.0	0.0	739.6	0.0	0.0
Estuarine	899.5	3,082.4	43.1	70.1	156.4	376.2
Freshwater emergent wetland	1,138.1	967.7	1,291.9	2112.7	1,404.9	0.0
Jeffrey pine	0.0	0.0	256.2	0.0	0.0	0.0
Joshua tree	1,020.8	506.2	692.7	452.4	1,062.2	0.0
Juniper	14,672.8	8,435.2	5,904.4	8,893.5	6,575.8	0.0
Klamath mixed conifer	1,177.4	0.8	1,392.0	0.0	0.0	0.0
Lacustrine	2,533.0	1,783.9	9,326.8	10,598.4	6,848.7	0.0
Lodgepole pine	0.0	0.0	0.0	0.0	0.0	0.0
Low sage	0.0	0.0	0.0	0.0	0.0	0.0
Mixed chaparral	6,860.3	3,148.7	2,747.7	3,366.0	623.2	0.0
Montane chaparral	3,300.5	552.7	514.1	4,323.2	963.9	0.0
Montane hardwood – conifer	14,444.6	155.4	64,266.5	61,606.7	59,307.4	0.0
Montane hardwood	29,423.4	385.3	72,408.7	64,236.8	43,857.1	0.0
Montane riparian	0.0	0.0	311.3	307.0	290.7	0.0
Perennial grassland	1,348.0	1,324.0	2,464.9	1,845.9	2,257.6	0.0
Pinyon – juniper	0.0	0.0	0.0	0.0	0.0	0.0
Palm oasis	0.0	0.0	0.0	0.0	0.0	0.0
Ponderosa pine	3,464.1	0.0	4,351.6	0.0	0.0	0.0
Redwood	70,272.2	70,018.1	72,669.1	70,007.7	73,640.5	70,020.6
Red fir	0.0	0.0	0.0	0.0	0.0	0.0
Riverine	5,325.2	5,325.3	4,197.9	3,936.9	3,763.8	4,494.4
Subalpine conifer	0.0	0.0	0.0	0.0	0.0	0.0
Saline emergent wetland	53.4	54.1	65.0	0.0	49.9	0.0
Sagebrush	426.5	0.0	0.0	1,896.8	209.9	0.0
Sierran mixed conifer	0.0	0.0	1,417.8	0.0	0.0	0.0
Valley oak woodland	48,972.6	48,972.6	48,996.3	50,125.3	48,972.6	48,972.6
Valley foothill riparian	2,669.4	2,792.4	2,110.3	2,197.4	2,183.1	0.0
White fir	37.6	0.0	0.0	0.0	0.0	0.0
Wet meadow	3,385.3	3,163.0	3,399.9	3,341.8	3,175.4	0.0
Total	**1,148,047**	**1,072,604**	**1,323,667**	**1,300,182**	**1,243,547**	**1,008,586**

Table 2.3. Comparison of statewide conservation shortfalls (in km²) across all five ecoregional schemes for five sample vegetation types at the 30% conservation threshold.

Vegetation Type	INACC	Jepson	TNC	USFS	WWF
Alkali Desert Scrub	29.8	24.9	227.3	228.6	195.2
Douglas-Fir	0.4	0.0	189.8	137.7	105.7
Desert Scrub	0.0	0.0	0.0	3.1	16.1
Montane Hardwood-Conifer	144.4	1.6	642.7	616.1	593.1
Montane Hardwood	294.2	3.9	724.1	642.4	438.6

vegetation type with the greatest disparity, differs by 720.2 km^2 in total conservation shortfall statewide. While the Jepson ecoregional scheme calls for additional protection of 3.9 km^2 in the lower reaches of the Sierra foothills (the upper edge of the Great Central Valley ecoregion), the TNC scheme calls for 465.0 km^2 in the North Coast, 257.7 km^2 in the Central Valley, and 1.4 km^2 in the Columbia Plateau ecoregions.

Temporal baseline

There was widespread land cover conversion in the Central Valley. Grasslands had the greatest remaining area of the five land cover types analyzed (Figure 2.6), with 46.9% remaining. However, this figure is potentially misleading in that virtually all (>95%) of the current grassland is comprised of exotic annual grass species as opposed to native grasses (Sims and Risser 2000). While ecosystem function might remain unchanged, the native biodiversity has been largely replaced. Remaining extents of the other four natural vegetation types ranged from 28.8% (desert scrub) down to 6.4% (wetlands).

The gap analysis using current land cover showed that, of the five vegetation types, freshwater wetlands was the only one that was already over the 30% threshold conservation target, at 35% representation within PCTL lands (Figure 2.7). Valley riparian was relatively close (24.6%) while the other three ranged from 7.6% protected to 5.8%. However, when the analysis was applied to historic land cover levels, the most protected vegetation type (grasslands) achieved only 3.0% protection levels. Three of the other vegetation types had 2.2% protection, while valley oak woodland only has conservation status covering 0.6% of its original Central Valley extent (Table 2.4).

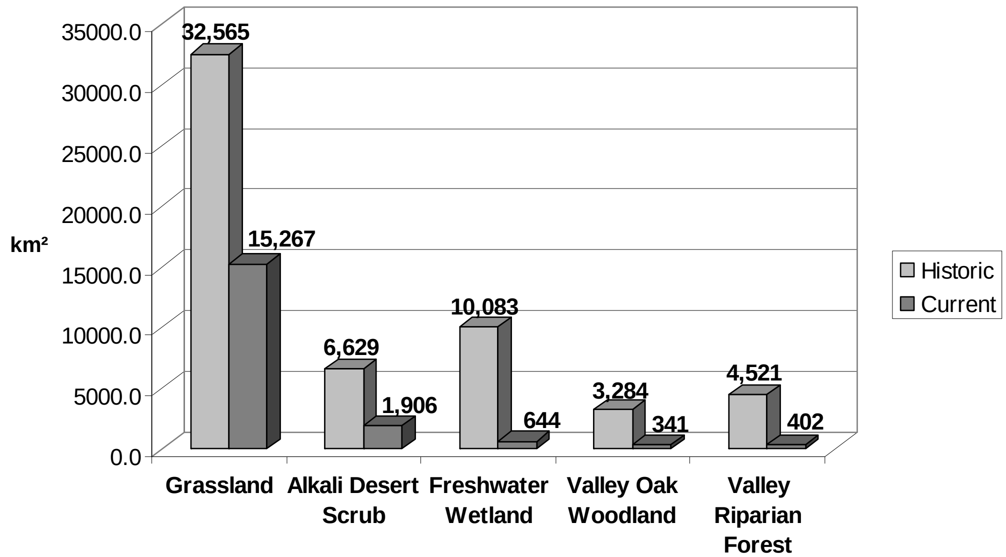


Figure 2.6. Extent (in km²) of the five major vegetation types of the Central Valley, both historically (pre-1900) and currently.

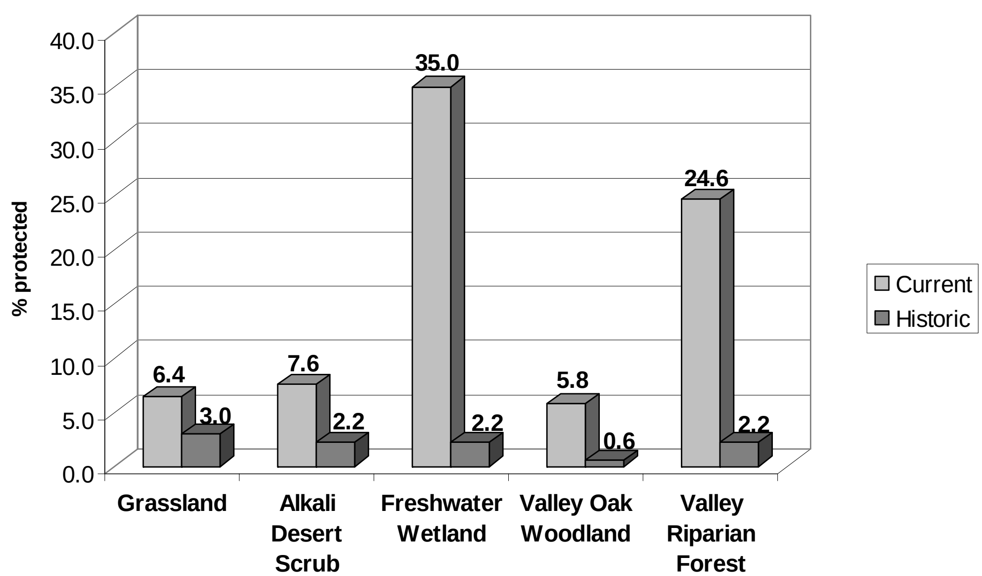


Figure 2.7. Conservation levels for the major vegetation types in the Central Valley, using historic (pre-1900) and current extents. The chosen threshold for adequate levels is 30%.

Table 2.4. The current and historic extents of major Central Valley vegetation types and the measured conservation shortfalls using a 30% threshold for protection (in km², except for "Current %" and "Historic %"). "Targets" are the minimum levels of protected lands of each vegetation type needed in order to reach the 30% threshold. "Protected" refers to current area of each vegetation type occurring on lands managed for conservation. "Current %" refers to the proportion of land protected currently, while "Historic %" refers to the protection proportion of the historic land cover type that is currently protected.

Land cover	Current Extent	Historic Extent	Current Target	Historic Target	Protected	Current %	Current Shortfall	Historic %	Historic Shortfall
Grassland	15,267	32,565	4,580.1	9,769.5	977.1	6.4	3,603.0	3.0	8,792.4
Alkali desert Scrub	1,906	6,629	571.8	1,988.7	144.5	7.6	426.9	2.2	1,843.8
Freshwater wetland	644	10,083	193.2	3,024.9	225.7	35.0	0.0	2.2	2,799.5
Valley oak woodland	341	3,284	102.3	985.2	19.8	5.8	82.5	0.6	965.4
Valley riparian forest	402	4,521	120.6	1,356.3	99.1	24.6	21.7	2.2	1,257.4

DISCUSSION

These results clearly show that the choice of planning area boundaries and temporal baselines can have significant impacts on conservation targets and goals. The TNC ecoregional scheme required the most additional area (13,237 km^2) added to achieve representational goals, 23% more total land (2,511 km^2) than did the Jepson classification, which required the least new acquisitions (10,726 km^2). Acquiring land for conservation purposes is an expensive endeavor. Using a rough estimate of $15,000 per acre for land in California (derived through a quick on-line survey of average land prices for parcels greater than 100 acres at scattered locations across California), this difference in land requirements amounts to some $9.3 billion. While there are other cheaper means of conferring conservation status on lands (e.g. easements or local zoning ordinances), choice of ecoregional scheme in this context is still a multi-billion dollar decision.

This analysis shows that increasingly finer geographical delineation of a given study area can lead to identification of greater conservation needs. The ecoregional schemes that expressed greater geographical diversity tended to require more new conservation lands. Thus if conservation planners wish to protect vegetation types across different portions of their range in order to potentially include varying patterns within the vegetation types, greater total area will be needed. Similar effects of scale in conservation planning have been observed by other authors (Erasmus et al. 1999, Vazquez et al. 2008). These studies found an even greater scale effect (up to an order of magnitude) on conservation targets than was identified in this study, but the same general relationship held.

While the relationship between the number of planning units, their diversity of habitat conservation targets within a state-wide classification, and the overall conservation area required, can be hypothesized, less obvious is the effect of location of the planning unit boundaries on the required total area (Figure 2.8). For instance, although both the INACC and Jepson schemes divide California into 10 ecoregions, the INACC scheme required 755 km^2 more land to achieve the conservation targets, while the TNC scheme (with 12 ecoregions) required 801 km^2 more than the WWF scheme. One potential explanation for these differences lies in the fact that both the Jepson and WWF schemes are heavily based on vegetation types, whereas the other schemes include other factors as well, such as landforms (TNC 2000) and management boundaries (INACC 1992). Thus the other schemes may separate the outer and possibly little-protected portions of the vegetation types' ranges from the core range, requiring additional conservation action in order to attain the conservation targets.

Even when ecoregional schemes lead to similar overall quantities of additional conservation land needed, the location of the underrepresented lands can vary considerably. For instance, the TNC and USFS schemes lead to conservation needs differing by only 235 km^2, yet the TNC scheme puts an emphasis on the Sierra Nevada foothills (Figure 2.7) while the USFS scheme has less of an emphasis there and sees a greater need for conservation in the inner portions of the north and south coasts. This change in conservation focus results from the identification of the Sierra Nevada Foothills, Northern Coast Ranges, and Southern Mountains and Valleys as ecoregions in the USFS scheme. In the case of the latter two ecoregions, this splitting results in relatively underprotected areas being separated from areas with higher conservation

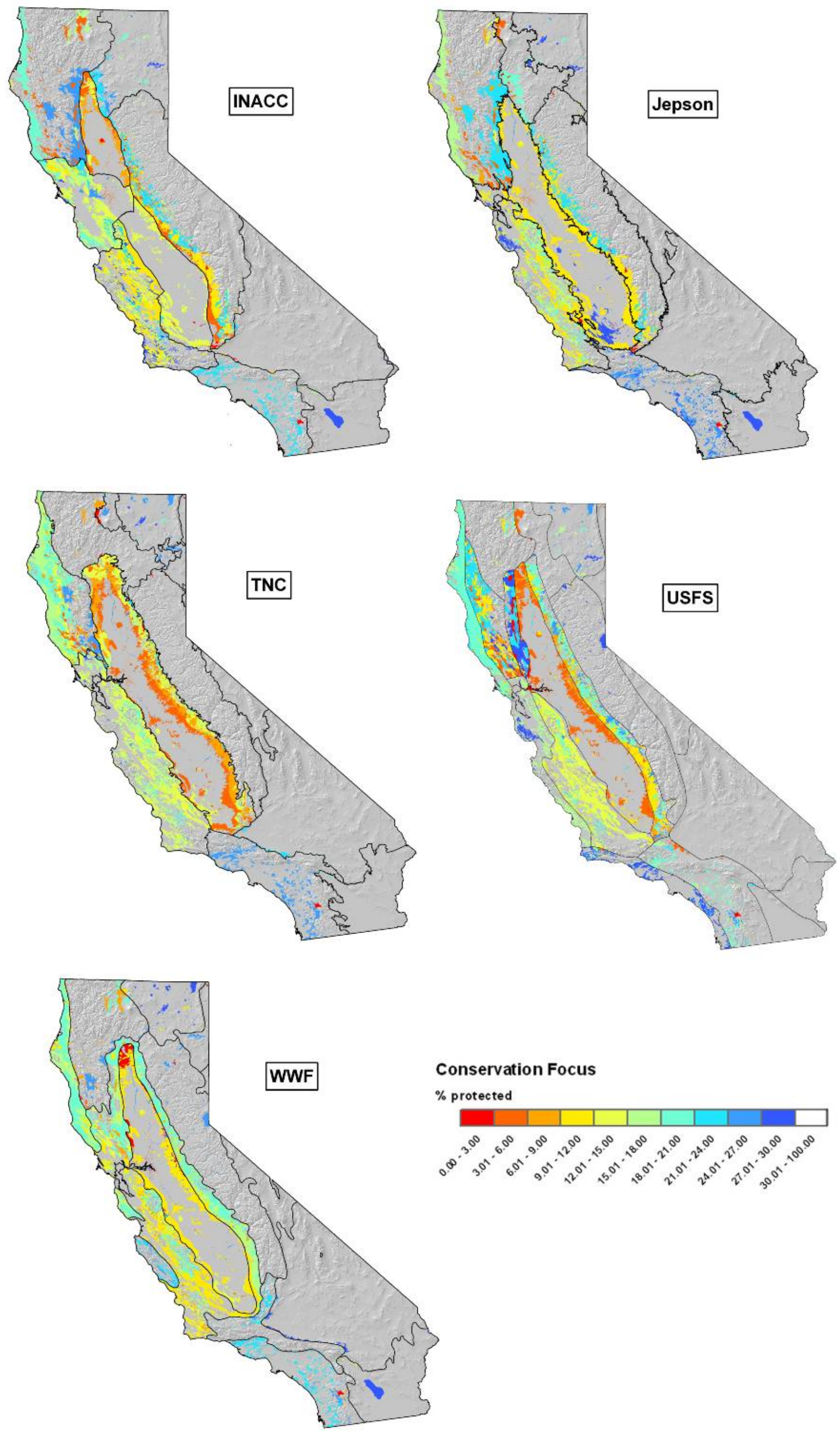


Figure 2.8. Areas of conservation focus for each of the ecoregional schemes. Red indicates those areas with the greatest conservation needs. Gray indicates either non-natural land cover (e.g. agriculture) or areas surpassing the 30% conservation threshold for that WHR vegetation type within that ecoregion.

levels in adjacent areas. However, in the case of the USFS Sierra Nevada Foothills, the area with the higher levels of protection is split from the Great Valley (i.e. the Central Valley), an ecoregion with relatively minimal levels of protection.

Further, individual vegetation types in different ecoregional schemes that require similar levels of additional protection statewide may also display spatial disparity in areas selected for conservation. An example of this can be seen in a comparison of annual grassland conservation needs (Figure 2.9). Only a 4.9 km² difference in total area of additional grassland needed for conservation statewide is identified between the INACC and Jepson schemes. However, the INACC scheme places a heavier focus on the Sierra Nevada foothills, Sacramento Valley, and North Coast grasslands, while the Jepson scheme emphasizes conservation of the San Joaquin Valley, San Francisco Bay area, and South Coast grasslands. The choice of ecoregional scheme can thus lead to substantially different conservation planning priorities for specific vegetation types.

The complex physical geography of California drives many of these boundary effects. The overlap analysis conducted on the Jepson ecoregions shows that in much of the state there is pronounced disparity in ecoregion definition. While several ecoregions show a high rate of congruence across classification schemes (particularly the Mojave Desert ecoregion), most of the remainder of the state has congruence rates of ecoregions of 50-75%. The low ecoregional overlaps are especially apparent in the northern and coastal portions of the state. There is little consensus as to how to classify this area into subregions which can lead to a large range of potential conservation needs scenarios for large portions of the state.

