

MOUNTAIN LION RESOURCE SELECTION IN THE CALIFORNIA
CENTRAL COAST: MODELING HABITAT SUITABILITY
FOR A LARGE CARNIVORE IN A RAPIDLY
CHANGING ENVIRONMENT

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TITLE: Mountain lion resource selection in the
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ABSTRACT

Mountain lion resource selection in the California Central Coast: Modeling habitat suitability for a large carnivore in a rapidly changing environment

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Land use conversion toward agriculture such as orchards and vineyards can have severe negative impacts on habitat and wildlife, particularly large carnivores, globally through habitat fragmentation and loss. The mountain lion (*Puma concolor*) population in the California Central Coast is thought to provide “stepping-stone” connectivity between several severely genetically compromised coastal populations throughout the Santa Cruz Mountains and several mountain ranges in Southern California; however, the California Central Coast is one of the fastest-developing regions of California with little protection against future land use conversion. Conserving areas of and corridors between high-quality mountain lion habitat through conservation easements should be prioritized. Our results showed that this is especially important in areas currently zoned for agriculture and residential but not fully developed yet. Conserving quality habitat is not only beneficial to mountain lions, but also many species underneath their ecological “umbrella.”

In my first chapter, I performed a literature review detailing what ecologists currently understand about human impacts on wildlife, with an emphasis on large carnivores, through habitat fragmentation and loss, land conversion, and human-carnivore conflict. I also reflected on mountain lion ecology and management in California and North America as a whole, before reviewing analytical methods most commonly used to study their home ranges and resource selection.

In my second chapter, I used GPS collar data from seven GPS-collared mountain lions on the Fort Hunter Liggett Army Base in Monterey County, California to compare minimum convex polygon, kernel density isopleth, and adaptive-local convex hull methods to elucidate the strengths and weaknesses of each when estimating wildlife home ranges and utilization distributions. Following this, I used the GPS data to create a resource selection function to model predicted resource selection patterns of the mountain lions on the Army Base before projecting my model out to the counties comprising the greater California Central Coast. I then overlaid this habitat suitability map with zoning and land protection status maps from each county. My results provide a clear visual representation of not only mountain lion habitat suitability throughout the Central Coast, but areas wildlife and land managers should prioritize for conservation in relation to adjacent areas of varying zoning and protection statuses.

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CHAPTER 1

LITERATURE REVIEW

INTRODUCTION

Large carnivores are integral to many ecosystems and the well-being of the communities within them (del Rio et al. 2001, Ordiz et al. 2013); however, the futures for many large carnivore species throughout the world are becoming increasingly uncertain. They are particularly vulnerable to population decline due to typically low densities, large home ranges, and high metabolic demands which force them to interact with ever-increasing human populations primarily through habitat fragmentation and loss due to land conversion (Ripple et al. 2014, Vickers et al. 2015, Dellinger et al. 2020). An estimated 61% of large carnivores throughout the world are classified as vulnerable, endangered, or critically endangered by the International Union for Conservation of Nature (IUCN) due to habitat fragmentation and loss (Ripple et al. 2014), though the impacts extend far beyond carnivores alone. Fragmentation and loss have negative consequences on wildlife biodiversity in general through loss of species richness (Schmiegelow and Mönkkönen 2002, Fahrig 2003), decline in population densities and distribution (Best et al. 2001, Abade et al. 2014), and loss of genetic diversity (Hilty and Merenlender 2004, Šálek et al. 2009).

Large carnivores have a strong impact on the composition of their ecological communities due to their roles as extensively ranging apex predators (Thorne et al. 2006, Barry et al. 2019). Large home ranges cause them to interact with many other wildlife species with overlapping habitat requirements. Their influence on prey population demographics and behavior through predation and predation risk has been well documented (Altendorf et al. 2001, Laundre et al. 2010, Pierce et al. 2010). For example, significant impacts on mesocarnivore and herbivore population dynamics, forage, and habitat selection have been observed in regions in which large carnivores presently occur, as well as in regions they once did but were locally or regionally extirpated or currently exist at drastically reduced numbers (Rominger 2018, Wang et al. 2015).

Because large carnivores, particularly large felids, are typically generalists in terms of both habitat and prey, and also possess large ranges containing a variety of communities,

ecologists often consider them “umbrella” species for habitat suitability assessments (Thorne et al. 2006, Dickman et al. 2015, Kittle et al. 2017, Barry et al. 2018, Dellinger et al. 2020). Indeed, conservation of these species is often linked to the conservation of the broad spectrum of species falling under their ecological “umbrella” (Redford 2005, Kittle et al. 2017). As members of the family Felidae are distributed across almost all continents (Nowell and Jackson 1996, Dickman et al. 2015) and many large felids are among the most charismatic species across the globe (e.g. African lions (*Panthera leo*), tigers (*Panthera tigris*), leopards (*Panthera pardus*), cheetahs (*Acinonyx jubatus*), and mountain lions (*Puma concolor*)), they are also often considered “flagship” species, and much more likely to garner widespread public support for their conservation (Sergio et al. 2008, Kittle et al. 2017). Their preservation is, therefore, often an important surrogate for biodiversity preservation (Kittle et al. 2017). For example, Thorne et al. (2006) tested the appropriateness of using mountain lions as an umbrella species to represent fine-scale biodiversity elements within the Central Coast of California and discovered that the network of core and corridor habitats that they developed was more than adequate to represent their study area for the majority of the elements analyzed. It is thus imperative to identify areas of core resource usage as well as linkages between those areas in order to conserve important habitat not only for the species itself but the species underneath the “umbrella” (Thorne et al. 2006).

