Chapter 12
Chaparral Landscape Conversion in Southern California

Alexandra D. Syphard, Teresa J. Brennan, and Jon E. Keeley

Southern California, where the American Dream came too true

–Lawrence Ferlinghetti

Abstract The low-elevation chaparral shrublands of southern California have long been occupied and modified by humans, but the magnitude and extent of human impact has dramatically increased since the early 1900s. As population growth started to boom in the 1940s, the primary form of habitat conversion transitioned from agriculture to urban and residential development. Now, urban growth is the primary contributor, directly and indirectly, to loss and fragmentation of chaparral landscapes. Different patterns and arrangements of housing development confer different ecological impacts. We found wide variation in the changing extent and pattern of development across the seven counties in the region. Substantial growth in lower-density exurban development has been associated with high frequency of human-caused ignitions as well as the expansion of highly flammable non-native annual grasses. Combined, increases in fire ignitions and the extent of grassland can lead to a positive feedback cycle in which grass promotes fire and shortens the fire-return interval, ultimately extirpating shrub species that are not adapted to short fire intervals. An overlay of a 1930s vegetation map with maps of contemporary vegetation showed a consistent trend of chaparral decline and conversion to sage scrub or

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grassland. In addition, those areas type-converted to grassland had the highest fire frequency over the latter part of the twentieth century. Thus, a continuing trend of population growth and urban expansion may continue to threaten the extent and intactness of remaining shrubland dominated landscapes. Interactions among housing development, fire ignitions, non-native grasses, roads, and vehicle emissions make fire prevention a complex endeavor. However, land use planning that targets the root cause of conversion, exurban sprawl, could address all of these threats simultaneously.

Keywords Chaparral · Fire · Housing development · Land use change · Non-native species · Vegetation change

12.1 Introduction

For thousands of years, humans have occupied the vast shrublands blanketing the foothills and mountains of southern California. Native Americans altered their environment to protect and sustain themselves, particularly via controlled burning to open up shrubland landscapes (see Chap. 4). Subsequently, the arrival of Euro-American settlers in the late eighteenth century brought about a sequence of progressively intense phases of rapid population growth and landscape conversion. The California Gold Rush and statehood brought one of the first population booms in 1850, and shortly thereafter, the region was linked to the railroad, enabling faster and safer immigration to the region from the rest of the country. Transportation via automobile soon became possible in the early 1900s, which facilitated even more immigration; plus, it enabled the beginning of suburban development outside of the region's main urban centers, such as Los Angeles and San Diego.

Throughout the progression of the twentieth century, southern California has continued to offer a wide range of economic opportunities. When coupled with the mild Mediterranean-type climate, these have made the region one of the most desirable places to live in the US. In particular, people flocked to the region with the discovery of oil at the turn of the century, which was then followed by growth in numerous other industries, including military defense production, agriculture, and the film industry. In the middle of the century, human population growth exploded; the accompanying massive change in land use dramatically altered the extent and composition of the native vegetation communities in the region. Although large expanses of native shrublands still exist in many areas, southern California has come to be viewed by the world as the land of freeways, strip malls, and endless housing developments.

In this chapter, we explore the trends and drivers of vegetation change in southern California since the early 1900s. In particular, we focus on the interactions between direct habitat conversion through urban growth and indirect changes brought on by non-native annual grasses, increased frequency of fires, and the resultant loss of native shrublands.

12.2 Habitat Conversion

12.2.1 Overview of Land Use Change

By the middle Holocene, Indian populations dominated much of coastal California, and they had a significant impact on landscape patterns through repeated burning and displacement of chaparral with more productive herbaceous communities (see Chap. 4). In the late eighteenth century, Spanish settlements initiated a new wave of changes with the introduction of a wide selection of non-native annual grasses and forbs (Mack 1989). The economy of these early settlements was based on cattle production, and the Mexican vaqueros would often burn off shrublands to increase grazing lands (Kinney 1887). Ever since then, rangeland management has had a significant component of repeated burning of shrublands to increase forage for livestock (Keeley and Syphard 2018).

One of the most significant changes in plant community composition with Euro-American settlement was the replacement of native vegetation with non-native grassland. As a result, non-native annuals were likely a large component of California grasslands by the 1850s (Burcham 1956). Livestock grazing undoubtedly has contributed substantially to this shift (D'Antonio et al. 1992), often in combination with severe droughts (Burcham 1956). Nevertheless, even in the absence of grazing, non-native annuals introduced by Euro-American settlers likely outcompeted native bunch grasses (Bartolome and Gemmill 1981). Intentional conversion of shrublands to create grassland for grazing was common across California (Burcham 1956; Keeley and Fotheringham 2003). Similar patterns of type-conversion have occurred over the 10,000-year history of human occupation in the Mediterranean Basin, where transitions from woody to herbaceous species have also been caused by human disturbance via livestock grazing and accelerated burning due to anthropogenic ignitions. However, in California, this loss of woody cover degrades natural systems and diminishes their conservation value by displacing native flora with non-native species. In the Mediterranean Basin, type-conversion replaces woody natives with herbaceous natives, and thus, native biodiversity increases.

In the early twentieth century, conversion of natural habitat into agricultural lands was the most dominant form of land use change, and by the 1930s, approximately 20% of the land within the South Coast Ecoregion had become croplands, with citrus and other fruit trees becoming especially extensive. At this time, southern California was considered one of the top agricultural regions in the US. However, with population growth and evolving economic opportunities, farming was largely wiped out in the middle of the century in favor of commercial and residential development, a trend that was common nationwide (Alig and Plantinga 2004). In the South Coast Ecoregion of California, less than five percent of the croplands mapped in the 1930s were still present by the early 2000s (derived from data described in next section).

Urban and residential development is now the top contributor to both direct and indirect habitat conversion in southern California. Not only have the major metropolitan areas become denser, but the freeway system developed in the 1940s initiated what has been an ongoing trend of "sprawl" outward from coastal cities into the inland foothills and mountains. This growth was so rapid and extensive that the San Fernando Valley outside of Los Angeles took on the name of "America's Suburb" (Roderick 2002). Across the world, southern California is still perceived as synonymous with urban sprawl.

