Los Angeles County Department of Regional Planning
Literature Review and Summary of California Native Grassland
Wesley Colvin, Ph.D.

Regional Overview of Valley and Foothill Grassland in California

In general, the popular treatment of the grasslands in California centered on the Great Central Valley grasslands of the Sacramento and San Joaquin Valleys, larger valleys in the south Coast Ranges, and cismontane southern California where native perennial bunchgrasses such as: needlegrasses (*Nassella* spp.), three-awned grasses (*Aristida* spp.), bluegrasses (*Poa* spp.), and rye grasses (*Elymus* spp.) historically predominated (Bakker 1984, Schoenherr 1992, Barbour et al 1993). Also detailed in these popular treatments, the coastal prairies on the Coast Ranges of northern California contain floristically different grasslands, such as fescues (*Festuca* spp.), hairgrasses (*Deschampsia* spp.), and oatgrasses (*Danthonia* spp.) due to cooler and wetter environmental conditions. For purposes of discussion, low-elevation vegetation dominated by herbs makes the terms *prairie* and *steppe* synonymous with *grassland*; and, for any given grassland community, 3 to 4 grass species make up the majority of the aboveground biomass, but forb species richness can be 4 times that of the grasses and grass-allies such as rushes and sedges (Keeler-Wolf et al 2007).

Bartolome et al (2007) begin their discussion of valley grassland as a vegetation type comprised largely of nonnative, naturalized grass and forb species (herbs) with an annual life-history compared to the original grassland dominants- perennial bunchgrasses. They continue their discussion by comparing the valley grasslands of California with the Palouse prairie of Washington and Oregon by pointing out that perennial grasslands in both locations share the following species in common: blue bunchgrass (*Festuca idahoensis*), junegrass (*Koeleria macrantha*), one-sided bluegrass (*Poa secunda* ssp. *secunda*), giant wild rye (*Leymus*

condensatus), and squirreltail grass (*Elymus elymoides*). The species gradually lose their importance from north to south and from the coast inland, thus forming a continuum between Palouse prairie and the valley grassland. The Palouse dominant, bluebunch wheatgrass (Pseudoroegneria spicata), and the presumed California dominant, purple needlegrass (*Nassella pulchra*) are not shared between the two regions. Other important grass species associated with purple needlegrass include: spidergrass (*Aristida ternipes* var. *gentilis*), blue wild rye (*Elymus glaucus*), California melic (*Melica californica*), small-flowered melic grass (*M. imperfecta*), and deergrass (*Muhlenbergia rigens*) (Bartolome et al 2007, Keeler-Wolf et al 2007). In the southern end of the valley grassland range, nodding needlegrass (*Nassella cernua*) increases in importance over purple needlegrass (*N. pulchra*) along with the native annual grasses as: annual hairgrass (*Deschampsia danthonioides*) and pacific fescue (*Vulpia microstachys* var. *pauciflora*) (Bartolome et al 2007). The perennial saltgrass (*Distichlis spicata*) is reported on alkaline soils in the southernmost of the San Joaquin Valley, including the Carrizo Plain, north to Contra Costa County while big squirreltail (*Elymus multisetus*) is known on rocky or clayey serpentine soils in the Central Coast Ranges (Keeler-Wolf et al 2007).

Valley grassland extends into the understory of oak savannas and woodlands, as well as into chaparral and other scrub vegetation types with little changes in its herbaceous characteristics (Bartolome et al 2007). Blue wild rye (*E. glaucus*) is typically adjacent to oak woodlands, meadow barley (*Hordeum brachyantherum*) near marshes and seeps on heavier soils, and beardless wild rye (*L. triticoides*) beneath valley oaks on poorly drained floodplain soils and around seeps and drainages where it is associated with deergrass (*M. rigens*), valley sedge (*Carex barbarae*), clustered field sedge (*Carex praegracilis*), and rushes (*Juncus* spp.) (Keeler-Wolf et al 2007).

Local Overview of Grasslands within Los Angeles County

The earliest botanical descriptions of the grassland now occupied by Los Angeles clearly fit the purple needlegrass (*N. pulchra*) alliance and ecosystem of the Great Central Valley (Keeler-Wolf et al 2007). A transition between valley grassland and Mojave Desert steppes occurs over a short distance on the eastern slopes of the Tehachapi and other southern California mountains. In this transition zone, nodding needlegrass (*N. cernua*) and desert needlegrass (*Achnatherum speciosum*) replace purple needlegrass (*N. pulchra*) and beardless wild rye (*Leymus triticoides*) (Bartolome et al 2007). These warm desert grasslands in the Antelope Valley are distinct from the valley grasslands to the west (Keeler-Wolf et al 2007). Pavlik (2008) reported that grasslands are rare in the desert and are dominated by species of perennial bunchgrasses. Pacific fescue (*V. microstachys* var. *pauciflora*) is the most common native annual grass of desert grassland (Bartolome et al 2007).