Through this thesis, I will use GPS location data from mountain lions on the Fort Hunter Liggett (FHL) United States Army Base in southern Monterey County to quantify how particular environmental variables influence landscape use within their home ranges. Though FHL is an active military base primarily used for training, it encompasses approximately 680 km² of contiguous wildlife habitat and effectively represents minimally disturbed wildlife habitat within the Central Coast ecoregion. The habitat and biodiversity of this region of California, like many others worldwide, faces impending decline primarily due to land use conversion toward agriculture such as orchards and vineyards as demand for resources from the landscape increases. Through review of the scientific literature, I will explore mountain lion ecology to inform predicted habitat suitability models. Put into context of the history and current status of mountain lion conservation, these models could ideally be used by wildlife and land management

agencies to inform planning for measures such as conservation easements and other habitat protections.

PART I: HUMAN IMPACTS ON WILDLIFE

Habitat Fragmentation and Loss

Habitat fragmentation is among the leading causes of decline in population fitness for most wildlife taxa. It has been defined as “a landscape-scale process involving both habitat loss and the breaking apart of habitat” (Fahrig 2003). Anthropogenic habitat fragmentation in the form of roads, housing, and agricultural development is especially detrimental as it breaks otherwise contiguous habitat into smaller fragments which may prevent individuals from fulfilling their biological needs (i.e. foraging, reproducing, etc.; Ries et al. 2017). Infrastructure such as pipelines, canals, powerlines, etc. for energy transport, primary and secondary roads, and exclusionary and boundary fencing, which come with expanding urbanization, all alter landscapes, resulting in disrupted ecological processes, directly and indirectly increasing mortality rates for wildlife (Jakes et al. 2018, Roehrdanz and Hannah 2014, Mbaiwa and Mbaiwa 2006). Fragmentation also increases edge effects, or detrimental ecological impacts on local populations or communities which occur at the interface of two or more habitats (Porensky and Young 2012).

Major roads

Road networks are increasing in response to global anthropogenic expansion and wildlife-vehicle collisions (WVCs) caused by motor vehicles are among the most frequent, and most visible, contributors to human-caused mortality rates in wildlife populations (Freitas et al. 2013, Collinson et al. 2015, Grilo et al. 2020). Ibisch et al (2016) estimated that 20% of the Earth’s terrestrial surface is within 1 km of a road, with an additional 25 million additional kilometers of road network expected to be established by 2030 (Leonard and Hochuli, 2017, Lawton 2018). An estimated 6.3 million kilometers of road cover the United States alone (Forman et al. 2003, Balkenhol and Waits 2009). WVCs are estimated to kill billions of individuals from a wide array of vertebrate species

globally each year (Visintin et al. 2017), and their ever-increasing prevalence can lead to population declines and local, or even complete, extinction of particular species via direct mortality (i.e. WVCs; Taylor et al. 2002, Shilling et al. 2020) or inhibition of gene flow to surrounding areas (Ng et al. 2003, Wilmers et al. 2013, Riley et al. 2014, Vickers et al. 2015, Zeller et al. 2017, Dellinger et al. 2019, Gustafson et al. 2021). The IUCN lists 193 *Critically Endangered*, 372 *Endangered*, 413 *Vulnerable*, and 351 *Near Threatened* species as threatened by roads (IUCN, 2017, Barrientos et al. 2021).

Though our knowledge of the long-term effects of roads on the persistence of wildlife populations is incomplete, the majority of what ecologists do know pertains to the impacts of WVCs on medium-large mammal populations - directly through collisions but also indirectly through avoidance of roads as a response (Barrientos et al. 2021, Poulin et al. 2023). Due to their often-low densities, long generation times leading to lower recruitment, and large home ranges making them more likely to overlap major roads, medium-large mammals are especially impacted by heavily trafficked roads and less able to withstand high mortality rates (Ascensão et al. 2019, Grilo et al. 2020, Barrientos et al. 2021). Indeed, out of >44,000 WVCs reported to the U.C. Davis California Roadkill Observation System (2021) from 2009-2020 in California, 22,538 involved medium-sized mammals and 10,689 involved large mammals. Vickers et al. (2015) determined that over a 13-year period, 28% of all GPS collared mountain lion mortalities in Southern California (a population known to be severely isolated due to highways) were attributed to WVCs. Similarly, Land and Lotz (1996) reported that vehicle strikes were the cause of mortality for 20% of all Florida panthers. Kauffman et al. (2018) saw constriction of ungulate migration corridors by roads in Wyoming, United States, resulting in migration bottlenecks and population decline. The problem is not unique to the United States; for example, Cullen et al. (2016) predicted that subpopulations of jaguars (listed as *Near Threatened*) in Brazil would have lower persistence within the next 100 years due to exacerbated rates of roadkill directly linked to habitat fragmentation.