12.2.2 Spatial and Temporal Patterns of Housing Growth

The spatial pattern of housing development has important implications for landscape conversion because low-density, sprawling-type development typically consumes more land and wildlife habitat than high-density development (Odell et al. 2003). As a consequence, low-density development may have a more negative impact on biodiversity and ecosystem services (Hansen et al. 2005). On the other hand, higher-density, clustered development may be more ecologically degraded with a larger dominance of non-native species (Lenth et al. 2006). Despite these trade-offs, compact urban development has been shown to minimize ecological disruption relative to sprawling development (Sushinsky et al. 2013).

The term wildland-urban interface (WUI) has emerged in the last couple of decades to describe the characteristics and social-ecological effects of those areas where housing development is adjacent to or interspersed with wildland vegetation (Radeloff et al. 2005). Two types of WUI are typically defined, largely as a function of housing density and the extent to which houses are surrounded by wildland vegetation. The "interface WUI" describes those areas where human settlements are denser and form an edge with wildland vegetation, whereas "intermix WUI" reflects areas where sparser, lower-density housing is interspersed with wildland vegetation. Although the exact definition of intermix or interface WUI may vary slightly with regards to how it is mapped (Stewart et al. 2007), these terms have provided a useful framework for understanding how and where human settlements interact with the natural environment, and how different forms of development may differentially affect habitat change and ecological impacts (Bar-Massada et al. 2014).

The spatial pattern of urban development in any given area can vary dramatically over time, but it typically emerges as a result of different characteristic growth types (Herold et al. 2003; Dahal et al. 2017). At one end, compact and high-density development patterns usually result from infill-type growth, where new structures are built within or expand outward from existing urban areas. At the other end, low-density, fragmented, exurban development patterns result from leapfrog-type growth in which new development occurs outside of urban areas and is typically surrounded by wildland vegetation. This lower-density exurban development, characteristic of the intermix WUI, is often the result of homeowner preferences and behaviors, including a desire to live closer to natural amenities (Netusil 2005) or lower land prices at greater distances from the urban core (Wu and Plantinga 2003).

Given the importance of both spatial extent and pattern of housing growth in terms of natural habitat conversion, we quantified historical housing trends in the South Coast Ecoregion from 1940 to 2010. To do this, we evaluated historical housing density maps (Hammer et al. 2004, available at http://silvis.forest.wisc.edu/maps/housing) within the footprint of a modified South Coast Ecoregion boundary (i.e., as in Syphard et al. 2011) that includes the full extent of the Los Padres National Forest. The maps were developed as part of a national data product in which housing density was mapped within partial census block groups and reported as housing units per square kilometer.

We quantified the extent of both low- and medium- to high-density housing from 1940 to 2010 within the seven counties that are located within the ecoregion. Instead of clipping the counties to the ecoregion boundary, we assessed housing growth for the complete extent of each county. To threshold the continuous housing data into classes of low- and medium-high-density, we selected all areas with a housing density between 6.17 and 49 houses per km^2 and classified them as "low density." The number 6.17 corresponds to the minimum housing density cutoff for defining low-density WUI (Radeloff et al. 2005). The threshold of ≥50 houses per square kilometer corresponds to those areas defined as medium- or high-density WUI. For each county in each decade, we summarized the total extent of each housing density type and calculated its proportion of the county area.

In all seven counties, housing development, and hence direct habitat conversion, increased substantially from 1940 to 2010 across the region, but the extent of development and pattern of housing growth varied over time and by county (as can be seen in the widely varying range of the Y axis in Fig. 12.1). Medium- to high-density development has dominated the counties closest to Los Angeles, but low-density housing growth has predominated in San Luis Obispo, Riverside, and San Diego counties (Figs. 12.1 and 12.2). Except for Los Angeles, which exhibited slow, steady growth in both housing-density types over time, a pulse in growth was apparent during and shortly after the 1990s for the other counties, which is consistent with nation-wide trends (Glaeser and Shapiro 2003). Orange County stands out in that, as medium-high density increased over time, low-density development has shown a slight decline across most of the record. This also has been evident in recent decades for Santa Barbara and San Luis Obispo Counties, suggesting that, in addition to urban expansion, existing urban areas in these counties may also be infilling and becoming denser. The two southern-most counties (San Diego and Riverside), on the other hand, show no sign of slowing in the expansion of low-density development.