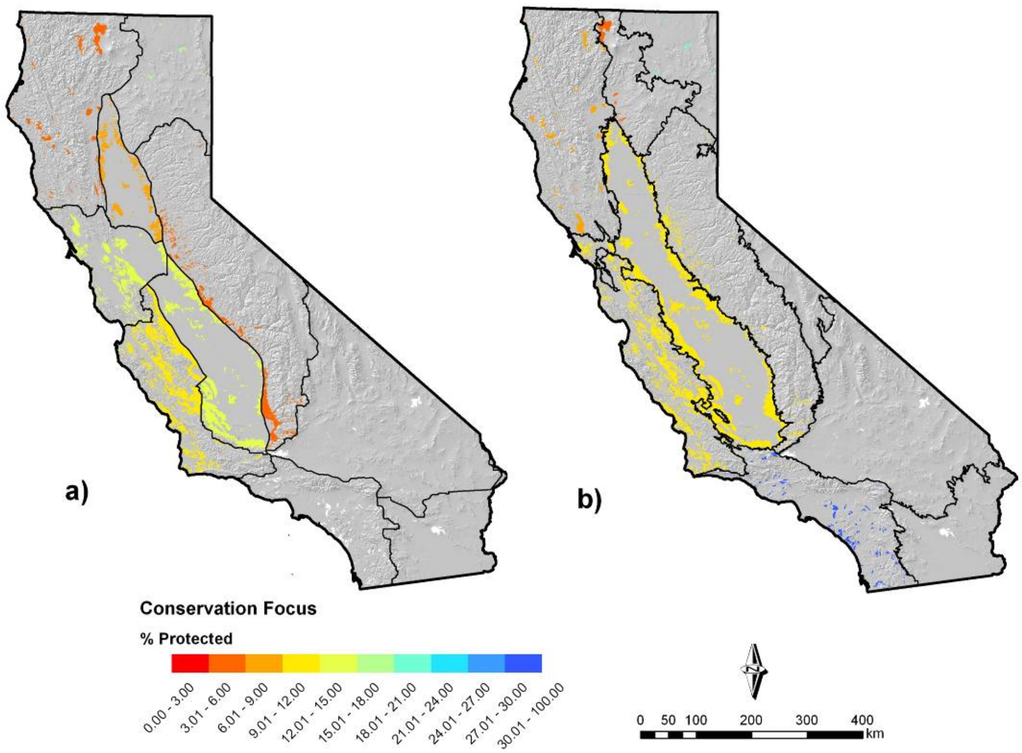


Figure 2.9. Conservation focus for annual grassland in two ecoregional schemes: a) INACC, and b) Jepson. Red indicates areas of greater conservation need; "% protected" refers to the proportion of annual grasslands found within existing conservation areas in that particular ecoregion.

Choice of temporal baseline can also have important ramifications in conservation assessments of highly impacted regions. While analysis of conservation shortfalls of the contemporary Central Valley ecoregion show that freshwater wetlands, and to a lesser extent valley riparian forest, can be considered to have surpassed the minimal protection threshold, adoption of an historical viewpoint identifies large conservation shortfalls for all of the major valley bottom land cover types. With the highest conservation level for historic land cover at 3.0% (grassland), there would obviously be a much larger conservation need if loss from this baseline were to be incorporated in the conservation assessment.

A conservation threshold, such as that chosen for this analysis, should indicate the minimal amount of intact habitat needed to ensure continued ecological function and process (Svancara et al. 2005). If an ecosystem is severely degraded, protection of the contemporary remainder of this ecosystem in its entirety might not be enough to ensure future ecological viability. This dynamic may well be at work in the Central Valley. With the exception of the grasslands component, current Central Valley vegetation cover (protected or not) falls short of the 30% threshold, and even the grasslands have been converted to non-native species for the most part. If 30% is the minimal level of protection necessary for future ecological viability, then restoration coupled with conservation becomes the necessary conservation approach for this region (Figure 2.10; Table 2.3). If all the remaining examples for the four non-grassland vegetation types were conserved in the Central Valley, it would be necessary to restore and then conserve from 82.7 km^2 (desert scrub) to 2,380.8 km^2 (wetlands) of converted land to achieve the 30%

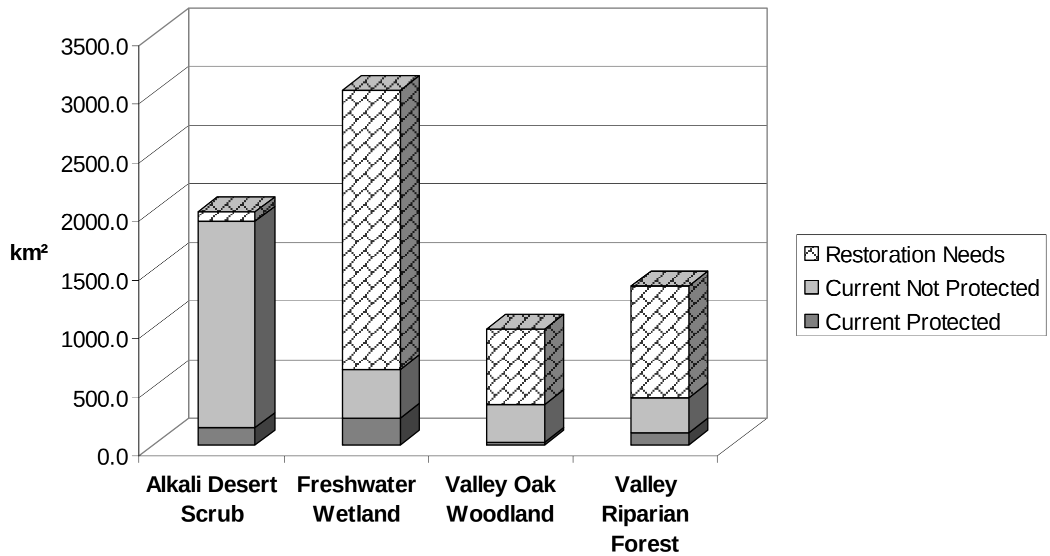


Figure 2.10. Restoration needs in the Central Valley, California, for the four non-grassland vegetation types in order to achieve the 30% conservation threshold based on historic vegetation extents. These restoration needs calculations assume that all current existing but non-protected vegetation occurrences receive conservation status. The tops of each bar indicate the total amount of that vegetation type that would need to be protected in order to achieve a 30% threshold based on historic vegetation extents.

Table 2.3. The conservation and restoration needs for four Central Valley vegetation types in order to achieve 30% protection of historic land cover for each type (in km^2).

Vegetation Type	Current Protected	Current Not Protected	Restoration Needs
Alkali Desert Scrub	144.5	1,761.4	82.7
Freshwater Wetland	225.7	418.3	2,380.8
Valley Oak Woodland	19.8	321.0	644.5
Valley Riparian Forest	99.1	303.3	953.8

threshold for minimal protection. The combined minimal restoration needs as calculated here for the Central Valley are one third larger than Yosemite National Park.

The results from this work can help focus the attention on specific areas displaying the greatest disparity of ecological need. Figure 2.11 illustrates the maximal difference in conservation focus between ecoregional schemes across California and hence the areas of potentially highest uncertainty as to levels of protection. Four of the locations of high disparity are indicated: the inland North Coast, the eastern San Francisco Bay area, the Sierra foothills, and the Tehachapi area. In some schemes these areas appear to contain relatively well-protected vegetation types, while in other schemes these vegetation types fall far short of conservation goals. These four areas are places of ecotonal transition, making regional definition a complicated process. The fact that these areas reflect uncertainty in conservation need is directly linked to their ecotonal location. As Bailey notes (1996), ecosystems exist along a continuum and thus any boundary demarcation is a necessarily arbitrary designation. It is precisely in these areas that we should expect to see ambiguity in the setting of conservation targets.

Analysis of the five ecoregional schemes identified locations of conservation needs recognized across all the schemes. The mean conservation focus across the schemes (Figure 2.12) identified several areas that are deficient in conservation lands. These areas with low rates of mean conservation and greater need for conservation include: the grasslands of Shasta Valley in northern California, the valley and coastal oak woodlands of Sonoma and Napa Counties in the interior northern portion of the state, the grasslands of the Central Valley, and a large wet meadow complex in northern San Diego County in southern California. Other areas with a relatively low mean conservation rate

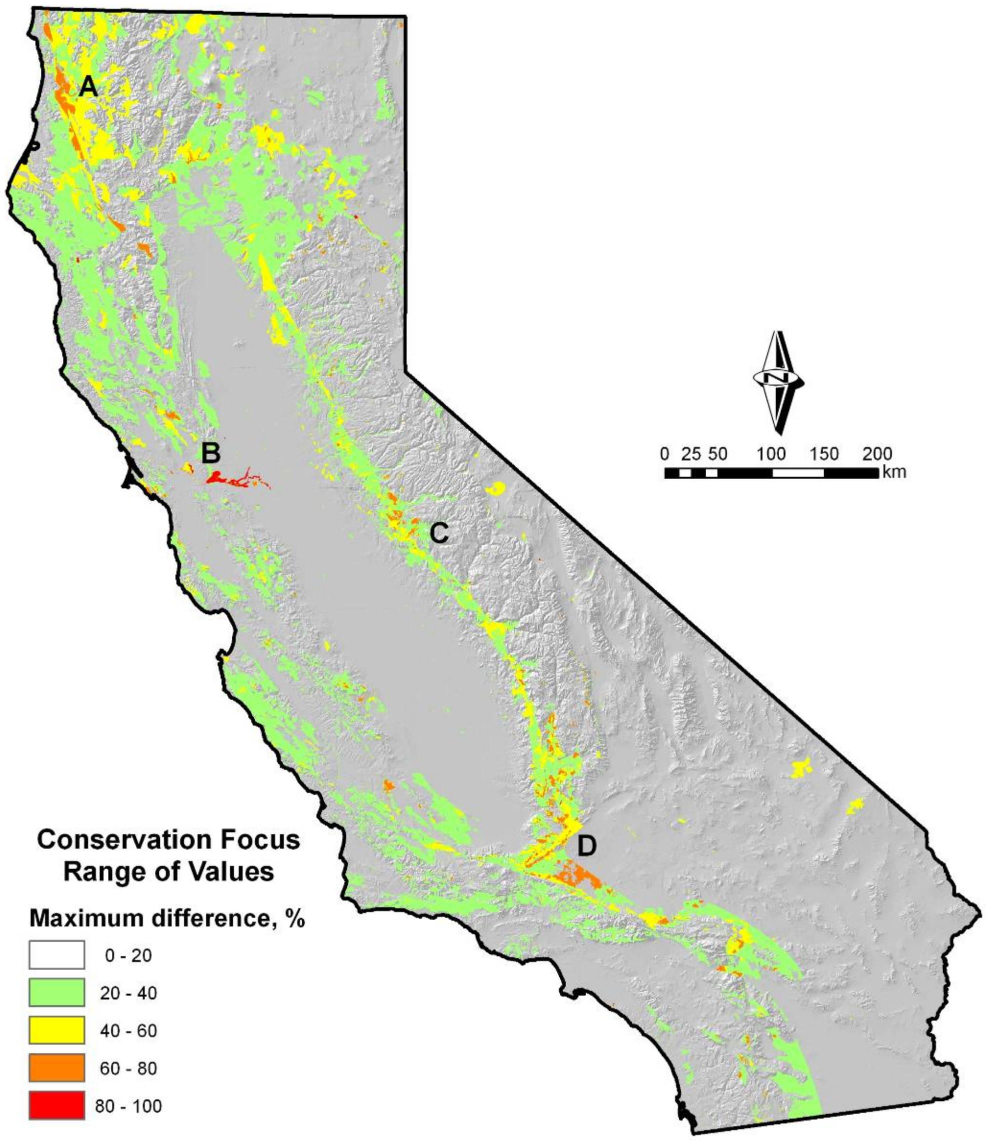


Figure 2.11. The range of conservation focus values between the results for the five ecoregional classification schemes. Raster cell values are equal to the difference between the highest and lowest rate of conservation across the ecoregional classification schemes of the land cover type found at that location. Red indicates areas with high variation in assessed conservation need, including: (A) the inland North Coast, (B) the eastern San Francisco Bay area, (C) the Sierra foothills, and (D) the Tehachapi area.

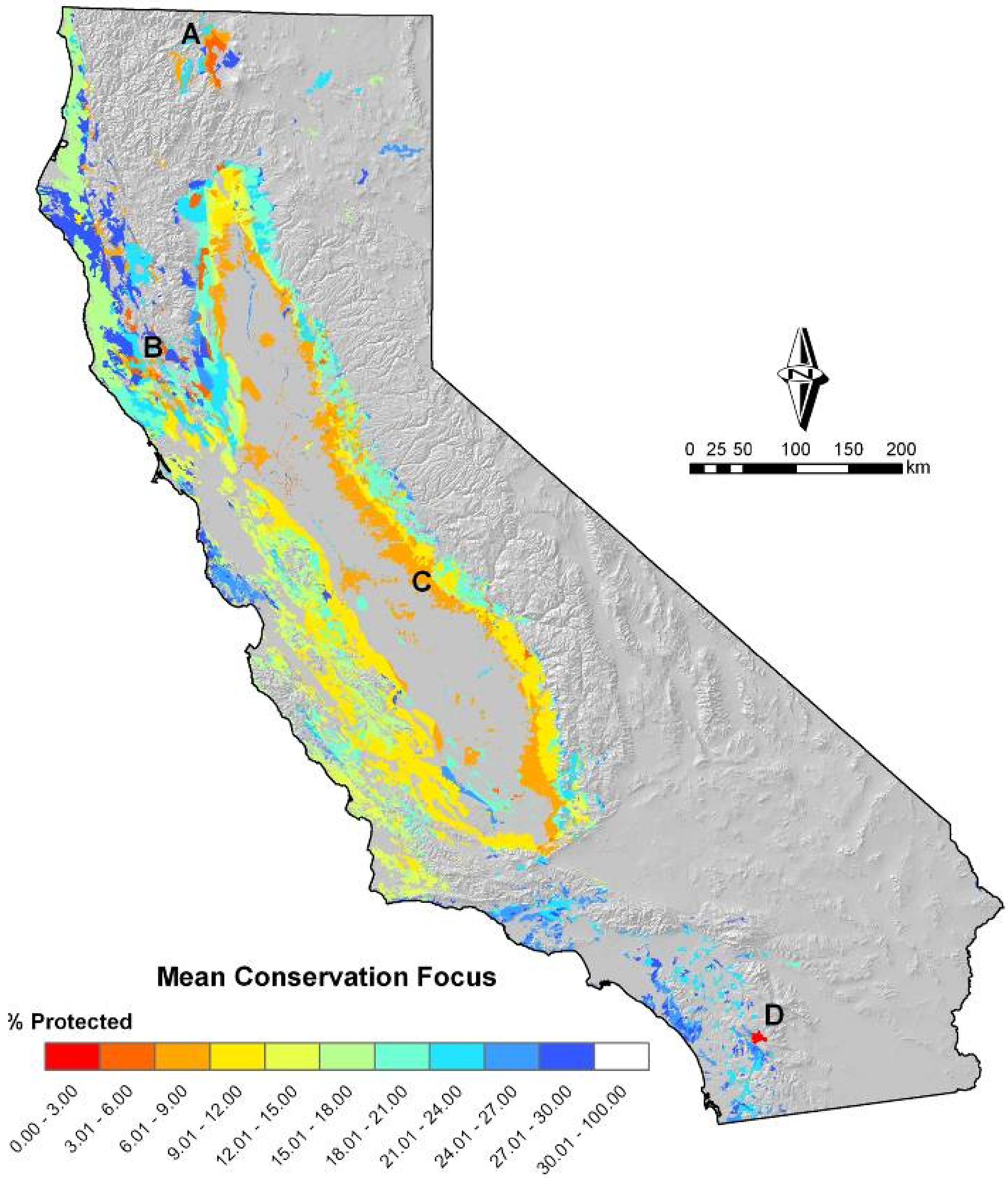


Figure 2.12. The mean conservation focus value across the five ecoregional schemes. Red indicates areas of low levels of protection across all schemes and thus greater conservation need. These include: (A) the grasslands of Shasta Valley in northern California, (B) the valley and coastal oak woodlands of Sonoma and Napa Counties in the interior northern portion of the state, (C) the grasslands of the Central Valley, and (D) a large wet meadow complex in northern San Diego County in southern California.

include the northern coast and the interior Central Coast region. This analysis is one potential direction that could be taken in prioritizing conservation actions designed to help achieve vegetation type representation in California.

While it would certainly be beneficial to planners to be able to take the results from this work and draw the conclusion that one of these ecoregional classification schemes is better suited than the others to address conservation needs in California, caution is urged in this respect. The specific questions being addressed and conservation objectives desired should drive the choice in spatial parameters. That being said, the classification scheme that recognizes the most geographic diversity, i.e. that which delineates the most discrete planning units (in this case the USFS scheme), has the greater chance of accounting for ecological differences across the state and hence the best chance of inclusion of this diversity in a conservation planning process.

Spatial and temporal parameter selection has a high likelihood (judging from these analyses) of affecting not only gap analysis but other conservation assessment programs as well. For example, biodiversity "hotspot" analysis (Myers et al. 2000) results could change markedly if spatial boundaries are shifted or if researchers opt for a different planning unit scale. We believe that the effects demonstrated in this paper should be explicitly addressed in planning and analysis processes.

While the results of this analysis can help elucidate the impacts of scale effects in conservation planning as well as provide some guidance to managers as to areas of broad conservation needs consensus (i.e. areas with a small maximal difference and low mean conservation focus) and unresolved questions as to conservation needs (i.e. areas with a large maximal difference in conservation focus), several caveats and limitations should be noted. Like many inquiries concerning broad geographic areas, data quality and availability are issues that need to be addressed. For instance, the conserved lands dataset we used includes information on fee title conservation land; but conservation easements are not included. In regions such as the Central Valley, much of the important conserved land is protected by private easements rather than publicly owned. The PCTL conservation lands dataset shows there to be 378,125 ha of fee title protected lands in the Central Valley. However, the central portion of the Sacramento Valley (itself a sub-region accounting for approximately 1/8 of the total area of the Central Valley) contains roughly 22,000 ha of conservation easements on private lands (D. Cameron, pers. comm.). If the same proportion of easement area holds for the full Central Valley then, we can expect roughly 176,000 ha of easement lands within the region, or approximately 30% of the total conservation lands. This could potentially prove to be important when trying to understand conservation shortfalls in areas such as the Central Valley grasslands, which in this analysis are identified as one of California's "hot spots" in conservation needs.