Loss of genetic diversity due to habitat fragmentation and physical barriers on the landscape is also an influential factor in predicting large carnivore populations' long-term potential for persistence (Ernest et al. 2003, Bouzat 2010, Vickers et al. 2015, Furlan et al. 2017). Because many of these species' resources (e.g. forage, breeding opportunities,

etc.) are found throughout wide ranges, behavioral and physical avoidance of roads mean that resources often become inaccessible. Further, as the landscape is fragmented into increasingly small patches, remaining habitat is quickly degraded (Dean et al. 2019). Minimally connected, low quality habitat combined with loss of genetic variation is commonly associated with isolated populations, increased spread of disease, intraspecific conflict, increased occurrences of inbreeding depression (Da Silva et al. 2006, Grueber et al. 2008, Hedrick and Frederickson 2010, Furlan et al. 2012, Riley et al. 2020) and genetic drift (Dellinger et al. 2019), and reduced litter sizes (Hedrick and Frederickson 2010). All of these decrease adaptability to environmental stressors (Lande and Barrowclough 1987), making the persistence of a population tenuous in the face of environmental changes (Fisher 1958). Given that large carnivores often already face a plethora of challenges from human encroachment in addition to roads, populations can quickly face local extirpation or even extinction.

On a worldwide scale, road mitigation structures (RMS) have been proven to have varying degrees of success, though research is still lacking due to lack of long-term assessments of species-specific and temporal variation in use (Edwards et al. 2022). The most effective RMS have been identified as wildlife exclusion fencing (Clevenger et al. 2001, Hamr et al. 2022) with the use of wildlife jumpouts (Edwards et al. 2022) and over- and underpasses (Ford et al. 2010, Jensen et al. 2022).

Wildlife exclusion fencing, or fencing intended to create a barrier between vehicular traffic and wildlife, is considered to be the most effective tool for WVC mitigation (Huisjer et al. 2015, Rytwinski et al. 2016). Banff National Park in Alberta, Canada began focusing efforts on ungulate and large carnivore mortality due to vehicles in the early 1980's and discovered that wildlife fencing reduced vehicle collisions with large mammals in the park, including moose (*Alces alces*), bighorn sheep (*Ovis canadensis*), deer (*Odocoileus spp.*), lynx (*Lynx lynx*), mountain lions, coyotes (*Canis latrans*), wolves (*Canis lupus*), black (*Ursus americanus*) and grizzly bears (*Ursus arctos horribilis*), and wolverines (*Gulo gulo*), by 86%, with collisions with ungulates, in particular, reduced by over 90% (Woods 1990, Clevenger et al. 2002). Multiple studies, including a review by Huisjer et al. (2015) and continuing research on RMS in Alberta by Edwards et al. (2022)

corroborated these findings, concluding that wildlife exclusion fencing can decrease WVCs with large mammals by 80-100%.

While wildlife exclusion fencing is an extremely effective tool for WVC mitigation, fencing not designed to funnel animals toward appropriate crossings can intensify the negative impacts of roads and traffic (Glista et al. 2009). Wildlife, particularly deer, can become trapped between roads and the fencing intended to keep them away, actually increasing the likelihood of being struck. Jumpouts, or structures along exclusion fencing which trapped animals to escape, help to alleviate this risk (Jensen et al. 2022). Huijser et al. (2016) observed that fenced sections of road <5 km in length were less effective than those >5 km as animals tended to attempt crossing at fenced ends and proposed that extended sections of fenced roads may alleviate this problem.

Exclusion fencing can also exacerbate barrier effects for wildlife if not constructed to guide animals toward safe crossings. The efficacy of underpasses and overpasses can be difficult to assess due to lack of long-term, multispecies monitoring (Glista et al. 2009); however, several studies have documented a diversity of wildlife utilizing them (Jensen et al. 2022, Edwards et al. 2022, Clevenger et al. 2005, Clevenger and Waltho 2000). In their study monitoring mammal use of overpasses, large underpasses, and small underpasses, Mysłajek et al. (2020) observed large ungulates and carnivores using overpasses most frequently. Many other studies have documented large mammals as more likely to use underpasses with wide, open dimensions, though some large carnivores, such as mountain lions, use smaller undercrossings such as culverts often (Edwards et al. 2022, Kintsch et al. 2019, Clevenger et al. 2005, Ng et al. 2004). It is therefore more open crossing structures which will benefit the majority of wildlife species.

Informed placement of RMS is also crucial, and recent research has placed increasing emphasis on identifying where connectivity would benefit the most from RMS (Clevenger et al. 2009, Gustafson et al. 2022, Hamr et al. 2022). Identifying habitat and habitat linkages where maintenance of gene flow through wildlife crossing features is especially crucial for the persistence of specific genetic populations (Thorne et al. 2006, Vickers et al. 2015, Dellinger et al. 2020, Gustafson et al. 2022, Benson et al. 2023).