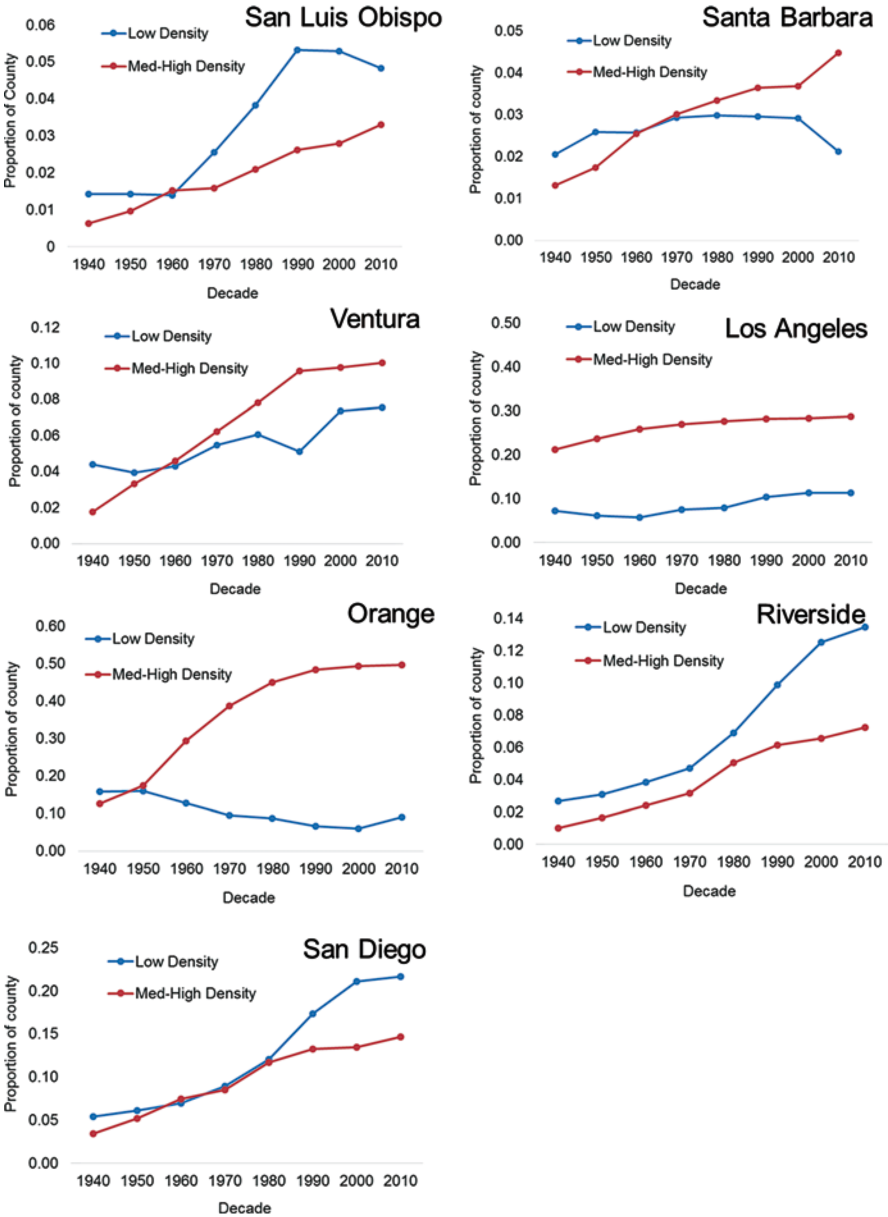


Fig. 12.1 Growth in area (proportion of county) of low and medium-high housing density from 1940 to 2010 within the full boundaries of the seven counties overlapping the South Coast Ecoregion of southern California

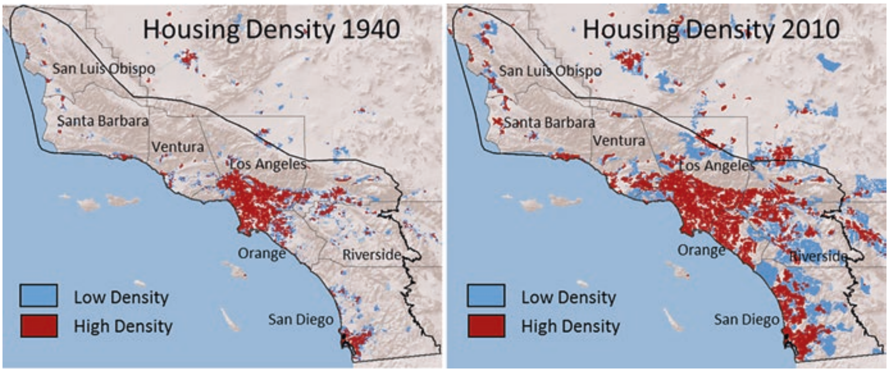


Fig. 12.2 Maps of low and medium-high housing density in 1940 and 2010 in the South Coast Ecoregion of southern California

12.2.3 Indirect Habitat Loss and Conversion

In addition to causing direct conversion of native habitat, housing development in southern California indirectly contributes to chaparral conversion, primarily by facilitating an interaction between increased fire frequency and the expansion of weedy non-native annual grasslands.

Although periodic wildfire is an essential component of chaparral ecosystems, fire frequency has been increasing in southern California to the extent that most of the landscape is burning at fire-return intervals (i.e., the time between fires at a defined area) that are uncharacteristically short relative to pre-EuroAmerican settlement conditions (Safford and Van de Water 2014). In some areas, fires are now so frequent that they outpace the historical baseline by a wide margin; for example, return intervals that once averaged 30 to more than 150 years are now shorter than 10 years in some areas (Keeley and Syphard 2018).

Given that humans cause more than 95% of the fires in the region (Syphard et al. 2007), the trend of increasing fire frequency is primarily explained by population growth and expansion of development into wildland vegetation. Although human-caused fires generally increase with human population, this relationship is mediated by population or housing density. That is, across California and other Mediterranean-type climate regions, studies show that the ignition frequencies tend to peak at low-intermediate population density, such as the WUI intermix areas (Syphard et al. 2007, 2009; Archibald et al. 2010; Syphard and Keeley 2015). The likely explanation for this is that these intermix WUI areas have both enough people to start frequent fires, which wild areas lack, and sufficient wildland areas to facilitate fire spread, which urban areas lack. These are also the areas most difficult to access for fire suppression (Gude et al. 2008).

In addition to increased fire frequency, exurban development provides conduits for non-native species to expand into wildland vegetation, via land disturbance, road networks, and residential landscaping (Gavier-Pizarro et al. 2010). Even fuelbreaks designed to control wildfires facilitate establishment and spread of non-native species (Merriam et al. 2006). In recent studies, which assessed the ecological effects of mechanical fuel treatments on chaparral (such as removing vegetation with bulldozers), it was found that treated sites had a significantly lower cover and density of shrubs and a significantly higher cover and density of herbaceous plants (Brennan and Keeley 2015). The increase in herbaceous plants was dominated by non-native species and in particular by non-native annual grasses. Sites that were treated a second time had more than twice the cover and density of non-native species than single treatments and were clearly showing more signs of degradation and type-conversion, that is, a shift in physiognomic structure from woody shrubland to herbaceous cover. These treatments are frequently used near housing developments within the WUI; and over time, with periodic retreatment, will most likely be completely type-converted to non-native annual grasslands.

The weedy annual grasses that have invaded vast portions of southern California are highly flammable and tolerant of rapidly repeating fires. In the absence of disturbance, chaparral shrublands are relatively resistant to invasion by non-native species, in part due to their dense cover and closed canopy. However, increased human ignitions in these fire-prone grasslands has lengthened the fire season, thereby increasing canopy opening and providing new establishment opportunities for these well-dispersed grasses. This positive feedback process between fires and grass expansion is typically referred to as a grass-fire cycle, and it is recognized as a potential problem in ecosystems across the world (e.g., D'Antonio and Vitousek 1992; Rossiter et al. 2003; Brooks et al. 2004; Bowman et al. 2014), including southern California shrublands (Keeley et al. 2012).