In Los Angeles County, the number of Federal listed endangered or threatened grassland-associated species ranges between 5 to 8 species, and this number of species increases from 13 to 18 for State listed endangered or threatened grassland-associated species (Jantz et al 2007). These numbers of listed species are greater than previously reported for valley grassland (Pavlik 2003). Comparing the projected location of future development with the current distribution of grassland-associated threatened and endangered species reveals that the Western Transverse Ranges, San Gabriel Mountains, and Mojave Desert ecoregions of the California Floristic Province (Hickman 1993) are where grasslands are likely to be heavily impacted from development within the County (Jantz et al 2007). Since the 19th Century, the state has lost 99 percent of its valley grasslands as a result of conversion for other uses (Dasmann 1981, Preston

1997, Pincetl 1999). This leaves a total of 10,751,121 acres (Jantz et al 2007) of grassland within the State, which is 13% less than reported 4 years earlier (Pavlik 2003). All published maps of undifferentiated (native perennial and introduced annual) grassland indicate that its greatest spatial extent occurs in the Antelope Valley of the northwest corner of Los Angeles County (Dasmann 1970, CDFG 2003, Bartolome et al 2007, Jantz et al 2007, Keeler-Wolf et al 2007), while maps differentiated by classification of grassland by native versus annual growth form also concur that the greatest spatial extent for either native perennial grassland (Keeley 1990) or introduced annual grassland (Minnich 2008) occurs in Antelope Valley. The last analysis to evaluate grasslands within the Los Angeles Basin, spanning Los Angeles and Orange Counties, indicated that introduced annual grasslands expanded their spatial extent from the 1930s to 1980s at the expense of native grassland and shrubland vegetation communities (Freudenberger et al 1987).

Persistence of Native Grasslands

Species accounts do not address the abundance (frequency of individual occurrences) or relative cover of the species within vegetation communities. As a result, the grass species listed in the accounts for proposed development projects may occur as members of other vegetation communities that occur on the site - such as the understory of oak savannas or Joshua tree woodlands or interspersed amongst shrubs in scrub vegetation. However, at its most basic level, species richness is a measure of biological diversity (Preston 1948, Grubb 1977, Shmida and Wilson 1985, Spellerberg and Fedora 2003), and diversity increases with area (Connor and McCoy 2001, Collins et al 2002, Scheiner et al 2000). Therefore, species accounts provide a simple measure of biodiversity in relation to geographic area: As the spatial extent increases, one

would expect to find greater species diversity. A parallel concept is that highly species diverse ecosystems are more stable through time than less species diverse ecosystems (Tilman and Downing 1994, Tilman et al 1996, Tilman et al 2006). Species diversity is not the driver of this relationship; rather, ecosystem stability depends on the ability for biological communities to contain a range of species (or functional groups) that allow ecosystems differential responses to change (McCann 2000). However, it should be pointed out that the diversity criteria of 3 or 4 grass species described by Keeler-Wolf et al (2007) above to classify a grassland does not take into consideration their center of origin (native vs. introduced) of the grasses. Considering that the main drivers of global environmental change (CO^2 enrichment, nitrogen deposition, climate, biotic invasions and land use) cause extinctions and alter species distributions, recent evidence shows those environmental changes exert pervasive impacts on various interactions among species (Tylianakis et al 2008). Susceptibility to invasion by exotic species is strongly influenced by species composition and, under similar environmental conditions, generally decreases with increasing species richness. Having a diversity of species that respond differently to different environmental perturbations can stabilize ecosystem process rates in response to disturbances and variation in abiotic conditions, which has prompted calls for the conservation of ecosystem network structure (Hooper et al 2005, Tylianakis et al 2010). Using practices that maintain a diversity of organisms of different functional group and response types will help preserve a range of management options (Hooper et al 2005). Since the early emergence of plant seedlings can offer strong competitive advantages over later-germinating neighbors through the preemption of limiting resources, this phenomenon may have contributed to the persistent dominance of European annual grasses over native perennial grasses in California grasslands because the former species typically germinate earlier in the growing season than the latter and grow rapidly

after establishing (Abraham et al 2009). Therefore, grasslands with fewer introduced species (a.k.a: alien, exotic, invasive, nonnative) in their species accounts have a higher conservation value. In other words, grasslands with a larger 'nativity' index value possess fewer introduced species. Conceptually, the 'nativity' index is based on the Index of Alien Impact (IAI) used to estimate the collective ecological impact of in-situ alien species (Magee et al 2008, 2010). IAI summarizes the frequency of occurrence and potential ecological impact of individual alien species for all aliens present in a particular location or community type. As a result, the ecological functions and processes that occur within grasslands with more native species are more likely to resemble conditions that existed prior to their degradation because these impacts generally shift the balance of species present from large, long-lived species (often native perennials) to small, short-lived ones (introduced annuals) (Mooney et al 1995).

Biodiversity and the Tehachapi Range

According to Hickman (1993), the function of providing geographic ranges in a botanical resource is to help the user predict where plant taxa can be expected to grow, and the emphasis on predictiveness and biological meaning in range descriptions automatically diminishes the importance of county lines that are the primary alternatives for subdividing the state. In contrast to the use of biologically arbitrary often politically determined delimiters, such as county lines, wherever possible, subdivisions of the floristic provinces are defined on the basis of all three of the main biologically relevant variables- topography, climate, and plant-community types (Hickman 1993):

> The Tehachapi Mountains are a small foothill and montane ecoregion creating a floristic melting pot between the Sierra Nevada, Great Valley, Transverse Ranges, and Desert. Its

boundary in the north with southern Sierra Nevada Foothills is along Highway 58 through Tehachapi Pass. In the west, it is bounded by the Great Central Valley (marked by prairie communities in the lower foothills vs. mixed-woodland communities). Tehachapi ends at the north end of the Western Transverse Ranges district (Tejon Pass on Interstate 5). The eastern boundary is the indistinct Califorina Floristic Province boundary with the Mojave Desert (chaparral or pinyon/juniper woodland vs. creosote-bush scrub).