Additionally, data has revealed that crossing structures are most successful when implemented where wildlife is already attempting to cross (Clevenger and Huijser 2011, Ascensão et al. 2019), particularly in proximity to native vegetative cover and riparian corridors (Clevenger and Waltho 2000, Riley et al. 2006, Thorne et al. 2006, Craveiro et al. 2019, Jensen et al. 2022) with suitable habitat present on each side (Ng et al. 2004).

Two strong examples of the positive outcome of the aforementioned research are the Wallis Annenberg Wildlife Crossing (WAWC) in Southern California and the Laurel Curve Wildlife Crossing (LCWC) in the Santa Cruz Mountains. Ground was broken for construction of the WAWC, a vegetated overpass over U.S. Highway 101 near Liberty Canyon in Agoura Hills, CA, in 2022. The overpass, complete with fencing to funnel wildlife toward the crossing, was constructed in response to mounting evidence that this part of the highway acts as a significant barrier to new genetic material for mountain lion and other wildlife populations in Southern California (Riley et al. 2014, Vickers et al. 2015, Riley et al. 2021, Benson et al. 2023). Not only are wildlife frequently known to use this area, extensive amounts of land on either side are protected (Riley et al. 2018). Similarly, Highway 17 bisects over 30,000 acres of protected land, presenting a challenge for wildlife attempting to access the habitat (Southwest Concrete Pavement Association 2022) and putting populations on either side of the highway at risk for genetic differentiation (Balderas 2023). The Laurel Curve Wildlife Crossing (LCWC), an underpass with accompanying wildlife fencing connecting several county parks and habitat preserves, became active in 2023.

Habitat Loss: Land Conversion

Significant reliance on resources such as grown food, water, natural gas, and timber have not only caused humans to expand into previously undeveloped areas, but also to convert areas previously utilized sustainably to landscapes heavily fragmented and/or devoid of suitable habitat for many key species. Land-use conversion has been recognized as one of the leading drivers of biodiversity loss (Mohring et al. 2021) and has led to a dramatic decrease in terrestrial species populations around the world (Shackelford et al. 2018). In 2010, it was estimated that 39% of the world's terrestrial habitats had been replaced with

cropland and urban infrastructure, while another 37% had been degraded and fragmented (Ellis et al. 2010). Alcamo et al. (2006) predicted an additional 10-20% of forests and grassland to be replaced by 2050.

In the United States, oak woodlands and grasslands are among the primary habitat types converted to vineyards (Merenlender 2000) and row-crops such as corn, soybeans, and wheat (Martinuzzi et al. 2015, Shackelford et al. 2018). These habitats, which are historically undeveloped in California, are among the most biodiverse ecosystems within California, and contain important resources for wildlife such as vegetated corridors, meadows, wetlands, water sources at lower elevations (i.e. 0-1,000m). Research in Sonoma County, California revealed that an estimated 42% of vineyards established between 1990 and 2000 were also established within lower elevation ranges on slopes greater than 10 degrees (11% slope; Merenlender 2000). Grasslands alone, including those historically used for livestock grazing, made up 88% of the land converted to agricultural production (Lark et al. 2020). As arguably the most threatened terrestrial ecosystem globally with the least amount of protection (Scholtz and Twidwell 2022), the loss of grasslands has a significant negative impact on carbon sequestration, soil integrity, and recruitment of native vegetation (Hendrickson et al. 2019). It has also represented extensive loss of wildlife habitat, and in turn, the level of wildlife biodiversity that regions can support (Hendrickson et al. 2019).

Loss of habitat has been especially detrimental to large mammal species including mountain lions and grizzly bears (*Ursus arctos horribilis*), pollinators including Monarch butterflies (*Danaus plexippus*), many Federally protected migratory waterfowl species, and native plant species like common milkweed (*Asclepias syriaca*) whose pods Monarch butterfly larvae depend on (Shackelford et al. 2018, Lark et al. 2020). Rangelands cover 45% of Earth's land surface, and whereas livestock grazing allows wildlife to continue to disperse and migrate through rangelands due to the preservation of native grasslands and shrublands, conversion to other forms of agriculture such as row crops, orchards, and vineyards can have much less habitat value. This is a threat to the sustainability of wildlife populations due to soil erosion, loss of agriculture and native species diversity, and direct removal of natural resources (Merenlender 2000). It also can result in

decreased tolerance of wildlife by landowners and hindrance of migratory and dispersing populations attributed to exclusionary fencing and complete removal of habitat and movement corridors (Merenlender 2000, Hilty et al. 2019, Katamaya et al. 2019, Kariuki et al. 2020).