The larger ecological issue is that, despite native shrublands' resilience to periodic wildfire, too-short intervals between fires can lead to their extirpation. This is because many species require a minimum amount of time between fires to recover and regenerate. Non-resprouting species—i.e., obligate seeders—may require up to 25 years to fully establish a seedbank that can effectively recruit new plants after fire (Keeley 1986). Although re-sprouting chaparral species are resilient to shorter intervals between fires than non-re-sprouters, even re-sprouters were reduced when multiple fires occurred within in a six-year interval (Haidinger and Keeley 1993). Thus, as native shrubland species are extirpated, providing opportunities for further grass expansion, the potential exists for large scale vegetation type-conversion.

A number of studies in southern California have provided evidence of vegetation type-conversion from shrubland to grassland. Particularly widespread has been the conversion of coastal sage scrub to non-native grasses (Minnich and Dezzani 1998; Cox et al. 2014). Talluto and Suding (2008) found nearly 50% replacement of sage scrub by annual grasses within a 76-year study period in parts of Orange and Riverside Counties, with a substantial amount being due to fire frequency. Because sage scrub is generally more tolerant of higher fire frequencies than chaparral, chaparral may be even more vulnerable to vegetation type-conversion, depending on species composition and site factors. In some cases, it may even transition to sage scrub vegetation before finally transitioning to herbaceous cover (Syphard et al. 2006).

Chaparral conversion to grasslands after repeated fires has been documented in many localized studies (e.g., Zedler et al. 1983; Haidinger and Keeley 1993; Lippitt et al. 2012; Keeley and Brennan 2012). Given the consistency in these findings across the southern California region, and the fact that large areas across the region have experienced short fire-return intervals, there is reason to suspect that widespread conversion due to repeated fires has already occurred (Keeley 2010). Nevertheless, the empirical evidence for larger landscape scale changes in chaparral has been sparse, with one recent study even questioning the potential for widespread vegetation type change in chaparral to occur (Meng et al. 2014).

12.3 Landscape Scale Vegetation Type-Conversion

As a general means of quantifying historical vegetation change in concert with mean historical fire frequency in southern California, we overlaid contemporary maps of existing vegetation with an historical map delineating broad scale vegetation types and then integrated data on fire frequency. We estimated change using maps from multiple data sources because of the potential for vegetation to be mapped differently. Although variation is much more likely given finer scale vegetation classification schemes, there may even be differences in the way broad vegetation types are mapped due to differences in mapping methods, scales, and definitions.

The historical vegetation type maps (VTM) were developed between the years 1929 and 1934 (Wieslander 1935) as part of an extensive statewide mapping project. In addition to detailed species level plot information, vegetation types and dominant species were mapped on 15-minute topographic quadrangles in the field with a minimum mapping unit of 16 ha (39.5 acres) (Kelly et al. 2005; Kelly 2016). The first contemporary map we evaluated represents existing vegetation and was produced by the US Forest Service using a combination of satellite imagery, field verification, and expert guidance (CalVeg, http://www.fs.fed.us/r5/rsl/projects/classification/system.shtml). The majority of the area in this map was most recently updated in 2002. However, the national forest lands were updated more recently, in 2003, 2009, or 2010. The entire region was mapped at a scale of 1:24,000.

Both the VTM and CalVeg maps provide classification according to the California Wildlife Habitat Relationships System (Mayer and Laudenslayer 1988). Therefore, for both of these maps, we grouped vegetation classes into life-forms, including tree, shrub, coastal sage scrub, and herbaceous. For the other categories, which are mostly unvegetated (e.g., urban/developed land) or wetland, we lumped them into a class named “other.”

We also evaluated the 2013 Landfire existing vegetation maps, which were developed based on a combination of decision tree models, field data, Landsat 7 imagery, elevation, and biophysical gradient data (http://landfire.cr.usgs.gov/viewer/ [2013, May 8]). The map comes as a grid at 30 m (0.2 acres) resolution. We developed map classes to match the vegetation types in the other two maps using the map

Table 12.1 Proportion of vegetation types within the historical (VTM) and contemporary (San Diego County, CalVeg, and Landfire) maps

Vegetation Type	VTM	San Diego	CalVeg	Landfire
Grass	0.06	0.09	0.12	0.27
Sage scrub	0.29	0.24	0.10	0.06
Shrubland	0.45	0.35	0.43	0.19
Tree	0.08	0.09	0.12	0.18
Other	0.13	0.23	0.23	0.29
Total	1.0	1.0	1.0	1.0

attribute based on the National Vegetation Classification System Physiognomic Order. Any area that was classified as "sparsely vegetated," "barren," "water," "developed," or "agriculture" in the Landfire vegetation type classification, we converted to the "other" class.

At a finer resolution for a subset of the South Coast Ecoregion, we compared the VTM map to a detailed 2012 vegetation community map that spans part of San Diego County (https://databasin.org/datasets/bcd5db8e6aa540e6b06a371bde0afde3). This map was developed with a 1 ha (2.5 acre) minimum mapping unit for terrestrial vegetation and has an accuracy of at least 80% as determined through extensive field verification reports. The map was classified according to Sproul et al. (2011), and again, we grouped these into the same life-form vegetation classes and an "other" class.

After re-classifying the vegetation maps into physiognomic types, we quantified the proportion of each vegetation or cover type within each map. We then overlaid the contemporary maps with the VTM map and summarized the mean historical fire frequency that occurred within each change class up to 2013. To estimate the transitions between life-form classes, we assessed changes from shrub to grass, sage scrub to grass, shrub to sage scrub, tree to earlier successional class (shrub, sage scrub, or grass), successional (e.g., grass to sage scrub, sage scrub to shrub, shrub to tree), no change in vegetation, or other (i.e., unvegetated in either map. We used the California Department of Forestry–Fire and Resource Assessment Program (CDF-FRAP 2013) map of overlapping historical fire perimeters (wildfire only) to create a continuous 30 m grid with each cell representing the number of times it had burned since 1878 (http://frap.fire.ca.gov/data/frapgisdata-sw-fireperimeters_download). In this database, any grid cell location may have burned 0–13 times during the time period, although this may under-estimate fire frequency due to the minimum mapping unit of this dataset (Syphard and Keeley 2017).