The Sierra Nevada Foothills are characterized by blue-oak/foothill-pine woodlands (vs. ponderosa-pine forest of higher elevations in the High Sierra Nevada) throughout most of their extent, and are dotted with serpentine. They are divided into three districts from north to south that are best differentiated from the High Sierra Nevada and Great Central Valley by community type rather than topography or map lines.

The Great Central Valley once supported grassland (California prairie), marshes, extensive riparian woodlands, and islands of valley-oak savanna- especially in the southern San Joaquin Valley. The San Joaquin Valley is the larger, southern subregion of the Great Central Valley. Hotter and drier than the Sacramento Valley, it supports some desert elements in the south. Some vegetationally similar outliers to the southwest are included, such as the Carrizo Plain.

The Transverse Ranges are an ecoregion of abnormally oriented (east-west) mountains characterized at lower elevations by chaparral and at higher elevations by southern oak forest and dry montane forests of white fir or Jeffrey, sugar, or lodgepole pines. Some high peaks are treeless, but apparently none exceed climatic timberline at their latitude. The northern boundary with the Mojave Desert tends to grade into desert vegetation on the lower slopes of these mountains. The Transverse Ranges are divided into three districts (Western Transverse Ranges, San Gabriel Mountains, and San Bernardino Mountains) that are progressively higher, hotter, and drier to the east.

The Desert Province makes up the southeastern portion of California. The province is divided into two ecoregions. The Mojave Desert ecoregion occupies the northern two-thirds of the Desert Province. It has greater temperature ranges and more diverse elevations than the Sonoran Desert to the south. Mojave creosote-bush scrub is the dominant vegetation type, with saltbush scrub characteristic of alkaline basins.

DRAFT

When Hickman (1993) delimited the northernmost boundary of the Tehachapi Mountains ecoregion with the southern Sierra Nevada Foothills along SR-58 through Tehachapi Pass and its southernmost limit end at the north end of the Western Transverse Ranges along I-5 through Tejon Pass, Barbour et al (1993) showed pictures of grasslands on both the Tehachapi Pass (SR-58) and Tejon Pass (I-5) sides of the Tehachapi ecoregion. The captions for the Tehachapi Pass pictures indicated that the native annual forbs of lupine and owl's clover produce showy displays in the spring. They postulated that these species of wildflowers probably clustered in patches between native bunchgrasses in the original Central Valley grassland. Then, they stated that in the Central Valley of 1993 these wildflowers are often scattered across a sea of non-native annual brome grasses. The caption for the Tejon Pass picture indicated that showy annuals of California poppy, goldfields, and lupine can dominate grasslands in the spring. As Lulow (2006) pointed out, the Central Valley of California is noted for its dearth of remnant native grass populations and for low native grass seedling establishment within grasslands now dominated by non-native annual species.

The Tehachapi ecoregion represents a crossroads between the other major regions of California: the Sierra Nevada, the Coast Ranges, the Central Valley, and the deserts of the southeast resulting from complex geology that includes three major faults that were active during the Pleistocene (Davis et al 2007, Vandergast et al 2008). Many species have ranges that extend from these areas into the Transverse Ranges, and it is the conjunction of these ranges that makes the Tehachapis important. Evolutionary processes in the Transverse/Tehachapi Ranges are probably dominated by both the environmental gradients in this transitional area and by the complex biotic interactions among the populations there (Davis et al 2007). As a result, this region is a hotspot for biological diversity.

The Tehachapi Ranges possess great biological diversity. It's considered a speciation hotspot for plants (Raven and Axelrod 1978), especially for Valley Oak. Research by Grivet et al (2008) found the distribution of multivariate genotypes and high allelic richness for these oaks and identified areas with distinctive histories and genetic composition that should be given priority in reserve network design, especially because these areas also overlap with landscape change and little degree of protection from development. Thus, without a careful preservation plan, valuable evolutionary information will be lost for valley oak (Grivet et al 2008). Valley oak distribution in Los Angeles occurs in the NW corner of the County. For mammals, it is the possible point of origin for southern California lineages of dusky-footed woodrat, Neotoma fuscipes (Matocq 2002), and Kellogg (1918) noted long ago by in his taxonomic revision of meadow mice the coincidence of so many endemic subspecies in the Transverse and Tehachapi Ranges and immediately adjacent Coast Range. However, it was the evaluation of 25 species of mammals by Davis et al (2007) that identified this area as a hotspot for mammals. Similarly, Rissler et al (2006) determined for 22 species of Californian amphibians and reptiles that the Tehachapis rank as one of the most irreplaceable locations in the State. Since, the least fragmented watershed occurs in the NW corner of Los Angeles County (Girvetz et al 2008), opportunities to protect multiple species assemblages here or restore connectivity among existing reserves may still exist, particularly along riparian corridors (Vandergast et al 2008).