Another dataset needed to conduct a truly effective gap analysis is an accurate statewide vegetation coverage with fine spatial resolution. This is especially important in highly ecologically degraded areas where much of the remaining natural vegetation is found in remnant patches smaller than current minimum mapping units. Without having a better understanding of current land cover in California, any conservation assessment will be at best an estimate of the true conservation needs.

Another caveat for this analysis is that it is not meant to be a full conservation needs assessment for California. The intent is rather to show that assumptions made during such an assessment might well have large impacts on the result. First and foremost is the ecoregional scheme selected as shown in this study. Additionally, the only conservation targets included in this analysis are vegetation representation components. A full needs assessment would need to examine representation of a suite of sensitive species as well as ecological processes and patterns (Rodrigues et al. 2004). For example this study does not include the effects of landscape fragmentation by roads, urban areas and agriculture that can severely limit the movement and population viability of numerous wide-ranging mammals. Further, a sensitivity analysis would be in order to more fully understand how assumptions in this project, such as the 30% representational goals, affect the results.

CONCLUSIONS AND IMPLICATIONS

This paper found that the scale of analysis affects conservation targets and planning outcomes. As the scale used to derive representational goals becomes finer, the overall level of the conservation effort necessary to achieve those goals also increases. Additionally, this study indicates that location of sub-unit boundaries can also play an important role in the determination of conservation targets even when the spatial scale remains relatively constant. When this parameter choice is coupled with a selection of a temporal baseline (especially in human-dominated landscapes) we can anticipate a great latitude in potential conservation target levels in a given conservation planning effort.

Land managers and decision makers will need to understand how these choices affect the final planning outcomes. To be most effective, the selection of sub-unit boundary criteria should reflect the overall goals of the planning effort. Similarly, a knowledge of the current functioning of the ecological systems in question can help point to an appropriate temporal baseline from which to frame the planning process. While there may not be clearly defined methods for making these selections, it is important for managers to be aware that important choices are being made when setting these parameters.

CHAPTER 3

SPATIAL SCALE AND ITS EFFECTS ON CONSERVATION NETWORK DESIGN: TRADE-OFFS AND OMISSIONS IN REGIONAL VERSUS LOCAL SCALE PLANNING

INTRODUCTION

Ecological processes take place at a variety of spatial scales (Forman 1995). Genetic exchange can occur at a local scale while disturbance processes can be regional in nature (Baker 1992, Brown et al. 1999, Greco et al. 2007), and migratory behavior may take place at continental or even inter-continental scales (Fuller et al. 1998). Successful ecological planning then should take these hierarchical scales into explicit consideration in order to provide the necessary conservation framework for maintenance of key ecological patterns and processes (Poiani et al. 2000).

One planning scale that has received much attention over the past several decades is the ecoregion (Omernik 1987, Bailey 1996). Conservation planning at this scale can allow for inclusion of important ecological disturbance processes such as fires and flooding. It also is crucial for preservation of potential animal movement within metapopulations of many large vertebrate species, especially in increasingly fragmented landscapes (Beier and Noss 1998). These processes are difficult to plan for at a local scale. Thus, many land use planning agencies and conservation organizations have devoted resources to creating ecoregional conservation plans (Shilling et al. 2002, Cowling et al. 2003, Miller et al. 2003, Thorne et al. 2006).

However, in countries such as the USA, there are few legal structures for implementation of regional conservation plans. Most land use planning authority resides at the local level, with entities such as cities and counties controlling most land use decisions within their borders (Theobald et al. 2005). This “home rule” regulatory environment generally leaves little room for state or federal agencies to implement regional conservation planning projects. While there are exceptions (e.g. Miller 1996, Hoctor et al. 2000, ICEBMP 2000), most regional conservation planning is conducted by non-profit organizations such as the Nature Conservancy (Anderson et al. 2006) or the Wildlands Project (Crumbo and George 2005). However, full implementation of a regional conservation plan is well beyond the capacity of any one organization to accomplish. Thus, unless new legal land use structures are created, implementation of regional conservation plans will require integration of multiple local-scale conservation networks.

One method for creating a regional conservation plan begins by identification of important core areas that contain either regionally important ecological features or are amenable to restoration of those features (Noss et al. 1999, Margules and Pressey 2000, Groves 2003). These features include rare or sensitive species (Rothley 1999, Wiersma 2007), high biodiversity (Margules et al. 1988, Prendergast et al. 1993, Arponen et al. 2004), endemic species (Spring et al. 2007), are areas largely free of human disturbance (Jaeger 2000, Crist et al. 2005), or contain unique ecological features or processes (Turner et al. 1999). Another type of potential planning approach is the use of focal species (Lambeck 1997), in which a species considered to be the most sensitive to perturbation of a particular ecosystem pattern or process is used as a surrogate (or “umbrella”) in the conservation planning process for other species sensitive to disturbance of the same ecosystem component. Core reserve areas that are identified to serve the needs of these focal species are then presumed to offer similar protection for the other species that fall under their “umbrella” (Lambeck 1997, Caro and O’Doherty 1999).

These core areas identified in the conservation planning process are then linked by corridors meant to permit the movement of animal and plant individuals or propagules between the cores in order to maintain the viability of the core populations (Rosenberg et al. 1997, Bennett 2003). While there is still concern whether corridors provide a universally effective means of conservation (Simberloff et al. 1992, Davies and Pullin 2007), several recent studies have shown that they can provide ecological benefits to regional populations (Tewksbury et al. 2002, Damschen et al. 2006).

Some effects of spatial scale on the conservation planning process have been noted in the recent ecological literature. Change in the grain size of datasets used in reserve selection has been shown to alter the resulting reserve networks (Andelman and Willig 2002, Rouget 2003), and species range size and location as defined by species distribution models is also affected (Seo et al. 2008). Additionally the size of the planning units can affect identified conservation networks (Warman et al. 2004, Pascual-Hortal and Saura 2007) as can the extent of the planning area (Pascual-Hortal and Saura 2007, Vazquez et al. 2008). However, little work has been done to date on investigating scale effects on whole ecological networks that incorporate connectivity analysis and corridor planning in addition to reserve selection. It remains to be seen how functional connectivity (sensu Noss and Daly 2006) changes as the spatial scale of analysis is changed.

This paper examines the level of spatial congruity between conservation networks designed to meet ecological needs that are identified at an ecoregional scale and at local scales within the larger region. Here we ask: is the whole equivalent to the sum of its parts? Our general hypothesis is that conservation networks derived at one scale will not be the same as those derived at another scale in the same location. If our hypothesis proves to be correct, the implication is that land planners and managers will need to balance the ecological needs identified at multiple spatial scales in order to create effective conservation networks.

METHODS

Study Area

The analysis in this paper was conducted in the Central Valley ecoregion of California (Figure 3.1). We used the ecoregion boundary as defined in the Jepson flora's ecoregion scheme (Hickman 1993). This ecoregion is largely agricultural, having been converted beginning 150 years ago with the advent of the California Gold Rush. The remaining natural areas in the region are generally small and highly fragmented remnants embedded within this agricultural landscape matrix. These land cover types are currently facing urbanization pressure with the population of the region expected to approximately double in the next 40 years (PPIC 2006; Figure 3.2). This ecoregion encompasses portions of 29 counties (only one of which is entirely within the ecoregional boundaries) (Figure 3.1).

Data preparation for core reserve selection

We created a regularly spaced planar tessellation surface of hexagonal cells (13.3 hectares in size; similar to a raster grid and frequently used in conservation planning

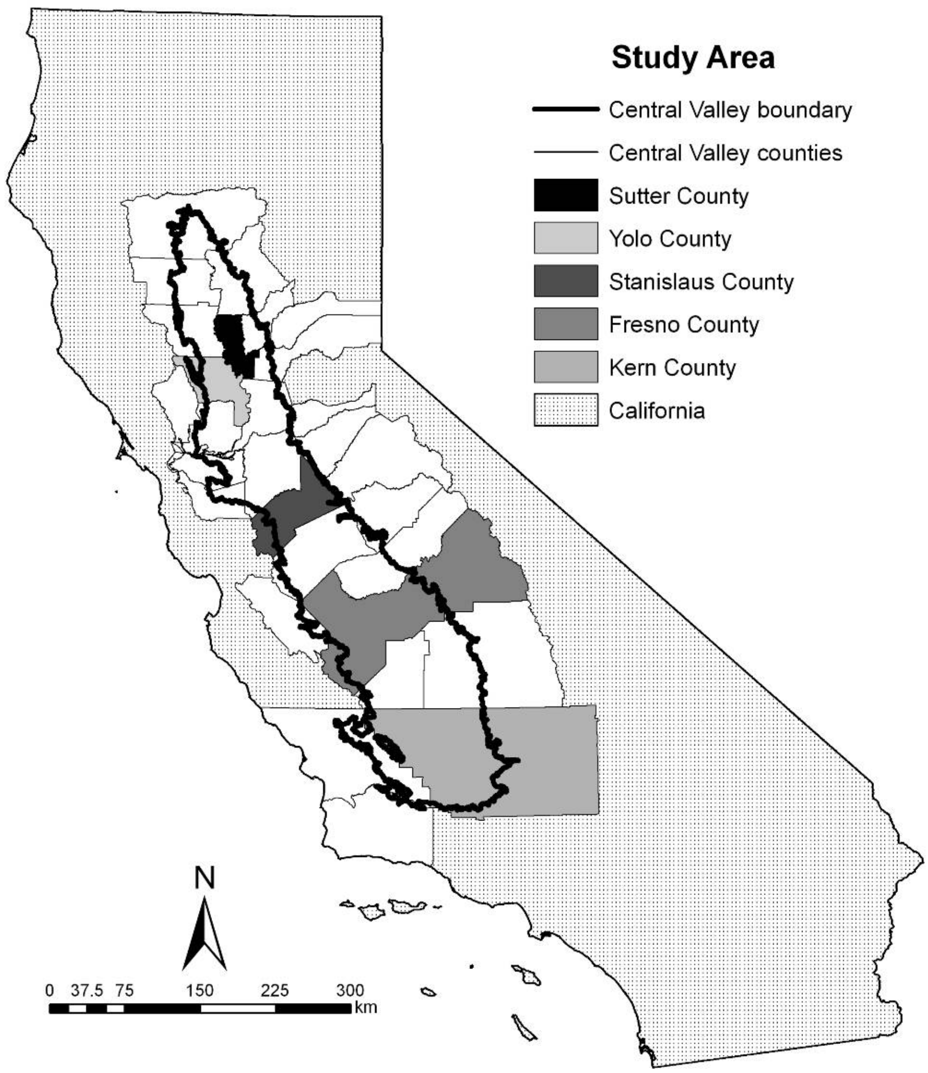


Figure. 3.1. The location of the Central Valley ecoregion within California. Also shown are the 29 counties portions of which comprise the ecoregion with the five used for analysis indicated in shades of gray.

a) 

b) 

c) 

Figure 3.2. Human-dominated landscapes in the Central Valley, California: a) remnant riparian forest surrounded by an agricultural matrix along Ping Slough, Sutter County, b) coyote in agricultural fields adjacent to new urban development in Davis, Yolo County, and c) new urban development in an agricultural area in Sacramento County. (Photos P. Huber).

because of the uniform boundaries with neighboring polygons) that constituted the planning units within the ecoregional boundary, by using a geographic information system (GIS; ESRI 2005). Conservation land polygons, as identified in the Public, Conservation and Trust Lands dataset (California Resources Agency 2005), were then embedded in the hexagonal planning unit surface using the ArcGIS Update tool. Polygons smaller than one hectare were merged with neighboring polygons, creating a final study area of 424,805 planning units.

Eight conservation targets, comprised of seven focal species and one unique ecological community were selected to represent the major ecological patterns and processes of the study area: tule elk (*Cervus elaphus nannodes*; chosen to represent large-scale lowland and upland connectivity); bobcat (*Lynx rufus*; fine-scale forested connectivity); giant garter snake (*Thamnophis gigas*; freshwater wetlands); pronghorn (*Antilocapra americana*; grasslands); western yellow-billed cuckoo (*Coccyzus americanus occidentalis*; riparian forest); San Joaquin kit fox (*Vulpes macroitis mutica*; desert scrub); Swainson's hawk (*Buteo swainsoni*; agricultural/natural areas interface); and vernal pool community complexes, included as a unique ecological feature. These focal elements, if adequately protected, can serve as "umbrellas" for other resident plant and animal species. We developed habitat suitability models for the seven focal species.

We used the full Central Valley ecoregion as the modeling area for tule elk, bobcat, pronghorn, and Swainson's hawk. For giant garter snake, kit fox, and yellow-billed cuckoo we used range maps to spatially restrict the habitat analyses as these species historically did not exist throughout the entire ecoregion.

The highly fragmented nature of the study area landscape necessitates substantial habitat restoration in order to assemble large enough or well-connected enough blocks of land suitable for self-sustaining populations of many species. Thus, we could not use standard habitat suitability indices solely based upon existing conditions, but rather we had to take into account the context and "restorability" of human-converted planning units (while these might also change with scale it was beyond the scope of this analysis to address this change). To do this we chose to use the following habitat variables (for all species unless otherwise noted):

- Current land cover: a value of 0 to 1 for each major land cover type for each focal species was taken from the California Wildlife Habitat Relationships (CWHR; CDFG 2005) dataset. The land cover dataset used was the statewide California Department of Forestry and Fire Protection land cover dataset (FRAP 2002).

- Road density: TIGER road data. Road density was calculated in km/km^2 at both a 3km and 5km radius. These densities were then converted to a 0 to 1 scale and inverted, so that low density raster cells would have a value of 1 while high density raster cells had a value of 0.

- Urban area density: Farmland Mapping and Monitoring Program (FMMP; FMMP 2004) urban data. Urban areas were given a value of 1 and non-urban areas a value of 0. The average urban area value within both a 3km and 5km radius was calculated. These density values were then converted to a 0 to 1 scale and inverted, so that low density urban areas had a value of 1 and high density urban areas a value of 0.

- Natural area density: FRAP land cover data (FRAP 2002). All native vegetation types (plus annual grasslands) were included in this category. This metric was used to lower the overall scores of those habitats that are largely surrounded by potentially incompatible land uses (e.g. agriculture or urban) and to raise those of habitats embedded within a natural matrix and thus less susceptible to detrimental human-caused edge effects. To calculate this value, the natural vegetation types were given a value of 1 and non-natural types a value of 0. The 3km and 5km radii were used to calculate natural area densities which were then converted to a 0 to 1 scale with areas of high natural area density receiving a value of 1.

- Current land management status: Public and Conservation Trust Lands data (California Resources Agency 2007), a dataset comprised of all public lands as well as private lands managed for their conservation value. Lands contained in this dataset were given a value of 1 and lands outside these boundaries a value of 0.

- Waterway density (for giant garter snake only): National Hydrography Dataset (NHD; USGS 1999) waterway data. Waterway density was calculated in km/km^2 at a 3km radius. These values were then converted to a 0 to 1 scale with areas of high waterway density receiving a value of 1.

- Tree/grass interface density (for Swainson's hawk only): FRAP land cover data (FRAP 2002). The density of forest land cover types was calculated with a ceiling of 0.5 put on this value. The same was done with grasslands plus field-type agricultural land cover types. Results were summed, thus the overall range for the interface density was 0 to 1.

Density surfaces were calculated using a 5 km radius for tule elk, yellow-billed cuckoo, bobcat, and a 3 km radius was used for all the other focal species. For each focal species' habitat suitability model, the current land cover value was given half of the overall weight in order to ensure that existing natural areas were given the highest priority for inclusion in the identified conservation network. The other variables (either 4 or 5 depending on the species) were then equally weighted and summed to provide the other half of the overall weight of the habitat score. The sum of the land cover score and the other variable's scores were then converted to a 0 to 1 scale (with 1 being the highest value for a particular species). The vernal pool focal element was given a simple binary score of 0 (not present) or 1 (present) determined by the boundaries of the vernal pool complexes dataset (USFWS 1998). This dataset represents not just the actual pools themselves but the surrounding uplands as well.

Planning units were given a habitat value for each focal element by multiplying the planning unit area by the average value of the raster cells representing focal element habitat suitability that fell within the planning unit. While the majority of planning units were identical in area (i.e. the hexagonal grid), existing conservation lands planning units ranged in size. Also, many of the planning units on the study area boundary or existing conservation lands boundaries were smaller than the standard size. Thus the hexagonal (i.e. non-conservation land) planning units were assigned values ranging from 0 to 133,000 (converting hectares to m^2 and multiplying by the habitat values ranging from 0 to 1) for each of the 8 focal elements.

Core reserve selection at the regional scale

The planning units along with their values for the 8 conservation targets were inputted into the MARXAN reserve selection algorithm (Ball and Possingham 2000). Planning unit status was assigned as either “Conserved” (the PCTL derived polygons), “Excluded” (planning units identified by the FMMP dataset as greater than 50% urban), or “Available” (all other planning units). The “cost” value in MARXAN was designated simply as the area of the planning unit.

A unit-less “boundary modifier” of 1,500 was selected after test runs showed that this parameter value led to what we considered to be a pattern of reserve clustering that balanced the desires for large, compact reserves and spatial dispersion of reserves across the study area. We conducted 100 runs with 10 billion iterations each. We selected these model run input values after conducting several test runs in order to achieve what we believe is a reasonable balance between output “optimality” and computing resource availability (because of the large number of planning units in most conservation planning efforts, MARXAN only approximates an “optimum” solution). This large number of iterations was required in order to account for the random assignment of the large number of planning units into status categories at the beginning of the MARXAN selection process.

For the large-scale regional planning effort, we identified only those reserves that were repeatedly selected by MARXAN and were part of larger blocks of existing or potential habitat rather than scattered parcels of land. Thus, we designated 30 runs as the minimum number of times out of 100 that a planning unit had to be selected as part of a MARXAN "solution" for it to be included in our identified reserve network. Further, we eliminated contiguous groupings of the selected planning units that were less than 2000 ha in size.

Connectivity analysis at the regional scale

Each of the identified core reserves was analyzed for potential use by the 8 conservation targets. We selected a mean habitat value of 0.33 within a specific reserve as the minimum for that reserve to be considered suitable for a particular focal species. This value was chosen in order to include reserves in a particular species' ecological network that contained marginal habitat but could probably be enhanced through restoration efforts over a long time frame. A connectivity analyses was then conducted individually for each of the 5 mobile terrestrial focal species between the cores that were selected for that species. We did not model connectivity for Swainson's hawk, yellow-billed cuckoo, and the vernal pools, because of the aerial mobility of the birds and non-movement of the vernal pools.