The contemporary vegetation maps showed consistent trends of increasing grass, tree, and other cover types and decreasing sage scrub and shrubs over time (Tables 12.1 and 12.2, Fig. 12.3). There were substantial areas of agreement in the delineation of all vegetation types that did not change between the VTM map and contemporary maps (Figs. 12.4 and 12.5), particularly in CalVeg and the higher-resolution San Diego County map. The Landfire map, however, delineated a much larger pro-

Table 12.2 Proportion of chaparral in the historical (VTM) map that transitioned to other vegetation types in contemporary (San Diego County, CalVeg, and Landfire) maps

Chaparral Change Class	San Diego	CalVeg	Landfire
Chaparral to chaparral	0.33	0.22	0.24
Chaparral to sage scrub	0.07	0.22	0.26
Chaparral to grass	0.12	0.10	0.26
Chaparral to tree	0.40	0.27	0.17
Chaparral to other	0.09	0.20	0.07
Total	1.0	1.0	1.0

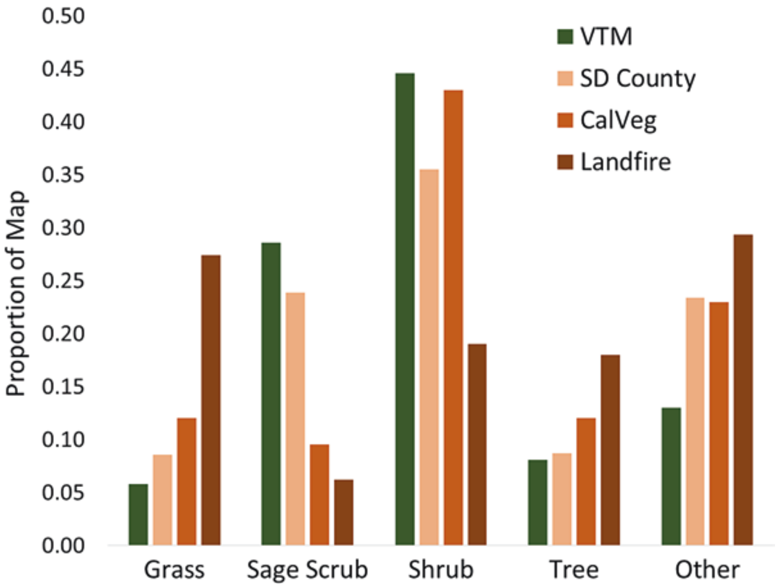


Fig. 12.3 Proportion of vegetation type within four vegetation maps of the South Coast Ecoregion (VTM 1930s; CalVeg 2002; Landfire 2013) and San Diego County (2012)

portion of the landscape as grassland than the other two contemporary maps (Fig. 12.4c). This is reflected in the vast areas of the landscape that were mapped as having changed from sage scrub or shrub to grass (Fig. 12.6a).

In terms of fire frequency, the analysis showed highest mean fire frequencies in classes where either sage scrub or shrub converted to grass, or where shrub converted to sage scrub (Fig. 12.7). The mean number of fires summed across grid cells in each change class ranged from 1.55 to 2.41, but the actual number of times areas burned during the 135-year span of the fire history data ranged from 0 to13.

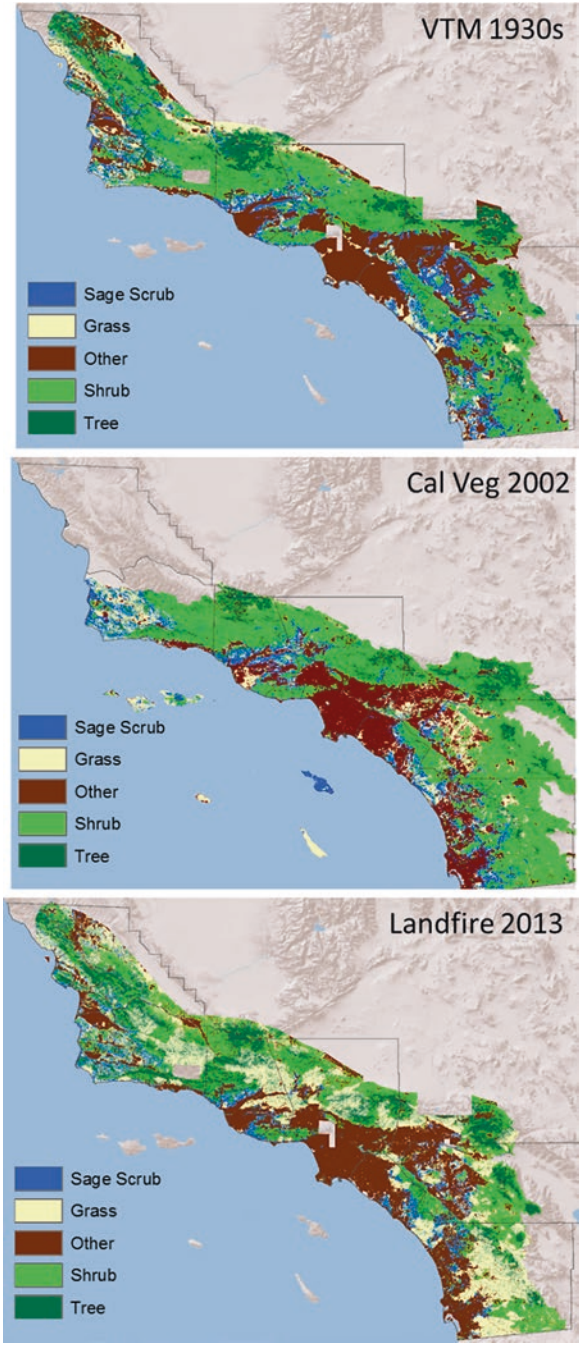


Fig. 12.4 Vegetation types as mapped in the 1930s (VTM), 2002 (CalVeg maps), and 2013 (Landfire)