The high rate of anthropogenic impact on natural systems mandates protection of the evolutionary processes that generate and sustain biological diversity (Davis et al 2007). Modeling climate change on California's endemic flora, Loarie et al (2008) found that areas projected to harbor species with shrinking ranges include many mountainous areas scattered across their study area. They identified the Coast and Transverse Ranges in Los Angeles County

as refugia that may disproportionately contain the most ‘‘threatened’’ species. These ‘‘future refugia’’ present valuable opportunities as conservation targets, since they may protect significant components of biodiversity into the next century. The number of species projected to survive in these refugia depends critically on the ability to disperse, highlighting the importance of landscape connectivity and potential restoration in the face of increasing urbanization, land use change, and disturbance (Loarie et al 2008). To maintain this connectivity, a preliminary linkage design from a union of eight single-species corridors was created by Beier et al (2009). The source areas for each single-species corridor were large blocks of species-specific suitable habitat in protected lands from the Coast Ranges to the Sierra Nevada ranges. The Tehachapi Mountains are the main mountain range in the matrix- with the western Antelope Valley providing another connection from the Transverse Ranges to the Tehachapi Mountains (Beier et al 2009).

Conservation Planning Informed by Conservation Biology

Examining patterns of rare plant occurrences among all counties in the United States revealed that high and medium density counties contained, on average, as many or more rare and endemic species than low density counties (Schwartz et al 2006). Examining species losses and noxious weed additions across high density counties, revealed a consistent pattern of low similarity among species that were extirpated from high density counties and a high similarity among noxious weeds that these counties now share. The consequence is that California’s urban county floras appear to be homogenizing. The process by which regionally distinct, native communities are gradually replaced by range-expanding, cosmopolitan, non-native communities (i.e. decreasing beta-diversity over time) has been termed biotic homogenization (McKinney and

Lockwood 1999). Examining homogenization using the entire flora for urban counties demonstrates that less similar counties become more similar (Schwartz et al 2006). Because species invasions and extirpations all too often result in the wake of environmental degradation, biotic homogenization seems an important dimension of the biodiversity crisis (Rooney et al 2007). Recent evidence showing increases in local species diversity owing to species invasions could well undermine efforts to battle invasive species and sustain native biological diversity (Olden 2006). Structural equation modeling suggests that while human activities, such as urbanization and agriculture, facilitate the initial invasion by exotic plants, exotics spread ahead of the front of human development into areas with high numbers of threatened native plants. The range sizes of exotic taxa are an order of magnitude smaller than for comparable native taxa. The current small range size of exotic species implies that California has a significant "invasion debt" that will be paid as exotic plants expand their range and spread throughout the state (Seabloom et al 2006). The effect of loss of rare species could outweigh the gain in exotics, under an assumption of strong extinction. Finally, a strong negative relationship between population density and the proportion of county land in public ownership suggests that high and medium density counties are in a poor position to protect rare plant populations on a localized basis (Schwartz et al 2006).

Over a 32 year period (1967 to 1999), the number of threatened or endangered species increased in Los Angeles County by 99 listed species (Rutledge et al 2001). With Los Angeles County identified as a hot spot of vulnerability - areas vulnerable to future species loss- based upon the presence of restricted-range birds and butterflies not listed as threatened or endangered under the Endangered Species Act due to expected increases in human population density (Abbitt et al 2000), the popularity of exurban living challenges regional planners as they seek to

conserve natural resources, scenic amenities, and natural open space (Marcouiller and Tremble 2009). The wildland–urban interface (WUI) is the area where houses meet or intermingle with undeveloped wildland vegetation. The WUI is thus a focal area for human–environment conflicts, such as the destruction of homes by wildfires, habitat fragmentation, introduction of exotic species, and biodiversity decline. The WUI in the conterminous United States covers 719,156 km2 (9% of land area) and contains 44.8 million housing units (39% of all houses). California has the highest number of WUI housing units (5.1 million) (Radeloff et al 2005). Reduced abundance of grassland bird species (vesper sparrow, grasshopper sparrow, and horned lark) along the urban edge is due to the loss of preferred grassland cover-types at the urban-wildland interface (Haire et al 2000). Only by addressing this growing conflict between cities and biodiversity can society achieve genuine conservation in an urbanizing world (McDonald et al 2008).

Conservation planning focuses on policies of protecting land in biological reserves because the primary threat to biodiversity is habitat loss (Polasky 2006). Fragmentation generally results in a landscape that consists of remnant areas of native vegetation surrounded by a matrix of agricultural or other developed land. The influences of physical and biogeographic changes are modified by the size, shape, and position in the landscape of individual remnants, with larger remnants being less adversely affected by the fragmentation process (Saunders et al 1991). In a review of biodiversity planning tools, Sarkar et al (2006) describe how the accelerated pace of habitat change and natural resource utilization since the 1960s and the resultant threats to biodiversity led to increased concern for protecting remaining natural areas since the 1970s. Sociopolitical considerations, including the legitimate desire for economic development, dictated that not all areas of biological interest could be protected. The design of adequate networks of