Connectivity analyses were focused on areas between adjacent core reserves identified as being potential habitat for each particular focal species. Not all combinations of identified adjacent reserves were analyzed for any one focal species. For example, we did not conduct connectivity analysis for giant garter snake between adjacent core reserves if only one of those reserves was a wetlands-focused reserve and the other was uplands not likely to be utilized as habitat by this species.

The connectivity analysis was performed using a modified version of the Least Cost Corridor ArcGIS function. This version, CorridorCreator (Gallo 2007), allows for analysis between multiple reserves simultaneously thereby reducing computing time. The cost surface used in these analyses was simply the additive inverse of the habitat value dataset (e.g. high habitat values became low cost values) for each focal species. This analysis produces a "connectivity surface" between adjacent reserves. In order to identify potential corridors from these surfaces, we selected those raster cells with a value of ≤2% of the value of the lowest cost raster cell and converted these to a polygon shapefile.

County Analysis—a locally-based approach for reserve selection and connectivity

We replicated the regionally-based methods to run independently in five counties (out of the 29 comprising the Central Valley) within the Central Valley: Sutter, Yolo, Stanislaus, Fresno, and Kern (listed north to south; Table 3.1). These counties were selected to provide for a wide range of local conditions within the ecoregion. Sutter County was selected because it is the only county wholly within the ecoregion; all others have portions of their area in the various surrounding mountainous ecoregions. Only those portions of the counties falling within the Central Valley ecoregion boundary were considered in these analyses.

Table 3.1. Extents of cores ("Cores") and corridors ("Corr.") in the five analysis counties. The "Region" network refers to the section of the regional network located within each county while "County" refers to the network identified at the local scale. The "%" columns refer to the portion of each county's Central Valley ecoregion area (rather than the full county extent) comprised by the conservation network components.

		Cores		Corridors			
County	**Network**	**Area (ha)**	**(%)**	**Area (ha)**	**(%)**	**Total (ha)**	**Total (%)**
Sutter	Region	9,641.0	6.1	19,494.9	12.4	29,135.9	18.5
	County	18,993.3	12.1	7,361.9	4.7	26,355.2	16.7
Yolo	Region	18,508.4	9.2	67,491.7	33.4	86,000.1	42.6
	County	22,568.0	11.2	42,156.5	20.9	64,724.5	32.0
Stanislaus	Region	37,157.5	12.3	93,849.2	31.0	131,006.7	43.3
	County	28,578.5	9.4	116,055.5	38.4	144,634.0	47.8
Fresno	Region	55,528.0	7.0	116,584.7	14.6	172,112.7	21.6
	County	61,893.2	7.8	45,829.5	5.8	107,722.7	13.5
Kern	Region	81,907.7	8.9	223,048.1	24.4	304,955.8	33.3
	County	77,257.6	8.4	206,790.6	22.6	284,048.2	31.0

The same focal elements were used as conservation targets in these analyses although for some counties not all of the focal species historically occurred there (e.g. the San Joaquin kit fox did not occur in Sutter County) and so were excluded from consideration in those respective counties. Because we considered the regional minimum reserve area (2000 ha) to be too large for a county-based approach, we scaled the county-based reserves to a percentage of the area of each county. Specifically, the minimum reserve size for both the ecoregion and the five counties was equal to 0.034% (determined by the original selection of 2000 ha as the minimum core reserve area in the ecoregional analysis) of the total area of that county-specific analysis area.

The county-specific corridors were identified in the same manner as with the regional approach. However, the county boundary was used as an analysis mask so that the full corridor area had to fall within the county.

Overlap Analysis

An overlap analysis was conducted between the results from the two spatial scales of analysis to identify differences between them. The county boundaries were used to extract the results from the ecoregional analysis for the five counties that were individually analyzed. For each county, the results from the ecoregional analysis were overlaid on the county-specific results. The sets were unioned together and the area of the unioned sets was calculated. Finally the total area of each of the unioned sets was broken out into three categories: (1) occurs in both networks, (2) only found in the regional network, or (3) only found in the countywide network. In addition, an overlap analysis was conducted solely on the reserves and likewise solely on the corridors for each of the five counties.

In addition, the number of network components (i.e. core reserves and corridors) that overlapped between scales was calculated for each county. The number and proportion of cores at each scale for each county that overlapped was calculated. The same analysis was performed for corridors. The number of network components that overlapped both across scale and type was also calculated.

Focal element coverage

The effect of scale on total coverage of the focal elements was measured through comparison of habitat value for each represented by the core reserves identified at the two scales of analysis. For each core reserve identified at both spatial scales within the five analysis counties, we calculated a habitat value area for each focal element by multiplying the reserve area by the average value of the raster cells representing focal element habitat suitability that fell within the reserve boundary. Within each county, the ratio of total habitat area within core reserves identified at the local scale versus that identified within regional-scale reserves was calculated. This value was then scaled to run from -1 to +1, with scores of -1 indicating habitat for that focal element was only found in regional-scale cores, +1 indicating only within local-scale reserves, and 0 implying equal habitat value areas between the scales. Mean values for both counties and focal elements was calculated. Finally, in order to measure the magnitude of this scale-driven variation of habitat value coverage, “absolute mean difference” (AMD) was calculated by taking the mean of the absolute value of the scaled (-1 to +1) habitat value ratios, leading to a value ranging from 0 to 1.

RESULTS

Regionally-based Analysis

The MARXAN reserve selection analysis scored each planning unit, with values ranging from 0 (never selected in 100 runs) to 100 (selected in every run) (Figure 3.3). Fifty-two core reserves within the Central Valley ecoregion were delineated (Figure 3.4). The spatial distribution of the core reserves was relatively uniform throughout the region, with the exception of several large areas in the southeast section. Core areas were also found in both the low lying valley floor and the foothill perimeter of the Central Valley. The core reserve areas ranged in size from 2,061 ha to 116,527 ha and covered 12.2% of the total area of the ecoregion.

The connectivity analysis identified 388 species-specific corridors linking the core areas (bobcat = 120, giant garter snake = 27, kit fox = 50, yellow-billed cuckoo, tule elk = 121) (Figure. 3.4). However, because many of these corridors overlapped, there were substantially less than 388 discrete identified units (it is difficult to provide an exact number as the overlap results in corridors linking multiple core reserve areas being unioned into a single polygon; it is unclear what it would mean to consider this as one discrete corridor). These corridors totaled 18.2% of the study region area.

The total area identified as cores in the regional network within each of the five analysis counties (Table 3.1) ranged from 6.1% (Sutter County) to 12.3% (Stanislaus

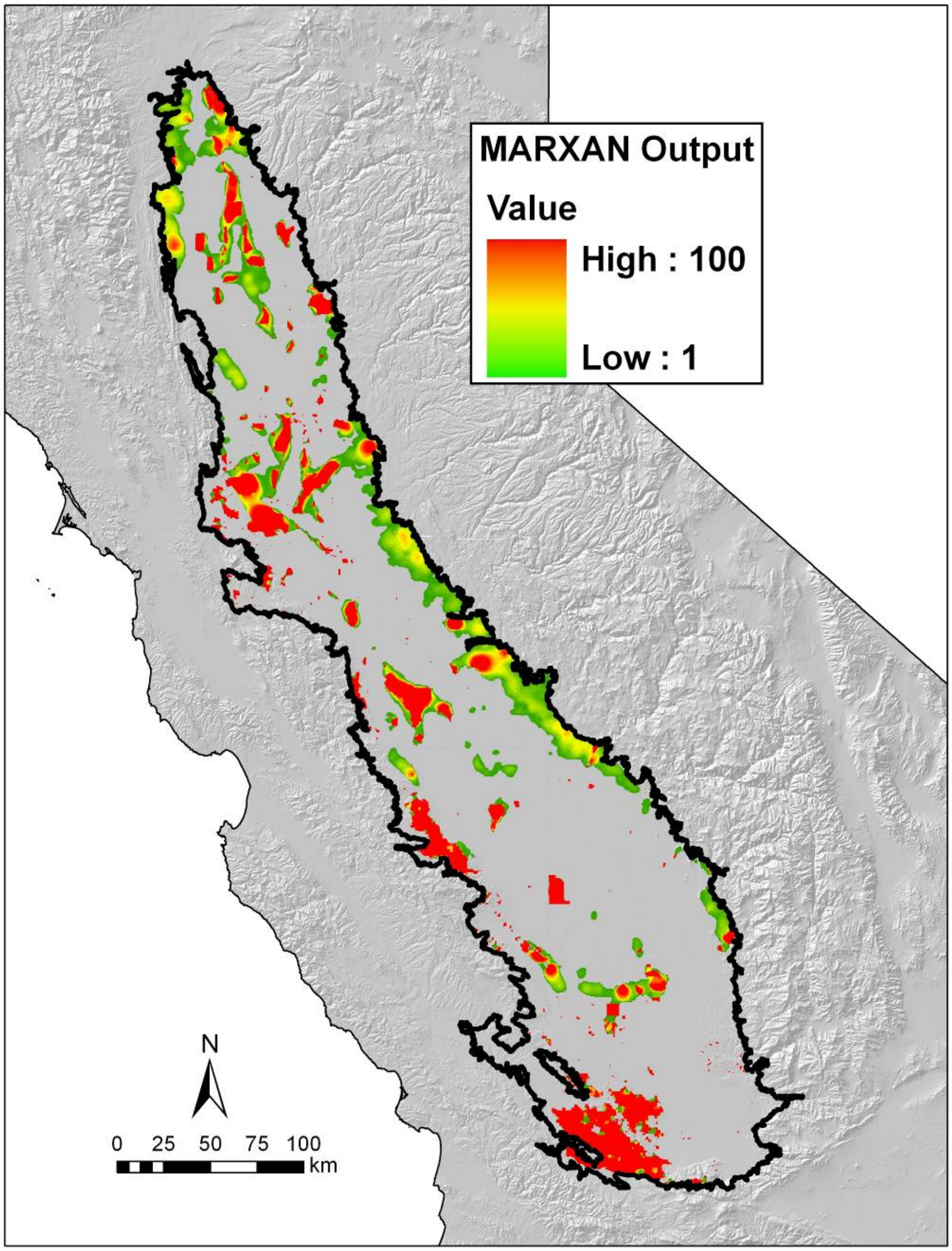


Figure 3.3. Results of the MARXAN reserve selection analysis. Planning units in red (Value = “High”) were selected in many of the MARXAN runs while those in green (Value = “Low”) were selected in few. Planning units in gray were never selected.

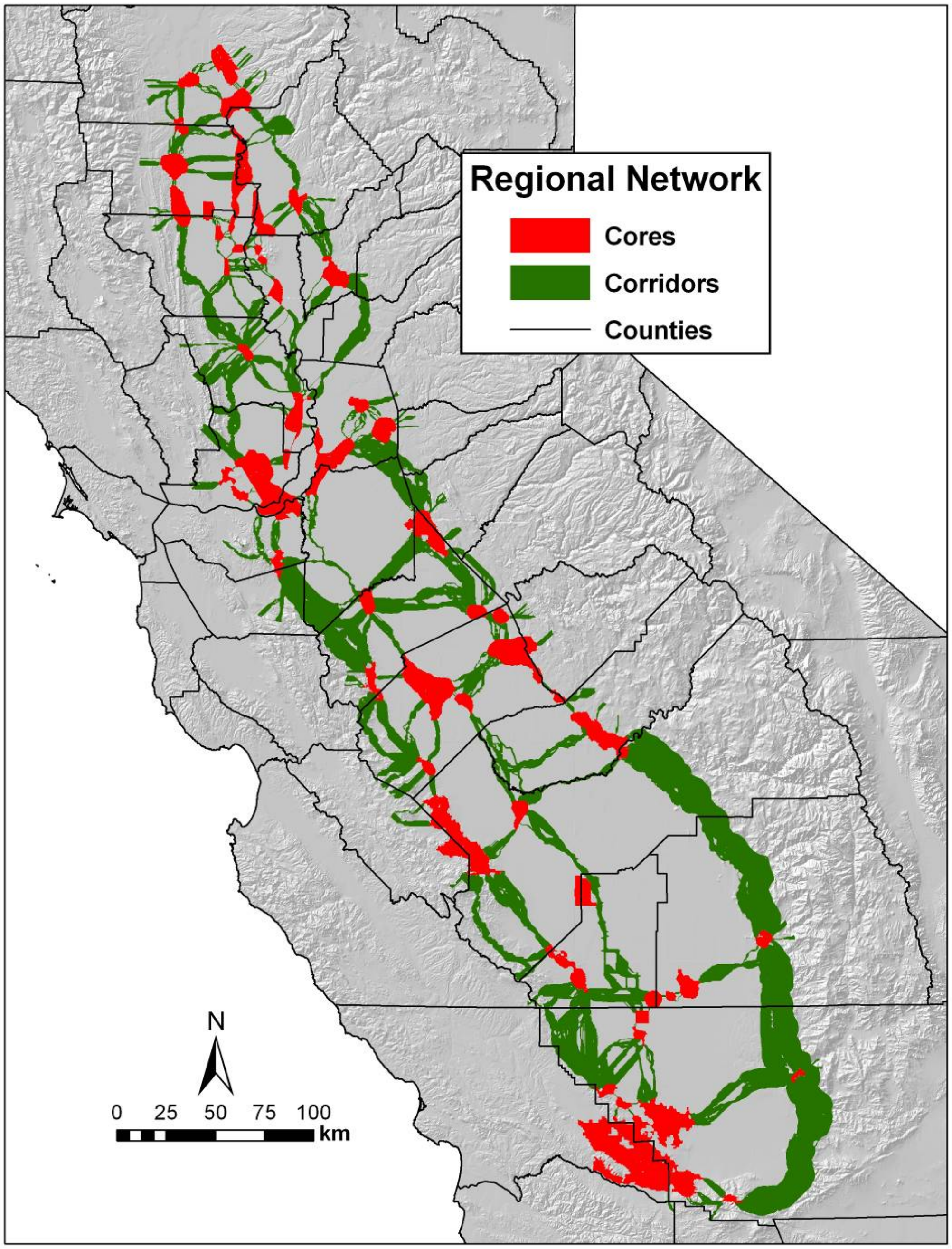


Figure 3.4. A potential Central Valley ecoregional conservation network. The network includes cores comprised of clusters of planning units each of which was part of at least 30 run solutions and collectively measured at least 2000 ha in area. The corridors were identified for all five focal species and include all pixels within 1.02x the value of those found on the true lowest cost path.

County) of the total county area located inside the ecoregional boundary (recall that Sutter County is the only one of the five located wholly within the ecoregion; Table 3.2). The area occupied by identified regional corridors ranged from 12.4% (Sutter County) to 33.4% (Yolo County). Footprints of the entire network ranged from 18.5% (Sutter County) to 43.3% (Stanislaus County).

Locally-based Analysis

The number of core area reserves identified during the local scale analyses ranged from 7 to 18 (Sutter County and Kern County, respectively) and from a minimum area of 61.8 ha (Sutter County) to a maximum area of 43,341.5 ha (Fresno County) (Figure 3.5). The number of species-specific corridors identified in these analyses ranged from 27 (Sutter County) to 169 (Kern County).

The total area identified as core reserves in the local networks ranged from 7.8% (Fresno County) to 12.1% (Sutter County) of the total county area located inside the ecoregional boundary. Likewise, the area occupied by local corridors ranged from 4.7% (Sutter County) to 38.4% (Stanislaus County). Footprints of the entire network ranged from 13.5% to 47.8% (Fresno County and Stanislaus County, respectively) of the total county area located inside the ecoregional boundary.

The mean percent of total county area covered by core reserves in the local networks was 9.8%, slightly more than in the regional network which was 8.7%. The reverse was true with the identified corridors: the local mean was 18.5% and the regional mean was 23.2%. The overall county mean was 28.2% for the local networks and 31.9% for the regional networks.

Table 3.2. Area (ha) of the five analysis counties and the proportion (%) of the counties that falls within the Central Valley ecoregion boundary.

County	Total Area (ha)	Ecoregion Area (ha)	% of County
Sutter	157,599	157,599	100.0
Yolo	264,442	202,062	76.4
Stanislaus	392,615	302,569	77.1
Fresno	1,558,286	797,258	51.2
Kern	2,113,040	916,010	43.4

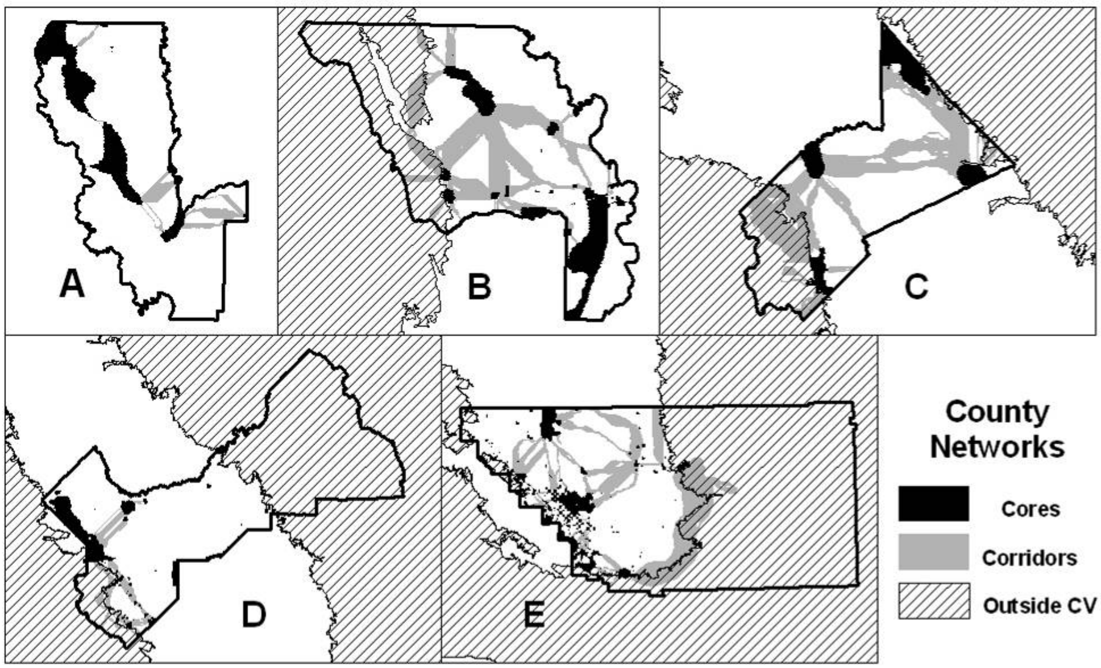


Figure 3.5. Results from the analyses run for each of five individual counties in the Central Valley ecoregion (A – Sutter; B – Yolo; C – Stanislaus; D – Fresno; E – Kern). Cores and corridors were determined in the same way as for the ecoregion, however minimum core areas for each were calculated as an identical ratio as that for the larger region. Thus each county had different minimum size cores. Corridors were calculated between adjacent cores as well as to the county boundary from outer cores.