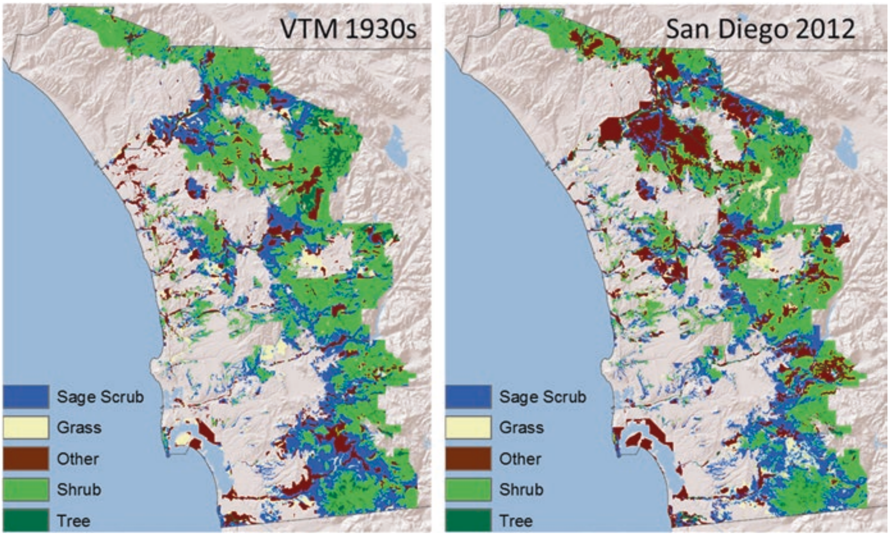


Fig. 12.5 Comparison of vegetation type classes as mapped in the 1930s (VTM) and in 2012 (San Diego County map)

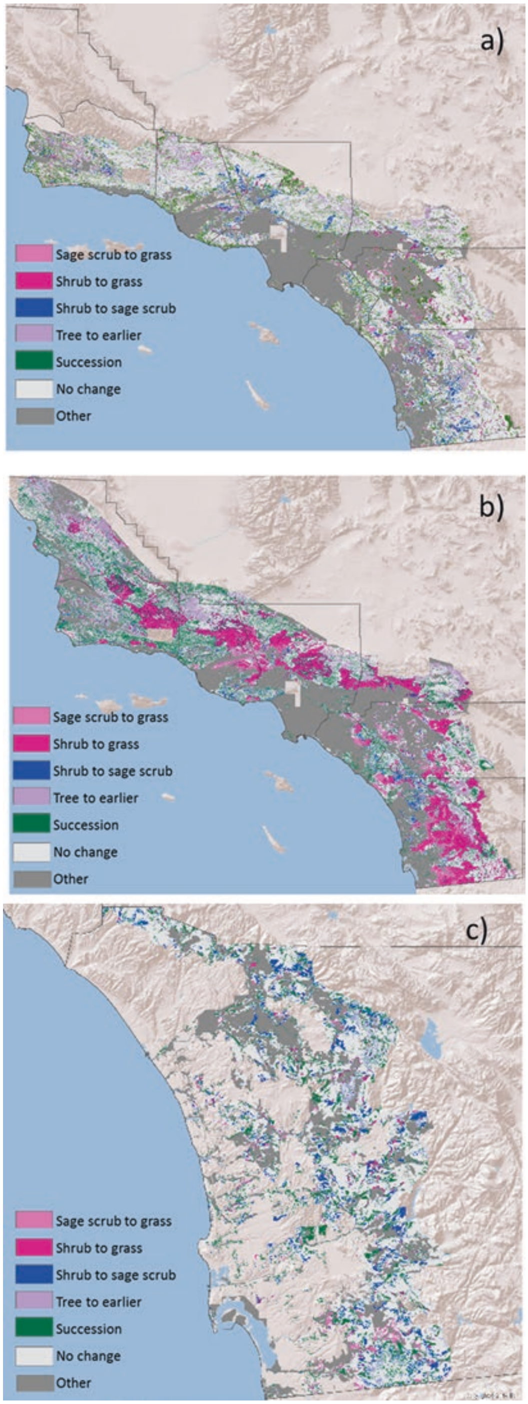
12.3.1 Challenges in Quantifying Vegetation Change

The wide variation apparent among the three contemporary maps illustrates the challenge in overlapping different vegetation maps to accurately delineate and quantify vegetation change, particularly if the objective is to map change at a fine scale. There are multiple sources of uncertainty inherent in any ecological analysis (Regan et al. 2002), and spatial data are particularly susceptible to errors in map boundaries and classification (Goodchild and Gopal 1989). Nevertheless, when vegetation map classes are collapsed into broad categories reflecting vegetation formations, map accuracy can be relatively high (Goodchild et al. 1991).

Clearly, the extent and location of vegetation type-conversion cannot be precisely determined from our analysis, and the vast areas of type change from shrub or scrub to grass mapped using the Landfire data should be interpreted with some caution given that many of these areas were not mapped as grass in the other two contemporary maps. Nevertheless, despite the variation among contemporary maps, the results of all three overlays were remarkably consistent in the kind of change measured. Thus, even using the most conservative estimates, there has been a clear trend of chaparral decline and conversion to either sage scrub or grassland over the last 70–80 years. Furthermore, fire frequency tends to be highest where these changes have been mapped (Fig. 12.7).

In the southern California landscape, the most likely explanation for the differences in maps is the treatment of mixed classes. Depending on the scale of the analysis relative to the heterogeneity of the vegetation, mixed grass and shrub stands must often be lumped into one class or the other. Thus, many of the areas mapped

Fig. 12.6 Maps of vegetation type change from (**a**) the 1930s to 2002 (CalVeg), (**b**) the 1930s to 2013 (Landfire), and (**c**) the 1930s to 2012 (San Diego County map)