conservation areas, protecting the most important sites in each region, became a central problem of the new interdisciplinary field of conservation biology, which emerged in the 1980s with the explicit task of halting the decline of biodiversity (Sarkar et al 2006). Establishing representative conservation area networks in which biodiversity can persist has also become a policy goal for major governmental, intergovernmental, and nongovernmental organizations (Margules and Pressey 2000). Early efforts at the design of conservation area networks were typically based on the theory of island biogeography (Diamond 1976). Nature reserves were supposed to be similar to islands in oceans of anthropogenically transformed habitats (Sarkar et al 2006). Species-area curves were often invoked to estimate optimal sizes for conservation areas (May 1975). Reserves are often designed to protect rare habitats, or ‘‘typical’’ exemplars of ecoregions and geomorphic provinces. This approach focuses on current patterns of organismal and ecosystem-level biodiversity, but typically ignores the evolutionary processes that control the gain and loss of biodiversity at these and other levels (e.g., genetic, ecological) (Vandergast et al 2008). Given limited resources, many researchers advocate focusing conservation efforts on hotspots, geographical areas with high numbers of species, (i.e. richness), endemic species, rare or threatened species, and /or high level of threat to species survival. The observed distribution of endangered species reflects the modification of the underlying pattern of species diversity by anthropogenic activities (Dobson et al 2001). The hotspot approach is an efficient and simple way to conserve species diversity, assuming that hotspots do not change over space or time (Rutledge et al 2001). As it is now, protected areas are unable to buffer against broad-scale shifts in the distribution of species or ecosystems that result from global environmental change (Lee and Jetz 2008). Many conservation biologists believe that a focus on endangered species is misplaced. They argue that the sheer number of species at risk makes a species-by-species

approach impractical or even futile (Wilcove 2010). Roberge and Angelstam (2004) found that single-species 'umbrellas' cannot ensure the conservation of all co-occurring species because some species are inevitably limited by ecological factors that are not relevant to the umbrella species. Although some surrogate species may have considerable publicity value, the utility of umbrella and flagship species as surrogates for regional biodiversity may be limited (Andelman and Fagan 2000). For example, conservation activities designed for the Bay checkerspot butterfly were found to provide a tenuous protective umbrella for other elements of the grassland community (Launer and Murphy 1994), and the context and methods of published studies were so diverse that Favreau et al (2006) could not draw general conclusions about the spatial or temporal scales, ecosystems or taxonomic groups for which surrogate species approaches will succeed. Experience suggests that a more comprehensive, refined, and proactive approach is needed to protect large areas that support whole communities of wildlife and other natural resources (Grossman 2003).

Conservation efforts would be more efficient and successful if they were focused at the level of whole ecosystems and landscapes, rather than individual species (Wilcove 2010). It may not be possible to conserve all species given human population pressures and desires for economic growth, but effective conservation planning would protect as much biodiversity as possible given the constraints (Polasky 2006) by setting aside as much area as possible in anticipation of multiple threats (Heller and Zavaleta 2008). Although protecting wilderness is valuable and relatively easy, conserving the most biodiversity will require greater focus on those areas that are also of highest value to humans (Dobson et al 2001). The most effective measures will be those that integrate ecological principles into arguments based on the existing standards for comprehensive planning and subdivision review (Broberg 2003). These design criteria

include: maximizing the area of functional habitats, identifying and maintaining ecological process zones, maintaining key migration and source habitats, and managing human proximity and edge effects (Hansen and DeFries 2007). Managers also must deal with the dilemma that human actions taking place outside of their administrative jurisdiction influence ecological processes within their management unit (Hansen et al 2002). Linking state or regional conservation planning with local land use planning is one way to achieve a more comprehensive approach to habitat and biodiversity preservation (Grossman 2003). In California, land use planning on nonfederal lands is done on the local scale- most commonly through municipal and county general plans (Leppig and White 2006). Due to high levels of urbanization in the Los Angeles Basin, undeveloped lands occur mainly as small fragments (Vandergast et al 2008). Since, the dynamics of remnant areas are predominantly driven by factors arising in the surrounding landscape. There is a strong need to develop an integrated approach to landscape management that places conservation reserves in the context of the overall landscape (Saunders et al 1991). Some of the more general recommended strategies, such as removing other threats, increasing connectivity, and expanding reserve networks, are all based on a relatively simple understanding of species biology and historic climate change effects on species distributions (Hannah et al 2002). At the normative level, conservation planning requires effective and integrated land-use planning, policies, and legislation use to halt or prevent the degradation of environmental conditions at the relevant local scales to integrate sectoral plans and strategies and to ensure they are implemented (Pavlik 1997, Theobald et al 2000, Pierce et al 2005).

DRAFT

References

1. Abbitt, R.J.F., J.M. Scott, and D.S. Wilcove. 2000. The geography of vulnerability: incorporating species geography and human development patterns into conservation planning. *Biological Conservation* 96:169-175.

2. Abraham, J.K., J.D. Corbin, and C.M. D'Antonio. 2009. California native and exotic perennial grasses differ in their response to soil nitrogen, exotic annual grass density, and order of emergence. *Plant Ecology* 201:445–456.

3. Andelman, S.J. and W.F. Fagan. 2000. Umbrellas and flagships: Efficient conservation surrogates or expensive mistakes? *Proceedings of the National Academy of Sciences* 97(11): 5954–5959.