Ecologists have only recently begun understanding the effects of intensive, permanent agricultural systems such as vineyards and other row-crops, in addition to the associated anthropogenic development, on wildlife populations. Landscapes with the highest suitability for agriculture are almost completely dominated by high human pressure (Scholtz and Twidwell 2022), and because of this, anthropogenic pressures have increased significantly in areas with only moderate suitability since the early 1990s. As urbanization continues to progress, human populations are expanding into previously undeveloped regions, impacting native wildlife species (Ellis et al. 2010). For example, while attempting to shift their home ranges or migration routes, many wildlife species are not able to adapt to, or even find, new resources to replace previous ones (Hilty et al., 2019, p. 10). Even species known to travel long distances and tolerate less than ideal conditions, such as mountain lions and bobcats (*Lynx rufus*), may actively avoid habitat and corridors with increased human activity and greater edge effects (Smallwood 1994, Machtans et al. 1996, Hilty and Merenlender 2004). When Hilty and Merenlender (2004) investigated mammalian predator use of vineyards versus riparian corridors for movement in Sonoma County, they found that detection rates of native predators were 11-fold higher in undeveloped corridors. Further, many ungulate migrations have been reduced due to land conversion, including building densities exceeding disturbance thresholds and intensive agriculture, resulting in habitat fragmentation, barriers to movement, etc. globally (Kauffman et al. 2021, Gigliotti et al. 2022).

Wildlife Corridors

Wildlife, or habitat, corridors are defined as “any space that facilitates the movement of populations, individuals, gametes or propagules, and plant parts capable of vegetative reproduction in a matter of minutes, hours, or over multiple generations of species” (Hilty et al. 2019). Corridors may be composed of natural or altered vegetation and structures

and can vary broadly over spatial and temporal scales. Land bridges between continents, wildlife over/underpasses along major highways, roadside vegetation, and national parks are all examples of wildlife or habitat corridors as they all enhance movement and allow for im-/emigration of genetic material, dispersal of individuals to new habitats, seasonal migrations, and range shifts in response to disturbances (e.g. anthropogenic development, climate change, etc.).

Wildlife corridors often function as linkages between biologically significant habitats, helping to mitigate negative impacts of fragmentation (Conrad et al. 2012). By providing topography, hydrology, vegetation, and structure uniquely fitted for wildlife movement such as migration and dispersal (Rudnick et al. 2012), corridors allow for increased mobility and gene flow (Andreassen et al. 1996). This can have the effect of increasing habitat area and diversity beyond the individual areas the corridor links, providing rescue for populations facing local extinction (Caro et al. 2009), and providing refuges for species with limited adaptive capabilities (Krosby et al. 2018). Riparian habitats, for example, are particularly important for wildlife movement as they offer plentiful vegetative cover, cooler microclimates, fresh water, and vegetational complexity, which in turn support a disproportionately high level of biodiversity (Hilty and Merenlender 2004).

Despite their importance to wildlife movement and resilience in the face of climate change, wildlife corridors are becoming ever more threatened due to land conversion. Permanent agriculture such as orchards and vineyards are expanding dramatically into previously undeveloped foothill grasslands and oak woodland, which often feature important corridors for natural communities (Hilty and Merenlender 2004, Roehrdanz and Hannah 2016). Many native species, including many large carnivores with large area requirements, tend to avoid heavily altered landscapes, compared to less-altered landscapes such as pastures and rangelands. This can limit even highly mobile species' abilities to obtain resources and retain genetic diversity (Machtans et al. 1996, Hilty and Merenlender 2004, Šálek et al. 2009, Wilmers et al. 2013, Vickers et al. 2015, Riley et al. 2021).

Maintaining functional connectivity is especially crucial for large carnivores which have historically been subject to especially large-scale habitat destruction and range contraction (Ripple et al. 2014). Even as habitat fragmentation and loss diminish connectivity and habitat quality, leading to the risk of local extirpation (Rudnick et al. 2012, Riggio and Caro 2017, Hilty et al. 2019), it is the same traits which make large carnivores so resilient to environmental stressors that make conservation still possible. Many carnivores, large felids in particular, are capable of a high degree of plasticity in terms of both habitat and prey preferences and are capable of dispersing long distances (Weaver et al. 1995). While remaining suitable habitat shrinks and becomes saturated, making it infeasible for individuals to bring new genes within each patch, single small areas of habitat that maintain connectivity to larger areas and/or many other small areas may still provide enough resources to maintain viable metapopulations (Primack 2004, Mills 2013).

Human-Carnivore Conflict

As human populations increase, the likelihood of human-large carnivore interactions, most often through livestock depredation, also increases (Dellinger et al. 2021, Galvez et al. 2021). While large carnivores typically avoid areas altered by humans (Riley et al. 2021, Dellinger et al. 2019, Støen et al. 2015, Rogala et al. 2011), human encroachment on their naturally large ranges often makes this unavoidable. Consequently, interactions with domestic pets and livestock in rural and residential areas as well as larger-scale livestock operations for agriculture, consistently pose an increased threat of mortality (Baruch-Mordo et al. 2008, Pettigrew et al. 2012, de Souza et al. 2018, Dellinger et al. 2021).

Studies have suggested that certain factors may contribute to the increase in HCC: proximity of humans and livestock (primarily small hoofstock operations, i.e. sheep and goats) to suitable available habitat, negative attitudes toward large carnivores, and lack of deterrents in place by livestock owners to offset depredation (Dellinger et al. 2021, Kuiper et al. 2021, Kirilyuk and Ke 2020 Kissui et al. 2018, Peebles et al. 2013). The proximity at which humans and wildlife are coexisting is decreasing by a substantial

amount each year due to human encroachment into, and up to, suitable natural habitat, resulting in habitat fragmentation (Crawshaw 2003, Michalski et al. 2006, de Souza et al. 2018).