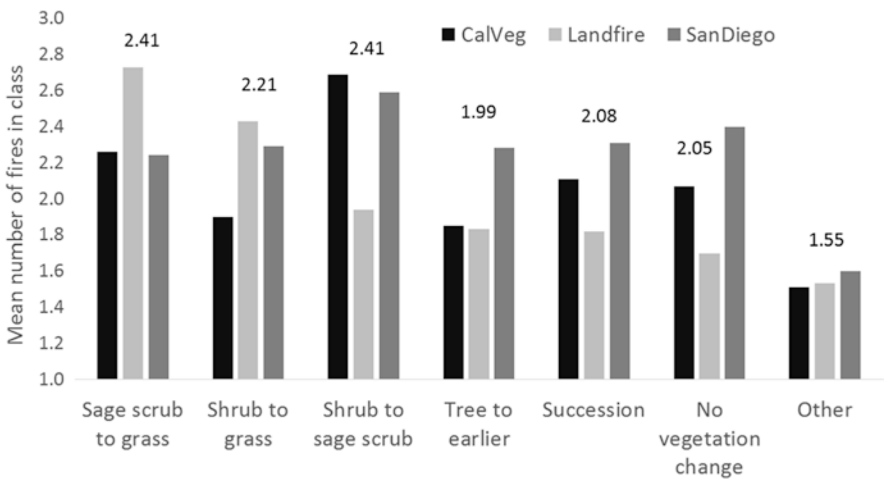


Fig. 12.7 Mean number of fires from 1878–2013 within each vegetation type change class between the 1930s and 2002, using data from CalVeg (2002), Landfire (2013), and San Diego County maps (2012). Numbers above the bars indicate the mean fire frequency averaged across the three maps. "Tree to other" reflects any changes in which trees changed to shrub, sage scrub, or grass. "Succession" reflects any changes in which grass changed to shrub or sage scrub, or sage scrub changed to shrub

as grass in the Landfire map, and mapped as some type of shrubland in the CalVeg or San Diego County map, were probably some mixture of shrub and grass.

Whether these classes were purely grass or represented some mixture with shrubs is one of the central challenges in quantifying landscape scale vegetation change. It also provides one reason for questioning Meng et al.'s (2014) conclusion that widespread vegetation type-conversion is not an immediate threat in southern California, as vegetation type-conversion does not occur as a complete shift at one moment in time. Instead, it occurs as a gradual and cumulative process, which often begins with the elimination of non-resprouting species within mixed stands, habitat simplification, and biodiversity loss (Keeley et al. 2005). In addition, because sage scrub can withstand higher fire frequencies than chaparral, vegetation change may begin with a gradual shift from stands of pure chaparral to mixed stands of chaparral, sage scrub, and grass. This type of transition is suggested in the results here that show substantial change from shrub to sage scrub under higher mean fire frequencies. Given that different species have varying sensitivities to repeat fires, and that overlapping fires exhibit fragmented spatial patterns, multiple repeat fire events are probably necessary for significant vegetation change to be discernable. Thus, one of the methodological challenges in landscape scale analyses like those in Meng et al. (2014) is that type-conversion is only inferred, and the gradual process of vegetation change cannot be documented at a specific location over time the way it can in field studies (Halsey and Syphard 2015). Another challenge is that substantial chaparral conversion had already occurred before vegetation maps became available for modern analysis. There is evidence of chaparral conversion prior to the twentieth century (Cooper 1922), and evidence has also been documented in field studies. In summary, vegetation change is complex, gradual, and related to site factors in addition to long-term fire history and plant community composition. These factors need to be resolved and better understood when considering the potential for future chaparral conversion.

12.4 Ecological and Social Consequences of Chaparral Loss

In addition to the loss of plant biodiversity that occurs with habitat conversion, many rare and sensitive animal species depend on vegetation structure for their habitat (see Chaps. 2 and 3). The native coastal sage and chaparral shrublands, as well as riparian areas and oak woodlands, provide important habitat for a wide range of bird, insect, mammal, and herpetofauna species, and the negative effects of habitat loss and fragmentation have been documented for decades in numerous studies (e.g., Bolger 1991; Soulé et al. 1992; Bolger et al. 2000; Riley et al. 2003; Ruell et al. 2012). Recent studies are also beginning to show how interactions among direct and indirect effects (e.g., fire, climate change, non-native species) of urban development contribute to biodiversity loss (e.g., Franklin et al. 2014; Conlisk et al. 2015; Jennings et al. 2016).

Changes in vegetation structure that occur with the conversion of shrublands to grasslands also impact the physical and hydrological properties of the soil (Martinez-Fernandez et al. 1995; Williamson et al. 2004). The increased density of plants combined with changes in the canopy shape and root distribution of individuals significantly alter how rainfall and organic matter are channeled into and through the soil (Lee and Lauenroth 1994; Martinez-Meza and Whitford 1996). The resultant changes affect the infiltration capacity and water retention of the soil as well as the concentration and dispersal of nutrients and carbon (Gutierrez et al. 1995; Martinez-Fernandez et al. 1995). Shrublands that have been converted to grasslands have more extreme soil temperatures and they tend to develop a thicker, more variable surface (A) horizon with a significantly higher soil bulk density (Williamson et al. 2004). These changes in root distribution decrease the stability of slopes while increasing the potential for hazardous debris flows (Gabet and Dune 2002). External factors such as fire and flooding can further exacerbate the system by increasing runoff and soil erosion, which in turn have the potential to affect water quality and reservoir infilling (Hubbert et al. 2012). Finally, shrublands have substantially better capacity for ecosystem carbon sequestration than grasses (Petrie et al. 2015), which has critical implications in this era of rapid climate change.

Development patterns and chaparral conversion are not only important in terms of ecological effects, but from a social perspective, the intermix WUI areas are also the locations where houses are most likely to be destroyed by wildfire in southern California (Syphard et al. 2012). Large fires at the WUI have been occurring for decades in the region, with an average of 500 houses lost per year in the last 50 years. Furthermore, the rate of destroyed houses and lost lives in the last 10–15 years has been unprecedented (Keeley et al. 2013).

12.5 Discussion and Future Changes

As we march into the twenty-first century, the acceleration of global change is bound to occur, especially given the projections of continued population growth. For example, the San Diego Association of Governments expects a 140% increase in population by 2050 across the county (www.sandag.org/2050forecast). Thus, continuation of direct habitat conversion, particularly in the form of urban development, will continue to reduce and fragment chaparral habitat, as well as increase the length and extent of the WUI (Landis and Reilly 2003; Hammer et al. 2009). Furthermore, these land use changes will likely continue to interact with indirect drivers of conversion, including fire and invasion by non-native grasses.