4. Bakker, E.S. 1984. An Island Called California: An Ecological Introduction To Its Natural Communities, 2nd edition. Berkeley, CA: University of California Press, 400 pp.

5. Barbour, M., B. Pavlik, S. Lindstrom, and F. Drysdale. 1993. *California's Changing Landscapes: Diversity and Conservation of California Vegetation*. Sacramento, CA: California Native Plant Society, 246 pp.

6. Bartolome, J.W., W.J. Barry, T. Griggs, and P. Hopkinson. Valley Grassland. pp. 367-393 *in* M.G. Barbour, T. Keeler-Wolf, and A. Schoenherr (Eds.). Terrestrial Vegetation of California, Third Edition. Berkeley, CA: University of California Press.

7. Beier, P., D.R. Majka, and S.L. Newell. 2009. Uncertainty analysis of least-cost modeling for designing wildlife linkages. Ecological Applications 19(8): 2067–2077.

8. Broberg, L. 2003. Conserving Ecosystems Locally: A Role for Ecologists in Land-Use Planning. *BioScience* 53(7):670-673.

9. California Department of Fish and Game. 2003. *Atlas of the Biodiversity of California*. Sacramento, CA: Resources Agency, 103 pp.

10. Collins, M.D., D.P. Vázquez, and N.J. Sanders. 2002. Species–area curves, homogenization and the loss of global diversity. *Evolutionary Ecology Research* 4:457–464.

11. Connor, E.F., and E.D. McCoy. 2001. Species-Area Relationships. *Encyclopedia of Biodiversity* 5:397-411.

12. Dasmann, R.F. 1970. *The Destruction of California*. New York, NY: MacMillan Company, 247 pp.

13. Dasmann, R.F. 1981. *California's Changing Environment*. San Francisco, CA: Boyd and Fraser Publishing Co., 77 pp.

DRAFT

14. Davis, E. B., Koo, M. S., Conroy, C., Patton, J. L, and Moritz, C. 2007. The California Hotspots Project: Identifying regions of rapid diversification of mammals. *Molecular Ecology* 17(1):120-138.

15. Diamond, J.M. 1976. Island biogeography and conservation: strategy and limitations. *Science* 193:1027–29.

16. Dobson, A.P., J.P. Rodríguez, and W.M. Roberts. 2001. Synoptic Tinkering: Integrating Strategies for Large-Scale Conservation. *Ecological Applications* 11:1019–1026.

17. Favreau, J.M., C.A. Drew, G.R. Hess, M.J. Rubino, F.H. Koch, and K.A. Eschelbach. 2006. Recommendations for assessing the effectiveness of surrogate species approaches. *Biodiversity and Conservation* 15:3949-3969.

18. Freudenberger, D.O., B.E. Fish, and J.E. Keeley. 1987. Distribution and Stability of Grasslands in the Los Angeles Basin. *Bulletin of the Southern California Academy of Sciences* (86)1:13-26.

19. Girvetz, E.H., J.H. Thorne, A.M. Berry, and J.A.G. Jaeger. 2008. Integration of landscape fragmentation analysis into regional planning: A statewide multi-scale case study from California, USA *Landscape and Urban Planning* 86: 205–218.

20. Grivet, D., V.L. Sork, R.D. Westfall, and F.W. Davis. 2008. Conserving the evolutionary potential of California valley oak (*Quercus lobata* Née): a multivariate genetic approach to conservation planning. *Molecular Ecology* 17: 139–156.

21. Grossman, E. (ed). 2003. Integrating Land Use Planning and Biodiversity. Washington, D.C.: Defenders of Wildlife, 62 pp.

22. Grubb, P.J. 1977. The Maintenance of Species–Richness in Plant Communities: The Importance of the Regeneration Niche. *Biological Reviews* 52:107-145.

23. Haire, S.L., Bock, C.E., Cade, B.S., Bennett, B.C. 2000. The role of landscape and habitat characteristics in limiting abundance of grassland nesting songbirds in an urban open space. *Landscape and Urban Planning* 48:65–82.

24. Hannah, L., Midgley GF, and Millar D. 2002. Climate change-integrated conservation strategies. *Global Ecology and Biogeography* 11:485–95.

25. Hansen, A.J., and R. DeFries. 2007. Ecological Mechanisms Linking Protected Areas To Surrounding Lands. *Ecological Applications* 17(4): 974–988.

26. Hansen, A.J., R. Rasker, B. Maxwell, J.J. Rotella, J.D. Johnson, A.W. Parmenter, U. Langner, W.B. Cohen, R.L. Lawrence, and M.P.V. Kraska. 2002. Ecological causes and consequences of demographic change in the new West. *BioScience* 52:151–162.

DRAFT

27. Heller, N.E., and E.S. Zavaleta. 2008. Biodiversity management in the face of climate change: A review of 22 years of recommendations. *Biological Conservation* 142:14–32.

28. Hickman, J.C. (Ed.). 1993. *The Jepson Manual: Higher Plants of California*. Berkeley, CA: University of California Press, 1424 pp.

29. Jantz, P.A., B.F.L. Preusser, J.K. Fujikawa, J.A. Kuhn, C.J. Bersbach, J.L. Gelbard, and F.W. Davis. 2007. Regulatory Protection and Conservation. pp. 298-318 *In* Stromberg, M.R., J.D. Corbin, and C.M. D'Antonio (Eds.). *California Grasslands: Ecology and Management*. Berkeley, CA: University of California Press.