Habitat fragmentation causes humans and wildlife to overlap more often in part because it creates areas of higher wildlife activity due to edge effects (Kirilyuk and Ke 2020, Zeller et al. 2017, Cerboncini et al. 2016, Ogada et al. 2003). High levels of human-caused mortality are not uncommon when high densities of humans, many of whose livelihoods depend on livestock, surround protected areas (Harcourt et al. 2001, Woodroffe and Ginsberg 1998). While investigating depredation rates by wolves on livestock along the borders of Russia, Mongolia, and China, Kirilyuk and Ke (2020) found that predation rates on livestock were particularly high in pastoralist camps nearby to protected suitable habitat where wolf ranges and livestock grazing overlap. Retaliatory killings by humans on carnivores can cause population declines and threaten populations already at risk with extirpation despite the conservation intentions of the reserves (Ogada et al. 2003, de Souza et al. 2018).

While depredation on livestock may be motivated by the severity of damage and threat to the livelihoods of property owners (Ekernas et al. 2017), it typically precedes negative social and cultural attitudes towards large carnivores (Kirilyuk and Ke 2020, Kissui et al. 2019, Hazzah et al. 2017). Their wide-ranging nature often results in conflict with humans globally, and as human populations continue to expand into natural habitat, this conflict is increasing (Ale 1998). For example, African lions have experienced an approximate 43% decline from 1995 to 2016 (Bauer et al. 2015) in large part due to high human-caused mortality. In East Africa, where nearly 60% of the population is found (Riggio et al. 2013), this is of special concern (IUCN 2006). Perceptions of lions are often based on negative interactions such as livestock depredation (Dickman et al. 2014, Hazzah et al. 2009), but are also influenced by cultural and social traditions (Abade et al. 2014), economic status of rural communities (Chinchilla et al. 2022, Kirilyuk and Ke 2020), and media influence (Mitchell et al. 2022). In Central America, jaguar populations (classified by IUCN as “near threatened”; Quigley et al. 2018) continue to be threatened by retaliatory and preventative killings deep-rooted in the culture of indigenous

communities (Chinchilla et al. 2022), while strong emotional attitudes of rural pastoralists cause high rates of predation of wolves in Central Asia to persist (Kirilyuk and Ke 2022). The proliferation of the internet and social media globally have also been significant drivers of public perception in the modern era. Instinctive fears of predators are amplified through sensational coverage, resulting in lower tolerance towards carnivores, decline in the support of their conservation, and exacerbation of human threats to particular species (Nanni et al. 2020).

Despite global conflict between large carnivores and human communities, coexistence is possible through improved depredation deterrents. Traditionally, the mission of many government wildlife agencies has been to remove animals causing property damage or threatening human safety rather than to protect wildlife (Graham 1973), and many formal and informal operations have resulted in the endangerment and/or extirpation of species (Beschta and Ripple 2020, Bauer et al. 2015). In situations where livestock operations experience significant financial loss due to carnivores or public safety is threatened, lethal removal of individual animals may be justified, and can even reduce retaliatory killings (Woodroffe et al. 2005); however, Peebles et al. (2013) found that lethal removal of mountain lions in Washington, USA does not actually reduce depredation incidents in the long term, while Krofel et al. (2011) concluded similarly with respect to wolves in Slovenia from 1995-2009. In recent years, wildlife agencies have begun putting more emphasis on proactive mitigation measures globally (Dellinger et al. 2021, Treves and Naughton-Treves 2005). This often includes, but is not limited to, fortified overnight enclosures (Kissui et al. 2019, Ogada 2003), on-site supervision of livestock (Kirilyuk and Ke 2020), and livestock guardian dogs (Iliopoulos et al. 2009). In studies testing the efficacy of each of the aforementioned mitigation measures, depredation rates were highest when livestock were left unprotected, and declined in all cases when at least one form of mitigation was employed. While lethal removal of carnivores is still the predominant method for managing HCC (Lorand et al. 2022), in part because it is perceived as “solving” the problem rather than overlooking it whether or not it reduces further conflict (Linnell 2011, Dickman et al. 2013) in addition to being cheaper financially as the government is often responsible for funding the removal but not improved husbandry (Lindsey et al. 2013), ultimately, data suggests that improved

livestock protection and husbandry techniques provide more effective long-term solutions to human-carnivore conflict when compared to simply lethally removing the depredating species.

PART II: MOUNTAIN LION ECOLOGY

Mountain lions (Felidae: *Puma concolor*) are the only extant member of the genus *Puma*. They are apex predators in most of the the ecological communities they inhabit and prey primarily on ungulates (e.g. deer, elk, wild boar, livestock; Wilckens et al. 2016, Knopff et al. 2010), but are opportunistic generalists and will also hunt smaller prey such as rodents, rabbits, and mesocarnivores (Cunningham et al. 1999). Mountain lions and their prey rely on cover for concealment for and from ambush and select for highly vegetated and/or rocky habitat types which aid in this (Logan and Irwin 1985).