Climate change will also likely result in chaparral species' range shifts, and possibly type-conversion, through habitat shifts and modifying phenology (Chen et al. 2011; Beltrán et al. 2014, see Chap. 14). However, it is the interaction of climate with the drivers discussed here that may be of most concern (Syphard et al. 2013b; Franklin et al. 2014). For example, future projections suggest that land use change will likely either override or compound the impacts of climate change on shrubland habitat conversion across the state of California (Mann et al. 2014; Riordan and Rundel 2014), and in southern California, loss of chaparral species' suitable habitat may be exacerbated by urban growth, with fire being the most serious threat for obligate seeding chaparral species (Syphard et al. 2013b; Bonebrake et al. 2014). Fire regimes, however, are more likely to be altered due to land use change rather than climate change in chaparral shrublands, as fire activity has not been significantly correlated with historical patterns of temperature and precipitation in these areas (Keeley and Syphard 2015, 2016, 2018). This may be due to the fact that climatic conditions are already suitable for extreme fire activity every year on these landscapes. On the other hand, changing patterns and timing of ignitions may have profound impacts on fire activity and its social and ecological consequences (Syphard and Keeley 2015).

Although the South Coast Ecoregion is relatively homogenous in terms of broad scale climatic and vegetation patterns, questions of scale and geographical context will be important when considering future management needs and priorities. For example, species with similar functional traits and sensitivities to certain threats may be differentially exposed to those threats depending on their distributions (Syphard et al. 2013b). That is, areas with the fastest climate change may not always be the same as the areas of fastest land use change or disturbance regime shifts.

Within the South Coast Ecoregion, different counties have unique histories of development and urban growth, which explains why our data show such variation in the extent and spatial pattern of housing density. Accordingly, habitat loss and fragmentation have and will continue to vary across the region. One of the most serious concerns related to chaparral conversion may be the ongoing expansion of low-density development in counties like San Diego, which still contain substantial areas of intact chaparral. Not only does continued development threaten to reduce shrubland extent and continuity, but intermix WUI is the area most prone to non-native annual grass expansion, increased fire frequency, and corresponding fire risk.

One major concern associated with the increase in fires in the southern California region is that vast areas are now covered with very young chaparral due to the enormous extent of recent wildfires. Also, there have already been extensive areas within southern California that have recently burned at anomalously short intervals (Keeley et al. 2009). These trends greatly increase the risk for future conversion to annual non-native grass. An additional potential factor is increased atmospheric pollution. Non-native grasses respond favorably to elevated atmospheric nitrogen deposition, which will likely accelerate with ongoing development (Cox et al. 2014).

Given the profound recent loss of human lives and property in southern California associated with wildfire, there has been a growing sense of urgency to identify new ways to reduce fire risk and ensure community safety. Aside from active fire suppression to control burning wildfires, the most prevalent form of management has been to burn, modify, or clear wildland vegetation to control fire behavior. While fuelbreaks can be safe and effective tools for firefighter access to chaparral communities, research shows that vegetation management in terms of prescribed fire and fuelbreaks provide little benefit for controlling the most damaging weather-driven fires (Syphard et al. 2011; Price et al. 2012; Penman et al. 2014). Given that vegetation management is a driver of chaparral conversion, trade-offs could be carefully considered in the design and placement of fuelbreaks, which ideally could be strategically placed for firefighter defense of communities.

In addition to strategically placed fuelbreaks, homeowner property preparation in terms of building construction and design and defensible space may significantly reduce the risk of a house being destroyed in a wildfire (Cohen 2004; Quarles et al. 2010; Syphard et al. 2014, 2016a). However, while defensible space does provide significant protection, the effect results primarily from modifying vegetation immediately adjacent to the structure. Research has shown there is no added benefit of treating areas farther than 100 ft. (30 m) from the property, even on steep slopes. In addition, only 40% reduction in woody cover was needed for significant protection (Syphard et al. 2014). This is important with regards to habitat, as there has been a recent push from county governments and insurance companies for homeowners to clear up to 300 ft. (60 m) of defensible space around their houses, which cumulatively could result in substantial areas of habitat loss (Keeley et al. 2013).

Considering house losses from wildfire at both local and landscape scales, the most significant factor that explains whether or not a house is destroyed has been its location and arrangement relative to other houses on the landscape (Syphard et al. 2012; Alexandre et al. 2015). Therefore, land use planning may be the most effective long-term solution for not only preventing house loss to wildfires, but also for maximizing biodiversity. Simulation studies showed that land use planning decisions, either through growth policies or through private land acquisition, could result in mutual benefits for both fire risk reduction and biodiversity conservation (Syphard et al. 2013a, 2016b; Butsic et al. 2017). In particular, both house loss and ecological impacts are likely to be most effectively minimized if future development is designed to be compact and clustered, with development restricted in either high-fire-hazard or species-rich areas, which tend to occur in the same areas (Syphard et al. 2016b). Ignition prevention efforts may also be highly effective as part of a comprehensive fire management program (Prestemon et al. 2010; Syphard and Keeley 2015).

12.6 Conclusion

The sprawling development pattern in southern California has been the primary driver of contemporary chaparral conversion, both through the direct removal and fragmentation of habitat, but also through its indirect role in driving annual grass expansion associated with increased fire frequency. It is also indirectly responsible for other factors such as fuelbreaks to protect communities scattered throughout the wildland, climate change, and perhaps even the increase of nitrogen deposition. For example, the increasing road density and traffic volumes associated with increased population and urban development have and will continue to have numerous effects that threaten chaparral ecosystems. Roads are often the source of fire ignitions (Syphard and Keeley 2015), promote the spread of non-native species (Bar-Massada et al. 2014), contribute to elevated ozone and nitrogen deposition that favors grasses over shrubs (Fenn et al. 2010), and fragment habitat needed for sensitive fauna (Poessel et al. 2014).

Thus, as we move into the future, it may be well worth the effort to seriously consider how developments are designed and arranged across the landscape. Land use planning could systematically address the root causes of fire risk as well as habitat loss (Moritz et al. 2014). It could lower ignitions through reduced human presence in flammable areas, lower non-native species expansion by reducing corridors to invasion, and lower the risk of property loss by arranging houses so that they are less fire-prone (Syphard et al. 2012, 2013a). Land use planning can thus address multiple impacts of global change across California shrublands, and may ultimately be the most powerful tool for a sustainable future.

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