Overlap Analysis between Regionally- and Locally-based Network Designs

The mean percent overlap (i.e. spatial congruence) between the two conservation network scales (Figure 3.6) within the five analysis counties was 35.9% (Table 3.3; Figure 3.7). Of the remaining area of the unioned regional and local conservation networks, a mean of 41.0% was identified only through the regional analysis and 23.2% through the local analysis. The county with the greatest overlap was Kern (the southernmost of the five) with a 49.9% overlap while that with the least was Stanislaus (in the middle position of the five) with a 23.4% overlap between the networks.

The county with the greatest overlap of cores was Fresno (72.4%) and that with the least was Sutter (41.4%). The mean core overlap for the five counties was 54.9%, with 15.3% the mean for each county identified as a regional core only and 29.8% as county core only.

The county with the greatest overlap of identified corridors was Kern (44.8%) and the least was Sutter (5.1%). The mean corridor overlap was 21.4%, substantially less than the mean core overlap. The mean corridor area identified during the regional analysis only was 55.2% of the total area of the unioned corridors while the mean county corridor area was 23.5% of the total corridor area.

The greatest overlap of network components occurred for regionally-based core reserves, of which 84.5% demonstrated at least partial overlap with locally-based cores (Table 3.4; Figure 3.8). All of the regional core reserves in both Sutter and Fresno Counties overlapped local cores. Other component types experienced lower overlap scores, ranging from 55.2% (local cores) to 43.8% (regional corridors). The single lowest overlap rate was 16.7%, with only 2 of 12 Sutter County regional corridors overlapping

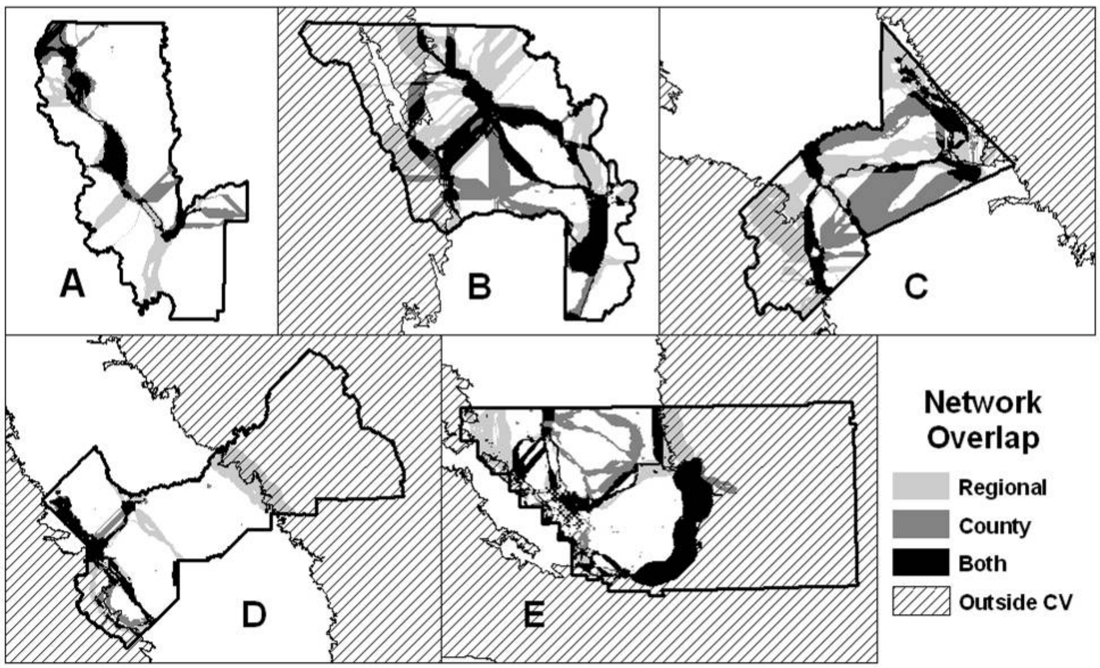


Figure 3.6. Results of the overlap analysis for each of the five individual counties. The ecological networks from the analyses for individual counties were overlaid on the portions of the regional network found within each county. The colors depict areas of overlap (black), areas that are only identified for the individual county (medium gray), and those found only in the regional network (light gray). Striping indicates areas outside the study area boundary.

Table 3.3. Percent overlap in each of the five analysis counties between the local and regional conservation networks. Shown are overlaps in identified cores, corridors, and overall network. Also shown are the means of each of these between the five counties. "Local" is the percent of the unioned regional and local networks only found in the local network, "Regional" in the regional network, and "Overlap" in both networks.

County	Network Element	Local %	Regional %	Overlap %
Sutter	core	52.6	6.0	41.4
	corridor	23.7	71.2	5.1
	both	28.7	35.2	36.2
Yolo	core	32.4	12.2	55.5
	corridor	20.8	51.4	27.8
	both	19.9	41.0	39.2
Stanislaus	core	17.2	35.6	47.2
	corridor	38.1	46.6	15.3
	both	33.1	43.5	23.4
Fresno	core	22.2	5.4	72.4
	corridor	13.2	72.9	13.9
	both	13.3	56.1	30.6
Kern	core	24.7	17.2	58.2
	corridor	21.5	33.8	44.8
	both	20.7	29.3	49.9
All (mean)	core	29.8	15.3	54.9
	corridor	23.5	55.2	21.4
	both	23.2	41.0	35.9

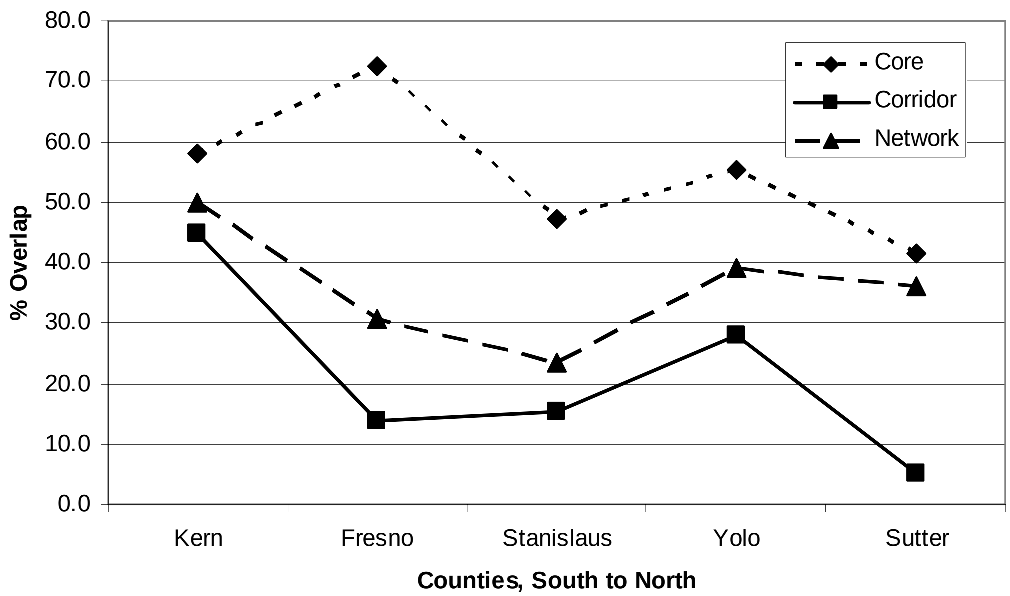
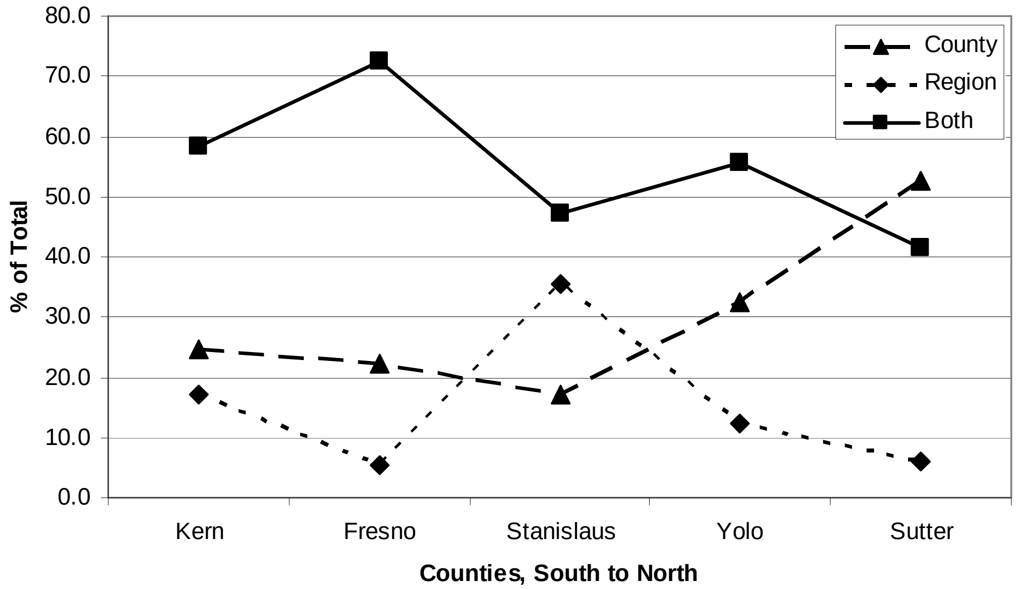


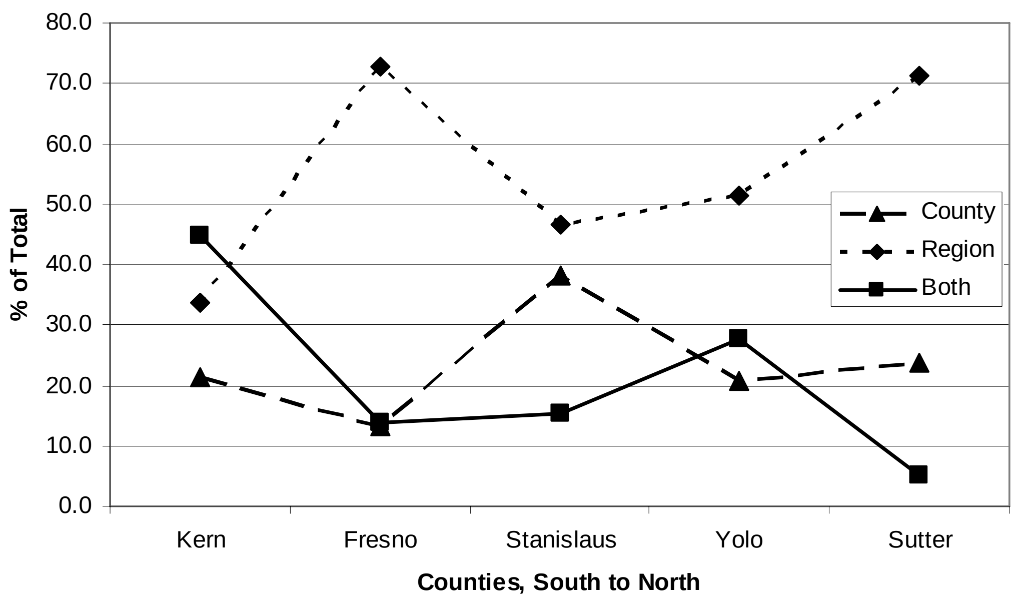
Figure 3.7. The percent of network overlap in each county. The large dashed line indicates the percent of overall network overlap, while the other lines represent the percent overlap of the cores (small dashes) and corridors (solid) individually.

Table 3.4. Overlap between network components. The number of cores and corridors at each spatial scale (regional and local) that overlapped with the same component types at the other scale was calculated. The rate of overlap (%) was then calculated for each component and scale type.

County	Regional cores	Core overlap, region	Core overlap, region (%)	Local cores	Core overlap, local	Core overlap, local (%)	Regional corridors	Corridor overlap, region	Corridor overlap, region (%)	Local corridors	Corridor overlap, local	Corridor overlap, local (%)
Sutter	4	4	100.0	7	3	42.9	12	2	16.7	9	5	55.6
Yolo	3	2	66.7	11	2	18.2	14	8	57.1	21	13	61.9
Stanislaus	6	5	83.3	13	7	53.8	10	4	40.0	30	15	50.0
Fresno	5	5	100.0	9	5	55.6	10	5	50.0	17	11	64.7
Kern	11	8	72.7	23	14	60.9	20	11	55.0	51	18	35.3
Mean			84.5			55.2			43.8			53.5



(a) Cores



(b) Corridors

Figure 3.8. The percent overlap of (a) cores and (b) corridors. Solid lines represent the percent overlap when the local (county) and regional cores and corridors (respectively) are overlaid. Cores or corridors identified only in the local networks are represented by large dashes and those found in just the regional network by small dashes.

local corridors. In addition, a mean of 1.8 regional cores overlapped local corridors per county while 5.6 local cores overlapped regional corridors (Table 3.5).

Focal element coverage

Two of the five analysis counties (Sutter and Yolo) had higher habitat value areas in identified local core reserves for all seven focal elements (San Joaquin kit fox was not analyzed for these counties because they fall outside its historic range) (Table 3.6). All vernal pools included in cores in these counties (as well as Kern County) were found in local-based reserves (scaled ratio score = 1.0). Sutter County focal elements had a mean scaled ratio score and AMD of 0.44, the largest of the five counties, while Yolo County had scores of 0.26. Conversely, in Stanislaus County, higher habitat value areas were included in cores for all focal elements except for yellow-billed cuckoo. The effect was less pronounced than for Sutter and Yolo Counties however, with a mean scaled ratio score of -.13 and AMD of 0.16. The only other county with a negative scaled ratio score (indicating more habitat value area found in regional core reserves than local) was Fresno County, although only three focal elements received negative ratio scores (with the other five focal elements having slightly positive ratio scores). Fresno County also had the lowest AMD (0.07) indicating a nearly equal distribution of habitat value areas in regional and local cores.

Five of the eight focal elements had mean scaled ratio scores between 0.06 and 0.08 (Table 3.6). Vernal pools had the highest mean ratio score (0.54), indicating that in the five analysis counties, roughly three times the total vernal pool area was included in

Table 3.5. Overlap between different component and scale types in each of the five analysis counties. Identified here is the number of instances of each of these two kinds of overlaps.

County	Regional core - Local corridor	Local core - Regional corridor
Sutter	1	4
Yolo	1	9
Stanislaus	5	7
Fresno	0	3
Kern	2	5
Mean	1.8	5.6

Table 3.6. Scaled habitat value area ratios for each focal element in the identified core reserves in each analysis county. The scale ranges from -1 to +1, with positive values indicating more habitat value area in locally-based core reserves and negative values indicating more habitat value area in regional cores. "AMD" (Absolute Mean Difference) is the calculated mean of the absolute scaled ratios, thus ranging from 0 to +1.

County	Tule elk	Bobcat	Pronghorn	Giant garter snake	San Joaquin kit fox	Swainson's hawk	Yellow-billed cuckoo	Vernal pools	Mean	AMD
Sutter	0.36	0.35	0.32	0.35	n/a	0.32	0.38	1.00	0.44	0.44
Yolo	0.13	0.14	0.17	0.12	n/a	0.17	0.08	1.00	0.26	0.26
Stanislaus	-0.15	-0.16	-0.21	-0.14	-0.18	-0.16	0.10	-0.15	-0.13	0.16
Fresno	0.02	0.03	0.04	-0.18	0.04	0.02	-0.09	-0.16	-0.04	0.07
Kern	-0.03	-0.04	-0.02	0.29	-0.04	0.03	0.20	1.00	0.18	0.21
Mean	0.07	0.07	0.06	0.09	-0.06	0.08	0.13	0.54		
AMD	0.14	0.14	0.15	0.22	0.09	0.14	0.17	0.66		

local core reserves than in regional cores. Of the eight focal elements, only the San Joaquin kit fox had a negative mean ratio score (-0.06), indicating greater inclusion of kit fox habitat at the regional scale than at the local scale. AMD ranged from 0.14 to 0.17 for five of the focal elements while vernal pools had the highest AMD (0.66).

DISCUSSION

From the results of the analyses it is clear that our hypothesis was correct: the two spatial scales of analysis identified substantially different conservation networks within the respective study areas. On average, just over one third of the area identified for inclusion within a conservation network at either scale was identified at both scales. In addition, with the exception of regionally-based cores, only roughly half of the network components overlapped at all with their other-scaled counterparts (Table 3.4).

Especially significant is the lack of congruence between corridors identified at the different spatial scales. This disparity suggests several things. First, many regionally important corridors are not identified at the local scale. As noted above, on average only 43.8% of regional corridors overlapped at all with locally-based corridors. Regional and inter-regional connectivity could be threatened if planning only occurs at this local scale. Second, corridors connecting locally important core areas can be missed if only the regional scale is taken into account in the planning process. Again, only 53.5% of local corridors overlapped with those identified at the regional scale. This could lead to isolation of locally important core areas.

These findings lead to the conclusion that planning at the local scale and amalgamating these efforts in lieu of formal regional planning process can lead to the overlooking of important regional conservation needs. Without referring to the regional context, there is little way to account for connectivity between areas separated by an intervening administrative unit (e.g. a county) or even between adjacent administrative units. While our county analysis identified corridors leading from cores to the county boundaries, there was little indication that these corridors would link up to anything deemed ecologically important on the other side of a county's boundary. Not only were important regional corridors not identified, but there was also a potentially fractured network across the region with subunits that do not integrate. We feel that resolution of these "myopic effects" cannot be addressed at the local scale alone and requires explicit regional integration.