Mountain lions are the most widely distributed terrestrial mammals throughout the Western Hemisphere (Peterson et al. 2021, Hornocker and Negri 2010). They range continuously from the Canadian Yukon to near the southern tip of South America (Pierce and Bleich 2003, IUCN 2015), though historically they also inhabited the Eastern United States before lethal removal, loss of prey, and urbanization extirpated them from most of this range (Burdett et al. 2010). They successfully inhabit a broad range of habitat types, from desert and semi-desert (Logan et. al 1996), temperate rain forests, coastal chaparral and forest (Dellinger et al. 2019), to swamp hammock forests in Florida (Belden et al. 1988). Within these ecoregions, mountain lions select for habitat types with vegetation which allows them to remain concealed stalking and ambushing their prey (Logan and Irwin 1985). Currently, the main threats to their numbers include vehicle strike (Vickers et al. 2015, Road Ecology Center 2021), depredation permits, poaching, wildfire, exposure to toxicants and disease, and prey depletion (Thorne et al. 2006, Riley et al. 2007, Riley et al 2014, Ha and Shilling 2017).

Mountain lions are mainly solitary by nature, though males and females will temporarily travel together prior to breeding, as mother and dependent offspring, or as siblings just after dispersal (Elbroch et al. 2016). Males are territorial based on available resources and maintain home ranges ranging from 100-1,000+ km² (Laundré and Loxterman 2007,

Karelus et al. 2021), while females keep smaller home ranges ranging from 30-300 km² based on season and presence of dependent kittens (Laundré and Loxterman 2007, Nickel et al. 2021, Riley et al. 2021; Figure 2.). The gestation period for females is about 90 days, with litter sizes typically between 2-4 kittens. Kittens remain with their mothers until dispersing between 1.5-2 years old (Jansen and Jenks 2012).

Mountain lion management throughout North America presents a considerable challenge. This is due to highly polarized agendas from political and special interest groups (Mattson et al. 2010) as well as inherent difficulties when attempting to quantify population demographics and trends of a highly elusive apex predator (Beausoleil et al. 2013, Cougar Management Guidelines Working Group 2005). Primarily managed as a game species in western states, except for California where they are classified as a specially protected mammal (Dellinger et al. 2019), it is only within recent decades that improvements to Global Positioning System (GPS) technology and funding for more comprehensive field research have provided more reliable data with which to better understand and manage mountain lions (Beausoleil et al. 2013, Kertson et al. 2011, Hornocker and Negri 2010, Lambert et al. 2006, Logan and Sweanor 2001).

Mountain Lion Management in California

Mountain lion population management in the Western United States, especially California, has evolved significantly throughout the 1900's (Dellinger and Torres 2020). In 1905, the Federal government began hiring trappers to kill wolves on grazing lands within United States national forests (Dunlap 1984), and while bounty programs had been a control tactic for terrestrial predators in North America for centuries, by 1915 Congress had established a continuing program for predator eradication which included bounty programs for coyotes, red foxes (*Vulpes vulpes*), brown bears, and lynx, among others (Proulx and Rodtka 2015). From 1907-1963, mountain lions were also hunted under a bounty system throughout the United States (Fitzhugh and Gorenzel 1986, Mansfield and Weaver 1989). In California, at least one full-time statewide mountain lion hunter was employed from 1919-1959 (McLean 1948, Nowak 1974). By the time this bounty period concluded, more than 12,500 mountain lions had been killed, an average of 224 per year

(Mansfield and Weaver 1989). In the years that followed, there was no longer a financial incentive to hunt mountain lions, but hunters did not need a hunting license to take unlimited lions year-round for sport and livestock protection. From 1970-1971 a hunting season was initiated with license and tag requirements; however, mountain lions' status as a game species was suspended by state legislature in 1972 in response to mounting public concern over population status in California. This suspension was extended several times until failing in 1985 (Fitzhugh and Gorenzel 1986). Ungulate conservation groups successfully lobbied to reinstate them as a game species soon after (Dellinger and Torres 2020), and in response the California Department of Fish and Wildlife (CDFW) began devising harvest quotas and management zones (Mansfield and Weaver 1989). These regulations never went into effect, however.

In 1990, California voters approved Proposition 117 (Fish & Game Code §4800-4809), which designated mountain lions as a “specially protected species.” As a result, it became illegal to “hunt, pursue, catch, capture, kill, or attempt any of these things” in California except under specific circumstances. Currently, mountain lions may only be lethally removed if CDFW issues a depredation permit for a lion confirmed to have killed livestock or pets (hereafter “domestic animals”), or to preserve the safety of the public or a federally listed wildlife species such as California bighorn sheep (*Ovis canadensis spp.*, Fish & Game Code §4801). Mountain lions are the only species in California to hold this status, and though the intent is to reduce mortalities, it has had a strong impact on CDFW's ability to track population dynamics as population estimation programs (e.g. mule deer and elk) are often funded through the sale of hunting licenses while data from hunter harvest informs population numbers (Lueck and Parker 2022).