30. Keeler-Wolf, T., J.M. Evens, A.I. Solomeshch, V.L. Holland, and M.G. Barbour. 2007. Community Classification and Nomenclature. pp. 21-34 *In* Stromberg, M.R., J.D. Corbin, and C.M. D'Antonio (Eds.). *California Grasslands: Ecology and Management*. Berkeley, CA: University of California Press.

31. Keeley, J.E. 1990. The California Grassland. pp. 1-23 *In* Schoenherr, A.A. *Endangered Plant Communities of California: Proceedings of the Southern California Botanists 15th Annual Symposium*. Claremont, CA: Rancho Santa Ana Botanic Gardens.

32. Kellogg, R. 1918. A revision of the *Microtus californicus* group of meadow mice. *University of California Publications in Zoology* 21:1–42.

33. Knops, J.M.H., J.R. Griffin, and A.C. Royalty. 1995. Introduced and Native Plants of the Hastings Reservation, Central Coastal Reservation: A Comparison. *Biological Conservation* 71:115-123.

34. Launer, A.E., and D.D. Murphy. 1994. Umbrella Species and the Conservation of Habitat Fragments: A Case of a Threatened Butterfly and a Vanishing Grassland Ecosystem. *Biological Conservation* 69:145-153.

35. Lee, T.M., and W. Jetz. 2008. Future battlegrounds for conservation under global change. *Proceedings of the Royal Society B* 275:1261–1270.

36. Leppig, G., and J.W. White. 2006. Conservation of Peripheral Plant Populations in California. *Madroño* 53(3): 264–274.

37. Loarie SR, Carter BE, Hayhoe K, McMahon S, Moe R, et al. (2008) Climate Change and the Future of California's Endemic Flora. *PLoS ONE* 3(6): e2502.

38. Lulow, M. 2006. Invasion by Non-Native Annual Grasses: The Importance of Species Biomass, Composition, and Time Among California Native Grasses of the Central Valley. *Restoration Ecology* 14(4):616–626.

DRAFT

39. Magee, T.K., P.L. Ringold, and M.A. Bollman. 2008. Alien species importance in native vegetation along wadeable streams, John Day River basin, Oregon, USA. *Plant Ecology* 195:287–307.

40. Magee, T.K., P.L. Ringold, M.A. Bollman, and T.L. Ernst. 2010. Index of Alien Impact: A Method for Evaluating Potential Ecological Impact of Alien Plant Species. *Environmental Management* 45:759–778.

41. Marcouiller, D.W., and D. Tremble. 2009. Mitigating Environmental Problems in Exurban Development: An Overview of Rural-Specific Planning Devices. Pp.235-253 *in* A.X. Esparza, G. McPherson (eds.), *The Planner's Guide to Natural Resource Conservation.* NY, NY: Springer Science+Business Media, LLC.

42. Margules, C.R., and R.L. Pressey. 2000. Systematic conservation planning. *Nature* 405:249-253.

43. Matocq, M.D., 2002. Phylogeographical structure and regional history of the dusky-footed woodrat, *Neotoma fuscipes*. *Molecular Ecology* 11:229–242.

44. May, R. 1975. Island biogeography and the design of wildlife preserves. *Nature* 254:177–78.

45. McCann, K.S. 2000. The diversity–stability debate. *Nature* 405:228-233.

46. McDonald, R.I., P. Kareiva, and R.T.T. Forman. 2008. The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biological Conservation* 141: 1695–1703.

47. McKinney, M.L. & Lockwood, J.L. 1999. Biotic homogenization: a few winners replacing many losers in the next mass extinction. *Trends in Ecology and Evolution* 14, 450–453.

48. McNaughton, S. J. 1968. Structure and Function in California Grasslands. *Ecology* 49(5):962-972.

49. Minnich, R.A. 2008. *California's Fading Wildflowers: Lost Legacy and Biological Invasions*. Berkeley, CA: University of California Press, 344 pp.

50. Mooney HA, J. Lubchenco, R. Dirzo, and O.E. Sala. 1995. Biodiversity and ecosystem functioning: Basic principles. *In*: Heywood, V.H. (ed.) *Global Biodiversity Assessment*. UN Environmental Programme; Cambridge University Press, pp. 275-326.

51. Olden, J.D. 2006. Biotic homogenization: a new research agenda for conservation biogeography. *Journal of Biogeography* 33:2027–2039.

52. Pavlik, B.M. 1997. Perspectives, Tools and Institutions for Conserving Rare Plants. *The Southwestern Naturalist* 42(4):375-383

DRAFT

53. Pavlik, B.M. 2003. Plants that protect ecosystems: a survey from California. *Biodiversity and Conservation* 12:717-729.

54. Pavlik, B.M. 2008. *The California Deserts: An Ecological Rediscovery*. Berkeley, CA: University of California Press, 365 pp.

55. Pierce, S.M., R.M. Cowling, A.T. Knight, A. T. Lombard, M. Rouget, and T. Wolf. 2005. Systematic conservation planning products for land use planning: interpretation for implementation. *Biological Conservation* 125:441–458.