The results of this study also suggest the converse to be true: a conservation plan that only addresses regional needs can miss areas of local ecological or cultural importance. There are several reasons why this is an important issue for conservation planning. First, if the components of a conservation network are not stratified across subunits within a region, fine-scale ecological variation may not be adequately represented or protected. The Central Valley in California is approximately 650 km in length; intra-taxon populations of many species may harbor important genetic diversity across that distance (Patton and Yang 1977). For example, Table 6 indicates that no vernal pool habitat was included in regional-based core reserves in three of the analysis counties. A heavier emphasis was placed on conservation of vernal pools in Stanislaus and Fresno Counties. Neglect of this focal element in several counties could mean loss of ecological variation if these pool complexes are lost to conversion. Second, many ecological conservation planning efforts are intertwined with city and county efforts to provide open space for their citizens. If a purely regional approach to conservation planning is adopted, many areas might not be deemed biologically valuable enough to warrant expenditure of finite conservation resources. This could become a social justice issue to local inhabitants if few resources are devoted to open space and recreation in locations accessible to them (Talen 1998). Sutter County is an example (Table 6) of an area that had substantially larger amounts of habitat identified for conservation at a local scale than at a regional scale. Finally, areas that are disproportionately targeted for inclusion in a regional conservation network might place an economic burden on the inhabitants (King and Anderson 2004). If one county in the region is a focus of conservation activity that reduces or eliminates property tax revenue as well as potential economic activity on that land, it will potentially be seen as unfairly assuming the financial burden of the conservation network for the larger region. Stanislaus County (and to a lesser extent Fresno County) is potentially an example (Table 6) of this phenomenon. Regional conservation analysis points to more core reserve habitat area in the county than does the local-scale analysis.

This study also reveals that even if there is considerable overlap of conservation networks between a county and a region, the designation as either core or corridor might differ. Looking at the results from Sutter County (Table 3.3), we see that while there is 36.2% overlap in the conservation networks (essentially the mean across counties, 35.9%), there is only a 5.1% overlap between identified corridors. Further, cores identified only at the regional scale account for just 6.0% of the combined local/regional core area. This indicates that much of the regional conservation network within Sutter County consists of corridors linking cores in adjacent counties, and many of those corridor linkages happen to coincide with areas identified as cores at the local scale (in Sutter County). Moreover, land management might differ for cores and corridors (Soulé and Terborgh 1999); thus with land being identified as important core reserve at one scale and corridor at another scale, managers would have two different mandates for these areas within a combined network. However, an alternative way to potentially view this phenomenon is that areas identified as cores at the local scale and corridors at the regional scale could serve as "stepping stones" in a regional linkage network. Table 3.5 shows a mean of 5.6 instances of this sort of overlap per analysis county.

In addition to these spatial scale effects, the overall footprints within the individual counties also changed as we moved between scales. On average, our modeled regional conservation networks accounted for an additional 3.7% of each county as compared with the local networks. This difference was largely driven by the greater extent of corridors called for at the regional scale. Thus, not only were potential corridors identified at different locations, but the regional corridors covered a larger area.

While the analysis in this paper identified that changes in spatial scale can have dramatic effects on design of a conservation network, some caveats should be noted. The first is that we only looked at two possible scales of analysis, county and ecoregion. However, given the results of this study, there are likely to be cross-ecoregional boundary effects as well. For instance, the identification of an effective and sustainable conservation network for the entire state of California would require integrative planning between each respective ecoregion as well as between individual counties and individual ecoregions. This would be an important future study to conduct. Such a study might reveal crucial corridors in the Central Valley ecoregion that were undetected by our analysis that connect to the Sierra Nevada ecoregion or to the Coast Range ecoregions in California. Additionally, we only used one of many potential techniques for identification of a conservation network. We focused on focal species habitat needs (including restorable areas); we did not include approaches such as representation or irreplaceability (sensu Margules and Pressey 2000). These other techniques, either as stand alone analyses or in conjunction with a focal species analysis, might yield conservation networks more resilient to change at different spatial scales.

A final caveat that points to a future direction for this research lies in the use of the same 13.3 ha hexagonal planning units in both the regional and local scales of analysis. Planning units of this size might be too coarse for the more fine-grained county-specific conservation planning efforts. A sensitivity analysis should in the future be undertaken to understand what effect, if any, results from changing the size of these planning units to potentially better fit the local spatial scale.

The overlap analysis that we present is but one means by which the scale differences can begin to be detected and resolved. Areas that appear in the identified conservation networks at both scales, especially those that retain their component designation (i.e. core or corridor), could serve as the foundation for a network that incorporates the conservation needs of multiple scales. Once these areas have been accepted as components of a network, decisions on inclusion of other areas to complement them can be decided on an individual basis, but with a greater understanding of the ecological role they might play in the overall network.

CONCLUSIONS

Conservation planning efforts should simultaneously occur in top-down (e.g. regional) and bottom-up (e.g. local) fashion. Planning at only one spatial scale does not necessarily lead to results that address ecological needs at other scales. Single-scale assessment and planning designs potentially omit important core areas and connections, and thus may undermine basic conservation goals. The key question that will need to be addressed on a case-by-case basis by those taking part in a conservation planning process is: what level of spatial stratification (i.e. scale selection) of conservation effort is appropriate in the given context? Successfully answering this question can lead to a more robust plan that addresses a variety of ecological processes and biological concerns. The ecological and legal/policy realities of the current planning environment in the USA require that local implementation of conservation efforts be conducted within a regional context for greater effectiveness. In this study we have demonstrated an emergent property of multi-scale planning and that the whole is, in most cases, greater than its parts.

CHAPTER 4

ASSESSING THE ECOLOGICAL CONDITION AND VULNERABILITY OF A POTENTIAL CONSERVATION NETWORK IN THE SAN JOAQUIN VALLEY WORKING LANDSCAPE

INTRODUCTION

Regional conservation planning for working landscapes, such as agricultural or timber-producing regions, is needed because these regions globally occupy over 50% of terrestrial ecosystems (Tilman et al. 2001). Regional conservation planning frequently consists of identifying an ecological network composed of core habitats ("cores") and linkages (Carroll et al. 2003, Noss et al. 1996) and assessing the threats to those components, such as the impacts of current human activities or of future climate change (Cowling et al. 2003). However, rarely do regional analyses incorporate restoration needs or projected future human impacts in the targeted ecological network. Inclusion of these elements can contribute to better design for long-term ecological sustainability and biodiversity conservation. This is particularly important for working landscapes (Polasky et al. 2005), which are usually (but not always) less impacted than fully urbanized regions but are generally more degraded than publicly-managed regions.

Regional conservation planning requires consideration of a number of key themes including the selection of important potential conservation areas based on biological features (e.g. sensitive or focal species and vegetation communities) and ecosystem processes (Chan et al. 2006), and the level of connectivity of the landscape that enables the flow of species and ecosystem processes between identified areas (Groves 2003). Biological features that might be included in the reserve selection process include: biodiversity hotspots (Myers et al. 2000), habitat for threatened or endangered species (Groves 2003), locations of endemic species (Caldecott et al. 1996), and roadless areas

(Soulé and Terborgh 1999); while ecological processes such as flooding and fire that give rise to and sustain the biological components also need to be considered.

Another ecological process vital to the continuing existence and function of ecosystems is the flow of ecological components such as nutrients as well as of individuals of a wide variety of species (Bennett 2003), which is enabled by landscape connectivity. As natural areas become increasingly fragmented, the role of connectivity between isolated core areas takes on ever greater significance for the continued viability of animal and plant populations. Connectivity has long been recommended for inclusion in landscape conservation designs (Noss and Daly 2006, Noss and Cooperrider 1994), which has led to efforts to devise effective means of measuring connectivity in landscapes. Approaches to quantifying connectivity have included: nearest neighbor distance (Moilanen and Nieminen 2002), spatial pattern indices such as patch cohesion (Schumaker 1996), and graph theory (Urban and Keitt 2001). While these metrics each describe some aspect of the connectivity of a given landscape, none addresses a measure of potential restorable connectivity, a feature particularly important in working landscapes. Restorable connectivity refers to linkages between core areas that have been eliminated by human land use patterns but are not precluded from future ecological viability by urbanization or other more permanent human infrastructure.

An effective ecological network evaluation can identify network components that are likely to be fully functional, those that are impaired, and those that have been degraded to the point of impassability. On working landscapes these categories can provide guidance for conservation acquisition priorities, and restoration opportunities. We modeled an ecological network across an intensively used working landscape and assessed its functionality by evaluating the current ecological condition of its cores and linkages. Ecological conditions ranged from fully functional (commonly used by wildlife for movement across the landscape) to non-functional (impassable to wildlife movement).

Modeling an ecological reserve network at the landscape scale also permits the prioritization of its components for conservation management, generally guided by two variables: irreplaceability (Noss et al. 2002, Margules and Pressey 2000) and threat (Rouget et al. 2003). Irreplaceability refers to uniqueness of the ecological features of a site (i.e. those found in few or no other locations; Noss et al. 2002) or its species richness (i.e. areas supporting large numbers of species including those found in many locations; Shriner et al. 2006). Threat refers to the probability of degradation or loss of the site by human activity such as: conversion of natural land to urban (Bierwagen 2007) or agricultural uses (Rouget et al. 2003), resource exploitation (Neke and Du Plessis 2004), or climate change (Pearson and Dawson 2005). We quantified the current ecological condition of the modeled network, and assessed the long-term future threats to it from urban growth, a common concern in working landscapes.

Development of methods to assess the expected magnitude of urban growth impacts is needed for conservation planning (Peterson et al. 2003). We conducted a threat assessment by using spatially-explicit urban growth scenarios that project the location of future urban growth and permit the simulation of different land use policy scenarios. This approach is potentially useful because it can allow for the introduction of ecological network designs into the public planning process.

This paper evaluates the current and projected future ecological condition of the connectivity of a modeled ecological network. We identified a network of core areas with concentrations of important ecological features and existing or restorable linkages between these cores. Each linkage was classified according to its current ecological condition to identify its conservation protection and restoration needs. A vulnerability analysis of the ecological network was then conducted to evaluate the future anticipated threat to each network component under seven potential urban growth scenarios, representative of different policy domains.

METHODS

Study area

We used the San Joaquin Valley of California, USA, as the area of analysis for the ecological network model. This large (43,000 km^2) valley has been largely converted to agricultural uses from an original mosaic of grasslands, freshwater wetlands, riparian forest, and oak woodlands (Ricketts et al. 1999). It is currently undergoing urbanization with the population expected to grow from 3.3 million to almost 8 million by 2050 (California Department of Finance 2004).

Core areas identification

In 2006, several meetings were convened in which representatives from state and federal agencies (specifically land use and regulatory), academic institutions, and non-profit organizations identified important ecological features of the study area. The participants of these meetings were either already involved in the ongoing Partnership process (SJV Partnership 2006) or were suggested by other participants. All identified features that were deemed important to any organization were selected for inclusion in the analysis. Fourteen key ecological features were thus selected (Table 4.1).

Conservation opportunity areas (COA) were defined as those regions most suitable as targets for future conservation management. They were delineated by identifying locations of overlap (or "hotspots") of the key ecological features. Maps of these features were entered into a geographical information system (GIS) (ESRI 2005), and converted from vector to raster datasets at a spatial resolution of 100m cells (1 ha). All ecological features were given the same weighting in the overlap analysis and most were treated as binary data. The two exceptions to this were conservation area buffers and sensitive species richness. Buffers around existing conservation areas, selected as important conservation targets for their ability to ameliorate edge effects, were assigned a value of 1.0 for the first 0.0-1.61 km distance from a feature, and 0.5 for buffers additionally extending 1.61-3.22 km. Raster range maps of 16 sensitive species in the region were summed to provide a sensitive species richness value. This value ranged from 0 to 16 (the maximum number of sensitive species present in a given location) but was then normalized to a 0-1.0 scale. These layers (ecological features, buffers, and sensitive species richness) were summed to give each raster cell a value ranging from 0 to 14. We then demarcated COA boundaries by hand which were then finalized by the participants.

We classed the COAs into one of three ecological categories. Riparian COAs were identified if they had a linear shape and contained predominantly current or historic

Table 4.1. Important ecological features of the San Joaquin Valley as identified by SJV Partnership participant organizations. "Reference" indicates the dataset(s) used to model these features.

Ecological feature	Reference
Desert scrub	CA GAP (1998)
Blue oak woodland	CA GAP (1998)
Sensitive ecological communities	CDFG (2006)
Grasslands Ecological Area	BuRec (2002)
Historic lake beds	ESRP (1999)
San Joaquin kit fox habitat	CA GAP (1998), CDFG (2006)
Conservation area buffers	California Resources Agency (2005)
100-year floodplain	FEMA (1996)
Riparian corridors	Teale (1998)
Perennial grassland	CA GAP (1998)
Tehachapi linkage	ESRP (2004)
Sensitive species density	CDFG (2006)
Vernal pool complexes	UWFWS (1998b)
Tulare Basin planning areas	Tulare Basin Wildlife Partners (unpublished data)

riparian vegetation. Valley floor COAs did not have linear features and contained a variety of land cover types. Upland COAs encompassed hillsides adjacent to the main valley floor.

Linkage connectivity

We used least cost path analysis (LCP) (Thorne 2006, Beier et al. 2008) to identify the linkages for the modeled ecological reserve network analyzed in this study. A linkage, as used here, is a linear feature of existing or restorable habitat that enables animal and plant movement across a landscape. LCP analysis optimizes least distance, fewest road crossings (or other barriers), and most suitable habitat. One advantage of this method is that it will always identify a linkage between designated end points, which is useful in ecological networks that require ecosystem restoration for future viability (Noss et al. 2006).

We conducted a connectivity analysis to identify a network of potential habitat linkages between the COAs by calculating a generalized animal movement cost surface. Cost here refers to the resistance that an individual animal experiences in moving across the landscape; high cost is equivalent to a high resistance to animal movement. Three general natural habitat types were identified within the study area: forest, open/shrub, and riparian/water. We created separate cost surfaces that represented movement costs for the suites of species that use each of these habitat types. To create the cost layers, vector data was converted to raster format (100 m cell size), weighted, and summed, creating a suitability surface of values from 0 to 1. Current land cover received the highest weighting in this model of potential animal movement, accounting for half of the overall suitability surface weight value (weighting scheme in Table 4.2). Weighting for the other half of the suitability surface value was calculated by aggregating 3 km radius density function values of a variety of predictors to identify a suitability value for every 100 m raster cell. Predictor variables used were: road density; waterway density (for the riparian/aquatic layer only); urban area density; and natural area density. A 100 m grid of land management status was also included in this second half calculation. These variables were selected as factors that could potentially influence animal movement through suitable land cover types. Alternatively, these non-land cover variables could indicate either areas where rare movement events might occur across unsuitable habitat or those areas most amenable to restoration of connectivity and animal movement if gaps exist in the current natural land cover. Because a high suitability score was equivalent to a low movement cost score, the final values were inverted to create a cost surface, where the highest suitability scores were 0 and the lowest were 1.

The GIS function “Gated” Least Cost Path Analysis (Gallo 2007) was used to identify potential areas of connectivity between adjacent COAs using the previously described cost surface. This function, a modified version of the Least Cost Corridor ArcGIS function (ESRI 2005) that allows the use of multiple source polygons in the analysis, was used to create connectivity surfaces between selected source polygons with grid cells ranging in value from 1.0 (found on the actual least cost path linking the polygons) to 1.2 (found on a path whose value is 1.2×[least cost]). We then extracted cells of value 1.00-1.02 and converted these surfaces to polygons in order to designate potential linkages between the COAs. Thus our polygon linkages included all the least cost paths whose connectivity value is within 2% of the single least cost path.

Table 4.2. Weighting and valuation scheme used to create the “cost surface” used in Least Cost Corridor analysis. Shown are weightings and valuations used for all three general habitat types with their associated cost surfaces.

Data Layer	Reference	Weight (Open)	Weight (Forest)	Weight (Water)	Valuation	Score (Open)	Score (Forest)	Score (Water)
Land cover	ESRP (2004)	0.5	0.5	0.5	Grassland	1.0	0.4	0.4
		0.5	0.5	0.5	Chaparral	0.7	0.7	0.5
		0.5	0.5	0.5	Coastal scrub	1.0	0.7	0.5
		0.5	0.5	0.5	Forest	0.5	1.0	0.5
		0.5	0.5	0.5	Desert scrub	1.0	0.5	0.5
		0.5	0.5	0.5	Riparian	0.5	1.0	1.0
		0.5	0.5	0.5	Wetlands	0.5	0.5	1.0
		0.5	0.5	0.5	Water	0.4	0.4	1.0
		0.5	0.5	0.5	Agriculture	0.3	0.3	0.3
		0.5	0.5	0.5	Orchards/vineyards	0.2	0.2	0.2
		0.5	0.5	0.5	Urban	0	0	0
		0.5	0.5	0.5	Other	0.3	0.3	0.3
Road density	California DFG (2002)	0.17	0.17	0.125	0.165 - 0.016(km/km²)	0.0-1.0	0.0-1.0	0.0-1.0
Waterway density	U.S. Geological Survey (1999)	0	0	0.125	0.332(km/km²)	0.0-1.0	0.0-1.0	0.0-1.0
Urban area density	FMMP (2004)	0.08	0.08	0.0625	0.0825*(1 - urban area density)	0.0-1.0	0.0-1.0	0.0-1.0
Natural area density	ESRP (2004)	0.08	0.08	0.0625	0.0825*(natural area density)	0.0-1.0	0.0-1.0	0.0-1.0
Management status	California Resources Agency (2005)	0.17	0.17	0.125	Public, private conservation	1.0	1.0	1.0

Ecological condition and linkage classification

The identified linkages displayed a wide variety of current ecological conditions. We quantified the condition of each linkage using land cover, barriers to movement, and length. The variables used were: percent agricultural land, percent riparian forest, percent other natural land cover, barrier effects (major highways and canals), and length of the linkage. These variables were used in a hierarchical clustering analysis to identify five groups of linkages containing different levels of degradation of ecological connectivity or functionality (SAS Institute 2003).

Urban growth modeling

In order to assess the impacts of future urban growth to the ecological network, we used UPlan (Johnston et al. 2003), a spatially explicit urban growth model. UPlan is raster-based and uses county and city general plans (that provide zoning information and a 20-25 year planning horizon), projected human population growth, and a series of location-specific attractors and detractors (e.g. distance to highway interchange; see Johnston et al. 2003 for details) to assign urban growth into seven development categories. There are four residential categories (high, medium, low, and very low density); two commercial (high- and low-density); and one industrial category.

We used UPlan to model seven projected urban growth scenarios across the eight county study area. These model runs were developed in response the a series of meetings with the Land Use Housing and Agriculture subcommittee of the San Joaquin Valley Partnership and represented different future zoning policies. The runs assigned future growth in a spatially-explicit manner according to each policy. The separate scenarios were: (1) “status quo”, which had no change in the current general plans; (2) “east-west connectors”, which placed an emphasis on development around cross-valley highways; (3) “compact growth”, where all future growth occurred within the current spheres of influence (the probable future service area [Governor’s Office of Planning and Research 1997]) of the incorporated cities of the region; (4) “farmland preservation”, in which all prime and statewide important farmland as determined by soil classification (FMMP 2004) was off-limits to development; (5) “exclusion zone”, that prevented virtually all urbanization between the major north-south highways of the region, Interstate 5 and Highway 99; (6) “new cities”, where much of the growth was accommodated by creation of 4 new cities in currently non-urban areas; and (7) “great cities”, with growth focused into the four existing major metropolitan areas of the region (Table 4.3).

The UPlan models were run for each scenario using population projections (California Department of Finance 2004) for each individual county in order to model the expected spatial location under each scenario of the growth anticipated to occur in that county (regardless of scenario). The resulting urban growth projections from each of the eight counties were then combined to create urbanization projections for the full study area. These seven rasters (one for each scenario) were finally converted to vector format for the impact analysis.

Table 4.3. Modeled urban growth scenarios for the San Joaquin Valley and their predicted spatial footprint.

Scenario	Description	Projected Urban Growth (km^2)
S1 - Status Quo	No change in current development patterns	4,185
S2 - East-West Connectors	Growth more focused around east-west highways	4,188
S3 - Compact Growth	Higher density growth in existing urban areas	876
S4 - Prime Agricultural Soils Protection	Prime and statewide important farmland protected	3,908
S5 - Exclusion Zone	Minimal growth between Highways 5 and 99	4,101
S6 - New Cities	Four new cities created	2,573
S7 - Great Cities	Growth focused around existing large cities	2,750

Future Ecological Condition

The final step involved overlaying the projected urban growth model outputs with the COA and linkage network to analyze the potential future ecological condition of the components of the ecological network. For the COAs, we intersected the COA and UPlan datasets and calculated the percentage of the total area of each COA that was urbanized under each of the seven scenarios. All urban categories were treated equally with no weighting based on intensity of the projected urban use of a given parcel. We calculated the mean urbanization rate for each of the three COA clusters under each scenario.

Similarly, we intersected the linkages with the UPlan model results and calculated urbanization rates for each linkage for each of the scenarios. However, placement of the projected urbanization can accentuate the effects of the general connectivity "degradation" caused by the expanded urban footprint; e.g. a very small amount of growth can effectively sever a linkage. Thus we also calculated a "chokepoint" effect. For this measurement we identified the portion of the linkage that experienced the highest degree of narrowing due to urban encroachment and using the ArcGIS Distance tool measured the percentage loss of linkage width (Figure 4.1). We excluded the "residential very low density" component of the UPlan growth scenario results from the chokepoint analysis because, unlike all other UPlan growth categories, the actual locations for residential very low are not deterministically driven, but rather randomly allotted within the designated development zone. Thus, it was not valid to state that this urban component would contribute to a specific chokepoint even while we could identify the amount of area that would convert to this type. We calculated the mean "degradation" and "chokepoint" values for each linkage and for each of the five linkage clusters.

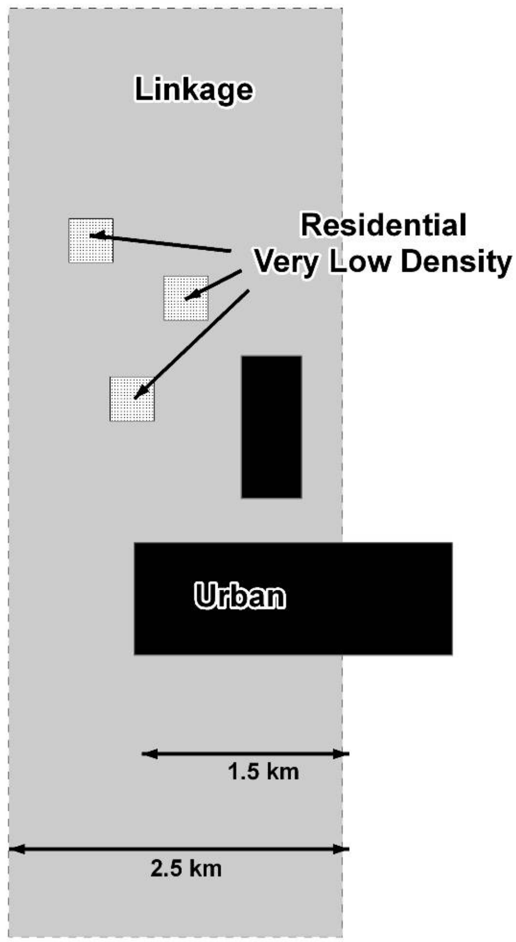


Figure 4.1. Sample "chokepoint" score calculation. In this example, the linkage is 2.5 km wide. Projected urban development will extend 1.5 km into the linkage. There are three "Residential Very Low Density" polygons anticipated within the linkage. The "chokepoint" score would then be: 1.5km/2.5km = 60%.

RESULTS

COA identification

We identified 24 COAs distributed uniformly across the study region (with the exception of one large gap in southern Fresno and northern Kings Counties) (Figure 4.2). These COAs ranged in size from 55 km² (Merced River [COA *G*, Figure 2; COAs will be denoted with brackets]) to 2504 km² (Western Kern Hills [*X*]). Of the 24 COAs, 6 were classed as riparian, 5 as valley floor, and 13 as upland (Table 4.4).

Linkage current condition

Using the calculated overall cost surface (Figure 4.2), the connectivity analysis identified 45 potential linkages between the 24 COAs (Figure 4.3). The linkages ranged in length from 1.5 km (Old River-Lower San Joaquin River (Linkage *36*, Figure 4.3; linkages will be denoted with parentheses) and Delta-Mokelumne River (*8*)) to 65 km (Mokelumne River-Vernal Pools (*34*)). They ranged from 100% natural vegetation (20 linkages) to 100% agricultural land cover (10 linkages). No urban areas were included in the linkages. Five of the adjacent COA pairs were linked by multiple potential linkages.

The hierarchical clustering analysis identified five clusters of ecological linkages (Table 4.5), ranked in descending order of probable function with natural land cover, few barriers, and short lengths assumed to be optimal for dispersing vertebrate species. The three moderately functional linkage clusters were: 1) non-riparian natural land cover, few barriers, and moderate length, 2) agricultural land cover, few barriers, and short length, and 3) non-riparian natural land cover, many barriers, and long length. The cluster that

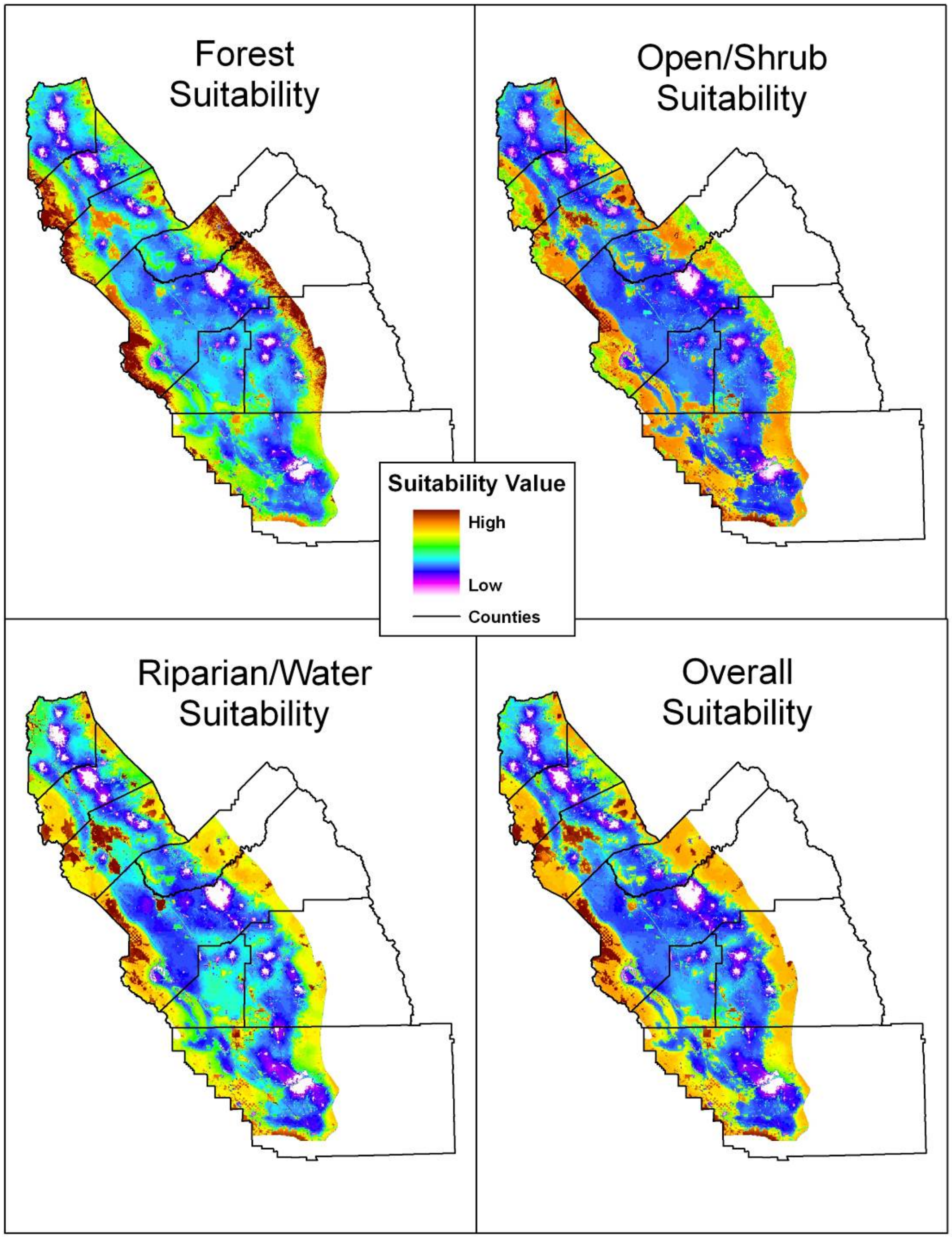


Figure 4.2. The suitability values (and thus inverse of "cost" values) for the three generalized species types in the San Joaquin Valley planning area. The "Overall Suitability" value takes the mean of the three habitat-based suitability scores.

Table 4.4. Conservation opportunity areas (COAs) identified by delineation of “hotspots” of overlap of important ecological features in the study area. They are classified into one of three ecological groups. Urbanization rates are predicted for each of seven potential growth scenarios (e.g. S1 = Scenario 1). Urbanization rates are given as percents of total area of the COA expected to be converted from non-urban to urban land use. The COAs are referred to in the text in solid brackets [ ].

COA	Name	Group	Area (km²)	S1 (%)	S2 (%)	S3 (%)	S4 (%)	S5 (%)	S6 (%)	S7 (%)
A	Delta	Valley Floor	449	0.8	1.2	0.1	0.6	1.9	0.0	5.1
B	Fresno Slough	Valley Floor	178	2.5	0.4	0.1	2.9	0.0	0.0	0.0
C	Grasslands EA	Valley Floor	1,067	6.0	6.4	0.8	9.3	0.0	2.4	0.8
D	KKT	Valley Floor	1,703	6.0	5.9	0.1	7.7	1.2	0.3	0.2
E	West Madera	Valley Floor	256	3.6	0.4	0.0	4.2	0.0	0.0	0.0
F	Lower San Joaquin River	Riparian	262	12.9	13.1	1.1	5.0	2.7	8.4	0.0
G	Merced River	Riparian	55	11.8	9.2	0.8	10.9	27.9	7.2	4.4
H	Mokelumne River	Riparian	110	18.0	19.9	0.7	4.9	16.2	10.9	12.1
I	Old River	Riparian	126	4.3	4.9	0.0	0.9	5.6	1.7	0.0
J	Stanislaus River	Riparian	57	24.0	21.0	2.0	3.8	19.6	16.1	46.0
K	Tuolumne River	Riparian	110	26.9	31.2	6.9	4.8	29.3	24.1	26.2
L	Ciervo Hills	Upland	1,436	1.0	0.9	0.0	4.3	3.5	0.0	0.0
M	Corral Hollow	Upland	107	9.4	9.4	8.6	21.8	10.0	7.4	0.0
N	Vernal Pool	Upland	784	5.8	5.5	1.7	14.4	14.6	0.7	0.6
O	Greater Henry Coe	Upland	1,105	2.0	2.3	0.6	5.3	4.5	0.7	0.0
P	Grizzly Gulch	Upland	91	4.6	0.2	0.0	11.9	6.3	0.0	0.0
Q	NE Bakersfield	Upland	582	4.8	4.9	2.5	6.8	6.9	2.6	5.4
R	Sequoia Foothills	Upland	573	3.1	3.1	0.0	17.2	4.7	1.3	0.1
S	Stokes Mountain	Upland	160	5.8	5.8	0.0	10.5	9.6	0.3	0.5
T	Tejon Hills	Upland	142	1.8	2.8	0.0	2.8	2.7	0.0	0.0
U	Upper Fresno River	Upland	362	1.8	3.1	0.0	3.1	3.6	0.0	0.0
V	Upper Kings River	Upland	202	1.0	0.7	0.0	10.4	3.8	0.0	0.0
W	Upper San Joaquin River	Upland	302	3.7	3.8	1.7	7.7	7.0	2.2	6.2
X	Western Kern Hills	Upland	2,504	2.0	2.0	0.1	3.4	2.7	0.1	0.1

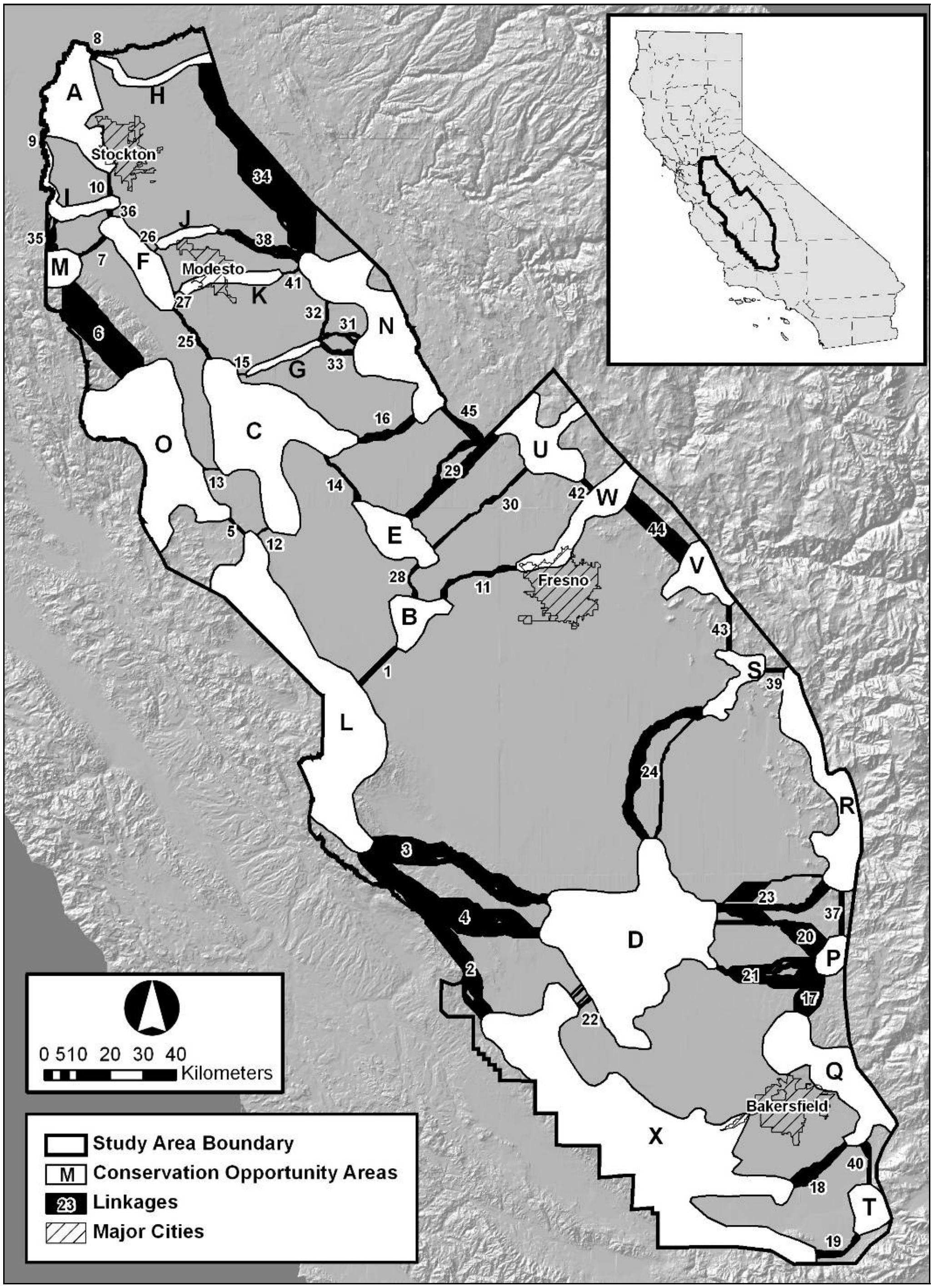


Figure 4.3. Potential ecological network for the San Joaquin Valley, CA (inset). White polygons represent potential COAs (conservation opportunity areas) and black polygons represent areas of the highest connectivity (current or potential linkages) between them. Letters and number shown are the identification keys to the COAs and linkages listed in Tables 4.4 and 4.7.