56. Pincetl, S.S. 1999. *Transforming California: A Political History of Land Use and Development*. Baltimore, MD: Johns Hopkins University Press, 372 pp.

57. Polasky, S. 2006. You can't always get what you want: Conservation planning with feedback effects. *Proceedings of the National Academy of Sciences* 103(14):5245–5246.

58. Preston, F.W. 1948. The Commonness, And Rarity, of Species. *Ecology* 29(3):254-283.

59. Preston, W. 1997. Serpent in the Garden: Environmental Change in Colonial California. *California History* 76:260-298.

60. Radeloff, V.C., R.B. Hammer, S.I. Stewart, J.S. Fried, S.S. Holcomb, and J.F. McKeefry. 2005. The Wildland-Urban Interface in the United States. *Ecological Applications* 15(3): 799–805.

61. Raven, P. H. and D. I. Axelrod. 1978. Origin and relationships of the California flora. *University of California Publications in Botany* 72:1–134.

62. Rissler, L.J., R.J. Hijmans, C.H. Graham, C. Moritz, and D.B. Wake. 2006. Phylogeographic Lineages and Species Comparisons in Conservation Analyses: A Case Study of California Herpetofauna. *The American Naturalist* 167(5): 655-666.

63. Roberge, J-M., and P. Angelstam. 2004. Usefulness of the Umbrella Species Concept as a Conservation Tool. *Conservation Biology* 18(1): 76-85.

64. Rooney, T.P., J.D. Olden, M.K. Leach, and D.A. Rogers. 2007. Biotic homogenization and conservation prioritization. *Biological Conservation* 134:447–450.

65. Rutledge, D.T., C.A. Lepczyk, J. Xie, and J. Liu. 2001. Spatiotemporal dynamics of Endangered Species Hotspots in the United States. *Conservation Biology* 15(2):475-487.

66. Sarkar, S., R.L. Pressey, D.P. Faith, C.R. Margules, T. Fuller, D.M. Stoms, A. Moffett, K.A. Wilson, K.J. Williams, P.H. Williams, and S. Andelman. 2006. Biodiversity Conservation Planning Tools: Present Status and Challenges for the Future. *Annual Review of Environment and Resources* 31:123-159.

DRAFT

67. Saunders, D.A., R.J. Hobbs, and C.R. Margules. 1991. Biological Consequences of Ecosystem Fragmentation: A Review. *Conservation Biology* 5(1): 18-32.

68. Scheiner, S.M., S.B. Cox, M. Willig, G.G. Mittelbach, C. Osenberg, and Michael Kaspari. 2000. Species richness, species–area curves and Simpson's paradox. *Evolutionary Ecology Research* 2:791–802.

69. Schoenherr, A.A. 1992. *A Natural History of California*. Berkeley, CA: University of California Press, 772 pp.

70. Schwartz, M.W., J.H. Thorne, and J.H. Viers. 2006. Biotic homogenization of the California flora in urban and urbanizing regions. *Biological Conservation* 127: 282–291.

71. Seabloom, E.W., J.W. Williams, D. Slayback, D.M. Stoms, J.H. Viers, and A.P. Dobson. 2006. Human Impacts, Plant Invasion, and Imperiled Plant Species in California. *Ecological Applications* 16:1338–1350.

72. Shmida, A. and M.V. Wilson. 1985. Biological determinants of species diversity *Journal of Biogeography* 12:1-20.

73. Spellerberg, I.F. and P.J. Fedora. 2003. A tribute to Claude Shannon (1916–2001) and a plea for more rigorous use of species richness, species diversity and the 'Shannon–Wiener' Index. *Global Ecology & Biogeography* 12:177–179.

74. Theobald, D. M., N. T. Hobbs, T. Bearly, J. A. Zack, T. Shenk, and W.E. Riebsame. 2000. Incorporating biological information in local land use decision making: designing a system for conservation planning. *Landscape Ecology* 15:35–45.

75. Tilman, D. and J.A. Downing. Biodiversity and Stability in Grasslands. *Nature* 367:363-365.

76. Tilman, D., P.B. Reich, and J.M.H. Knops. 2006. Biodiversity and ecosystem stability in a decade-long grassland experiment. *Nature* 441:629-632.

77. Tilman, D., D. Wedin and J. Knops. 1996. Productivity and Sustainability influenced by Biodiversity in Grassland Ecosystems. *Nature* 379:718-720.

78. Tylianakis, J.M., R.K. Didham, J. Bascompte, and D.A. Wardle. 2008. Global change and species interactions in terrestrial ecosystems. *Ecology Letters* 11:1351–1363.

79. Tylianakis, J.M., E. Laliberté, A. Nielsen, and J. Bascompte. 2010. Conservation of species interaction networks. *Biological Conservation* 143:2270–2279.

80. Vandergast, A.G., A.J. Bohonak, S.A. Hathaway, J. Boys, and R.N. Fisher. 2008. Are hotspots of evolutionary potential adequately protected in southern California? *Biological Conservation* 141: 1648–1664.

DRAFT

81. Wilcove, D. 2010. Endangered species management: the US experience *in* Sodhi, N.S. and P. R. Ehrlich (eds.) *Conservation Biology for All*. New York, NY: Oxford University Press Inc., pp. 220-235.

DRAFT