Table 4.5. Landscape linkage clusters grouped using 3 variables. "Land cover" refers to the predominant land cover type in the linkage. "Barriers" are either highways or major canals. "Linkages" are the number of linkages in each cluster.

Cluster	Land cover	Barriers	Length	Linkages
1	riparian	few	variable	8
2	other natural	few	moderate	13
3	agricultural	few	short	4
4	other natural	many	long	5
5	agricultural	many	variable	15

we assumed to have the lowest ecological function contained linkages comprised of agricultural land cover, many barriers, and variable length. Of the 45 total barriers, 22 (48.9%) crossed mostly natural land cover and few major barriers. The other 23 traversed mostly or fully agricultural lands.

COA future condition

The urbanization rates predicted for the COAs by each scenario can be used to rate the scenarios for their overall ecological effect. The Compact Growth scenario predicted the least amount of urbanization of the COAs (Table 4.4) as well as the smallest extent of lands in the study area converted to urban land use overall. For COAs clustered into the three groups, the average percent conversion to urban was: 0.2% valley floor, 1.9% riparian, and 1.2% upland (Table 4.6). The range for individual COAs was from 0% (9 COAs) to 8.6% (Corral Hollow [*M*]).

The scenario showing the greatest amount of impact on the COAs was less obvious. The Status Quo scenario pointed to five COAs experiencing greater than 10% conversion to urban land use (Table 4.4). However, the Status Quo scenario did not result in the highest conversion rate of the seven scenarios for any of the COAs. The "Farmland Protection" scenario (avoiding Prime Agricultural Soils) had the greatest impact on the majority of COAs, with 14 COAs showing their highest conversion rate under this scenario. However, the riparian COA group experienced few impacts, its 5.0% conversion rate under this scenario being the second lowest (after Compact Growth). In contrast, valley floor and upland COA groups experienced their highest conversion rates

Table 4.6. Mean urbanization rates of COA within clusters for each scenario.

Cluster	S1 (%)	S2 (%)	S3 (%)	S4 (%)	S5 (%)	S6 (%)	S7 (%)
1 - Valley Floor	3.8	2.9	0.2	4.9	0.6	0.5	1.2
2 - Riparian	16.3	16.6	1.9	5.0	16.9	11.4	14.8
3 - Uplands	3.6	3.4	1.2	9.2	6.1	1.2	1.0

under the Farmland Protection scenario. The scenario showing the heaviest impact on any single COA was Great Cities, with the Stanislaus River COA [*J*] anticipated to experience a 46.0% conversion rate. Also, while Farmland Protection predicts one COA to experience heavy conversion (> 20%), four other scenarios predict that two COAs will be subject to this level of impact. The scenario showing the greatest effect (16.9% mean) on the riparian group was Exclusion Zone.

Of the three COA ecological types, urbanization is expected to most affect the riparian group (11.8% mean) under these seven scenarios while it is predicted to have the lowest impact on the valley floor group (2.0% mean). The individual COA expected to experience the highest minimum amount of conversion across the seven scenarios is Tuolumne River [*K*] (4.8% under Farmland Protection) while that predicted to experience the lowest maximum impact is Tejon Hills [*T*] (2.8% under Farmland Protection and East-West Connectors).

Linkage future condition

As with the COAs, urban growth scenarios can be rated for their effects on the future viability of the linkages. Generally, Compact Growth had the lowest impact on the linkages (Table 4.7). All five clusters experienced their lowest degradation rate under this scenario and three of the five clusters experienced the lowest chokepoint rate (Great Cities has lower chokepoint rates for clusters 4 and 5) (Table 4.8). The lowest mean degradation rate was experienced in Compact Growth (0.5%), however the lowest mean chokepoint rate was found in Great Cities (9.7%).

Table 4.7. Degradation (Deg) and Chokepoint (CP) rates (%) for all linkages under all seven scenarios. Gray shading indicates high impact. Referred to in text in soft brackets ( ).

Linkage ID	Cluster	S1 – Deg	S1 – CP	S2 – Deg	S2 –CP	S3 – Deg	S3 – CP	S4 – Deg	S4 – CP	S5 – Deg	S5 – CP	S6 – Deg	S6 – CP	S7 – Deg	S7 – CP
1	5	7.2		7.2				0.8							
2	4	1.4		1.3				2.9		3.1	4.6				
3	4	3.2	38.3	3.2	38.3	1.3	**90.5**	14.5	**98.6**	15.4	**98.6**	1.2	32.3		
4	4	2.4		2.4				11.2	42.7	17.4	63.5				
5	2	1.2		3.3				4.0		5.8					
6	2	0.7		0.7				2.4	31.1	1.2					
7	5	**50.2**	**100.0**	**49.6**	**100.0**	11.8	**88.0**	11.7	**100.0**	**49.4**	**100.0**	**41.3**	**100.0**		
8	3	5.8		10.0											
9	1														
10	3	14.5		8.5						17.3				4.5	44.7
11	1	3.1		4.1				2.0	29.0					4.9	51.4
12	5			4.2				6.8							
13	5	**48.2**	**100.0**	**42.6**	**100.0**	3.5	**75.0**	**77.9**	**100.0**	1.0	37.7	**79.0**	**100.0**		
14	2	1.6		3.0				8.5							
15	1	5.1		7.2				6.8	61.0						
16	5	5.5		4.3				8.5	72.6	7.9	**84.5**			0.9	16.0
17	2	3.9		4.1				7.2		6.0					
18	5	17.5	**100.0**	16.6	**100.0**			**55.3**	**100.0**	**42.8**	**100.0**	14.8	**100.0**	8.5	59.3
19	5	5.1		3.6				10.2		6.5					
20	5	12.0	14.5	11.9	14.5			**24.5**	25.3	17.1	16.6	0.1	16.6		
21	5	6.5	**100.0**	5.7	**100.0**	0.5	68.2	5.3	**100.0**	14.2	**100.0**	1.2	**100.0**	1.7	**100.0**
22	5	8.2		5.6				0.1		8.4					
23	5	13.1	24.1	13.1	24.1			16.5	**100.0**	17.4	26.4	0.3	26.4		

Table 4.7. Continued.

Linkage ID	Cluster	S1 – Deg	S1 – CP	S2 – Deg	S2 –CP	S3 – Deg	S3 – CP	S4 – Deg	S4 – CP	S5 – Deg	S5 – CP	S6 – Deg	S6 – CP	S7 – Deg	S7 – CP
24	4	6.3	57.0	6.3	57.0			6.0	60.9	2.0	5.5	1.4	57.0	5.7	5.5
25	1	14.1	65.8	12.6	65.8			12.7	**80.3**			8.4	65.8		
26	1							6.6	**100.0**						
27	1														
28	2	1.3		2.0				3.6							
29	5	6.2	42.3	7.5	61.3	1.5	42.3	8.4	61.3	6.0	51.5	2.8	48.3		
30	5	16.1	**100.0**	12.7	**100.0**	3.5	**100.0**	15.7	**100.0**	15.8	**100.0**	9.0	**100.0**		
31	1	3.6		3.3				6.9	37.0	18.4	73.0				
32	3	3.3		4.5				3.8		8.1					
33	2	6.1	66.3	5.0	66.3			9.0	**81.7**	**22.6**	**100.0**	0.2	18.9		
34	4	9.1	23.4	9.3	23.4			**25.0**	65.6	16.0	18.3	0.8	18.3		
35	5	14.6	**92.6**	15.4	**97.5**	0.6	38.9	**32.3**	**100.0**	16.6	**94.9**	9.8	**92.6**	0.6	39.1
36	3														
37	2	10.5		10.5				**39.0**		13.9					
38	5	16.7	**100.0**	16.0	**90.8**	0.6	35.8	**36.8**	**100.0**	**31.8**	**100.0**	7.8	**96.7**	15.6	**100.0**
39	2	5.5		5.5				19.6		4.9					
40	2	4.9		5.2				0.6		8.0					
41	1	0.9						16.8	63.0	3.7					
42	2			1.2				2.8		2.3					
43	2	1.9		1.9				12.2	**100.0**	12.6	**100.0**				
44	2	4.6	65.5	4.6	65.5			**44.7**	**100.0**	18.7	**91.4**	1.6	43.1	0.1	18.5
45	2	3.2		2.6				8.1		9.2					
Mean		7.7	24.2	7.5	24.5	0.5	12.0	13.1	42.4	9.8	30.1	4.0	22.6	0.9	9.7

Table 4.8. Mean "degradation" (Deg) and "chokepoint" (CP) rates (%) of linkage clusters for each scenario.

Cluster	S1 – Deg	S1 – CP	S2 – Deg	S2 – CP	S3 – Deg	S3 – CP	S4 – Deg	S4 – CP	S5 -Deg	S5 – CP	S6 – Deg	S6 – CP	S7 – Deg	S7 – CP
1	3.4	8.2	3.4	8.2	0.0	0.0	6.5	46.3	2.8	9.1	1.1	8.2	0.6	6.4
2	3.5	10.1	3.8	10.1	0.0	0.0	12.4	24.1	8.1	22.4	0.1	4.8	<0.1	1.4
3	5.9	0.0	5.8	0.0	0.0	0.0	1.0	0.0	6.4	0.0	0.0	0.0	1.1	11.2
4	4.5	23.7	4.5	23.7	0.3	18.1	11.9	53.6	10.8	38.1	0.7	21.5	1.1	1.1
5	15.1	51.6	14.4	52.5	1.5	29.9	20.7	63.9	15.7	54.1	11.1	52.0	1.8	21.0

The Farmland Protection scenario displayed the highest rates of both degradation and chokepoints for all clusters except cluster 3 (short agricultural linkages). Cluster 3 experienced the heaviest degradation under the Exclusion scenario (6.4%) and the highest chokepoint rate under the Great Cities scenario (11.2%). Figure 4.4 shows the results of three of the UPlan model runs (Status Quo, Farmland Protection, and Great Cities) on two of the linkages for a visual portrayal of some of these results.

DISCUSSION

More effective conservation planning on working landscapes requires both the identification of restoration priorities to re-establish landscape connectivity and the assessment of different future impacts of urban growth, as well as the traditional identification of important remnant ecological features for protection. This study quantified the current ecological condition and future vulnerability of cores and linkages of a modeled ecological network in a working, urbanizing, agricultural landscape. By doing so, it provides a model of how to develop conservation designs in such landscapes. The model's steps include landscape connectivity modeling, ecological condition assessment, and analysis of multi-scenario urban growth modeling, which provide a framework for assessing threats to landscape connectivity. The quantification of the current and future ecological condition of habitat linkages can be used to identify a prioritization scheme for their protection and restoration. The assessment of current and future linkage condition permits their classification into appropriate conservation management actions including acquisition, restoration, or regulatory approaches.

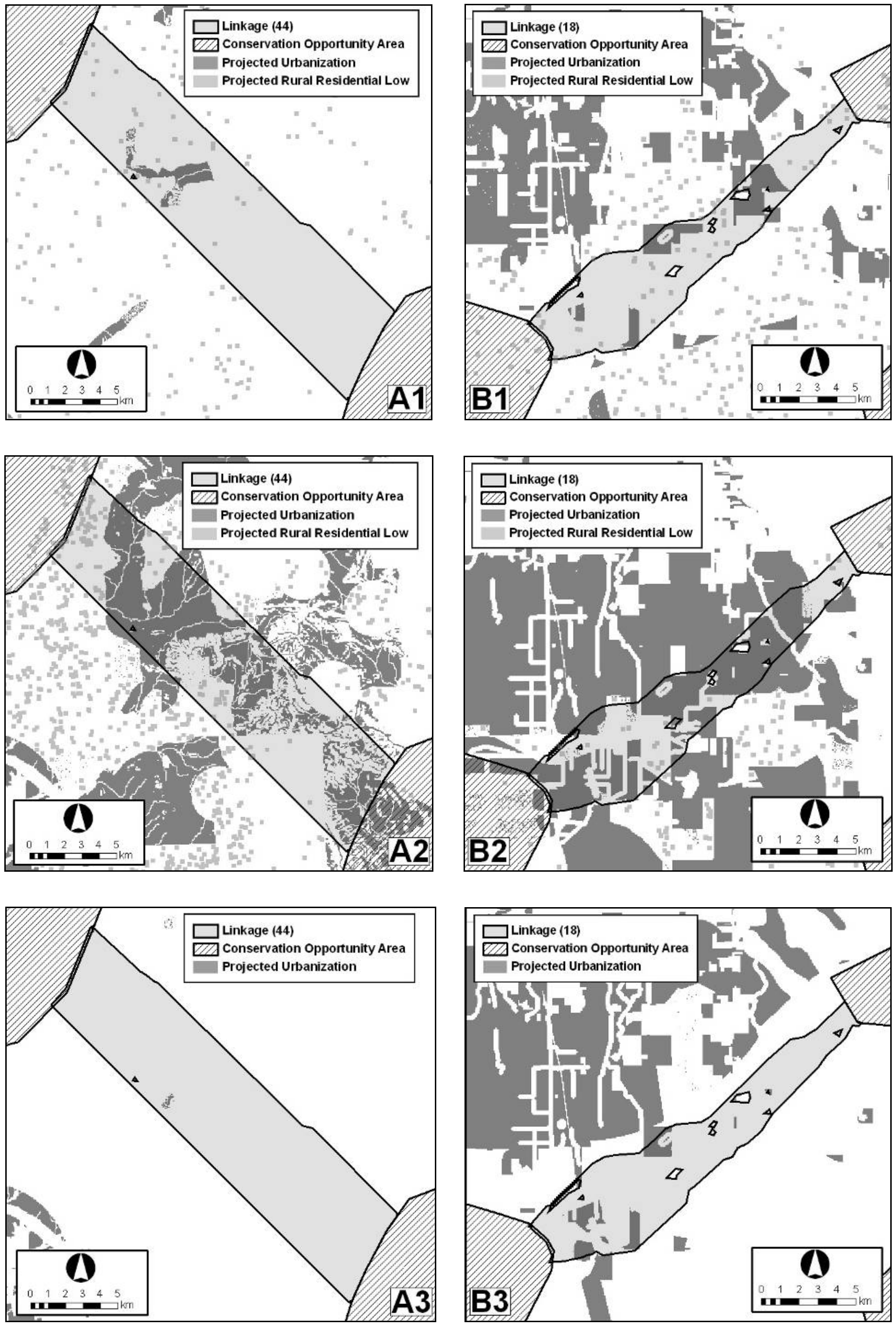


Figure 4.4, Results of three urban growth scenarios overlaid on two linkages. Maps A1-A3 show Linkage 44 (Upper San Joaquin – Upper Kings) while B1-B3 show Linkage 18 (West Kern – NE Bakersfield. A1 and B1 show Scenario 1, A2 and B2 Scenario 4, and A3 and B3 Scenario 7.

Simulation of future urban growth scenarios representing different land use policies was useful in analyzing potential threats to the COAs and linkages. For example, one surprising finding is that Status Quo, a development scenario whose projected negative effects had driven much of the effort to integrate land use planning across the region, was not the highest impact scenario for any of the identified COAs. The largest loss of ecologically significant land to urbanization was predicted to occur under the two agriculturally-centered scenarios, Prime Agricultural Soils Protection and Exclusion Zone. These scenarios force most of the projected urbanization into areas that are currently more ecologically intact, i.e. the grasslands and oak woodlands in the uplands surrounding the valley floor. In addition, the Prime Agricultural Soils Protection scenario leads to highly fragmented development across the valley floor wherever there are areas of lower quality agricultural soils. However, the Status Quo scenario did have impacts: seven of the 45 important linkage areas will be heavily impacted by urbanization, with many others facing considerable reduction of functional capacity as ecological linkages.

The habitat linkage cluster analysis permitted the determination of conservation actions appropriate for various types of preservation or restoration of linkages. For example, efforts to preserve the existing land cover should be the focus for the 21 linkages assigned to the riparian forest (cluster 1) and intact natural land cover (cluster 2) clusters, which are currently the least degraded land cover classes. In contrast, the 15 linkages of cluster 5 (agricultural) will require substantial effort to restore ecological function. Much habitat restoration and creation would be needed as well as mitigation of the effects of many substantial barriers. These classes of conservation action may prove useful from a rule-based perspective, for groups attempting to identify conservation actions for linkages in other working landscapes. Additionally, these linkages may be more difficult for planners to identify, appearing as agricultural land, and thus they are more likely to be overlooked during the planning process. In this case the ecological network modeling helped to identify these linkages.

The linkages most threatened by projected urbanization are those in cluster 5 (agricultural lands). These linkages currently are fully agricultural and cross numerous substantial barriers and thus display low ecological integrity. Only Compact Growth (four linkages) and Great Cities (two) scenarios predict less than 7 of these 15 linkages being substantially (≥75% chokepoint) or fully (100%) impacted by development. Further, under all seven scenarios, the mean urbanization impact (as measured by both degradation and chokepoint) is higher for cluster 5 linkages than for other linkages. These results suggest that a major concern about the effects of urbanization on ecological connectivity in the San Joaquin Valley should be the loss of restoration opportunities rather than the impacts to current ecological networks. The fact that these linkages have little natural land cover indicates that they will be less noticeable on the landscape and potentially easily overlooked in the process of conservation planning.

As a note of caution, comprehensive conservation prioritization schemes should include network analysis in order to better understand the key locations of nodes and linkages in a conservation network. While it was possible to rank the COAs by the number of other COAs to which they are linked (e.g. Grasslands EA [*C*], KKT [*D*], and Vernal Pools [*N*] are all linked to six other COAs) it was beyond the scope of this paper to analyze the results of loss to the network of any particular COA or linkage. However, to do so would provide a clearer picture of spatial irreplaceability (and its effect on important ecological features such as population viability) than we currently have.

An additional area for future research lies in species-specific modeling. This study modeled "general connectivity" for terrestrial species in the study area by creating a single cost surface to model all animal movement. However, individual species will react differently to local conditions when navigating the landscape. Thus, it would be useful to select a suite of focal species and model each of their movement patterns (e.g. Beier et al. 2008). Nevertheless, this study provides a foundation from which to understand and begin to respond to imminent threats to current and restorable landscape connectivity for wildlife in the San Joaquin Valley. The methods presented here could be effective in other regions, especially those that are historically degraded in ecological integrity and those that face increasing human pressure for development in the near future.

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