Between 2001 and 2020, 1,834 mountain lions were lethally removed under depredation permits in California (California Department of Fish & Wildlife, 2020). Counties such as El Dorado and San Luis Obispo reported steadily high numbers removed within the timeframe compared to other counties (El Dorado County removed an average of 8 per year, and San Luis Obispo County removed an annual average of 4), and San Diego's removals increased steadily (an average of one lion removed per year between 2001 and 2010 before increasing to an average of 3.5 lions from 2011 to 2020). All three counties

encompass substantial wildlife habitat; however, despite differing habitat types, they have all experienced significant human population growth (El Dorado Co. experienced 5.5% growth, San Luis Obispo Co. experienced 3% growth, and San Diego Co experienced 7% growth) and anthropogenic expansion into previously undeveloped wildlife habitat between 2011 and 2019 (California Department of Finance, 2020). Indeed, Dellinger et al. (2021) found that presence/quality of mountain lion habitat was correlated with predicting depredation rates in California. Further, Dellinger and Torres (2020) found that not only has the human population in California increased from 20.7 to 39.9 million people (California State Association of Counties 2019) between the 1970's and 2019, since the moratorium placed on hunting in 1972 the mountain lion population has grown as well. As humans continue to expand into wildlife habitat, interactions between humans, domestic animals, and wildlife are expected to also continue to increase. The lethal removal of mountain lions is also expected to increase.

A series of changes to the existing California mountain lion depredation policy began in 2017 in response to mounting evidence that populations in Southern California were increasingly in peril due to extreme loss of genetic variability and inbreeding from not only increasing depredation, but high levels of habitat fragmentation (Ernest et al. 2014, Riley et al. 2014, Benson et al. 2016). This situation can largely be attributed to major interstate highways (e.g. I-10, I-15, I-210, I-405, etc.) and associated development creating nearly impermeable barriers to gene flow and islands of minimally connected, low quality habitat (Vickers et al. 2015, Riley et al. 2014). The results are isolated populations, low genetic diversity, inbreeding depression (Dellinger et al. 2020, Riley et al. 2021, Gustafson et al. 2022), and increased intraspecific conflict (Riley et al. 2020) within these ranges. Benson et al. (2016) estimated a 20% probability of local extinction for the Santa Monica Mountains population within the next 50 years, with expedited extirpation if inbreeding depression increases mortality.

In December 2017, the California Fish and Game Commission made an amendment to the existing depredation policy resulting in more stringent requirements for depredation permits primarily in the vicinity of the Santa Monica and Santa Ana Mountain Ranges in an effort to preserve the genetic diversity of the population. Under the initial policy put in

place via Proposition 117, CDFW was required to issue a depredation permit upon request from the property owner authorizing lethal removal of a lion following any single incident where a domestic animal was killed. Following the amendment, however, domestic animal owners within the Santa Ana and Santa Monica Mountains must experience three attacks on their domestic animals despite non-lethal efforts carried out to deter mountain lion predation before a lethal depredation permit may be issued (California Department of Fish & Wildlife 2017).

In February 2020, the geographic boundaries of this updated policy were expanded in response to a statewide study by Gustafson et al. (2019), in which they genotyped over 990 mountain lions throughout California and Nevada to investigate genetic diversity among individuals throughout both states. Their results identified nine genetically distinct populations in California (Figure 1.1) and though they found that genetic diversity was highly variable between them, the populations within the Santa Ana, Santa Monica, and Santa Cruz Mountain Ranges tended to be genetic sinks with highly fragmented gene flow and low genetic diversity. As a result, all Southern California was included in the boundaries for the updated depredation policy, encompassing the Eastern Peninsular, San Gabriel, and San Bernardino Mountain Ranges. The Santa Cruz Mountains were also included in the new boundaries. The greater Central Coast, (defined as Monterey, San Luis Obispo, Santa Barbara, San Benito, and northern Ventura Counties; Figure 1.1), beyond just the Santa Cruz Mountains, was also included in the policy change.

Dellinger et al. (2020) built upon Gustafson et al.'s (2019) findings, discovering that the amount of variation in not only allelic richness, or the number of alleles at a given site on a chromosome, but heterozygosity and genetic effective population size (N_e) could be explained in large part by the amount of overall suitable habitat available as well as the amount of protected suitable habitat. Specifically, Dellinger et al. (2020) revealed that that while the mountain lions in their study required a minimum of 14,591 km² of suitable habitat in a given area to maintain allelic richness ≥ 3.00 and $N_e \geq 50$ and mitigate the negative effects of low genetic diversity, only 7,923 km² was necessary if the habitat was protected from anthropogenic development such as urbanization and/or agricultural expansion as opposed to of mixed-status protection. Allelic richness values in the Santa

Cruz, Santa Monica, and Santa Ana Mountain populations were found to all be < 3.00 , and the areas of overall and protected suitable habitat they inhabit to be well below the overall and protected thresholds (Table 1.1).

Ample protected habitat is beneficial to wildlife in large part because protected habitat tends to be of higher quality than unprotected habitat, less vulnerable to fragmentation, and maintains greater permeability for gene flow from other populations. Protecting habitat is exponentially more effective at conserving biodiversity if the habitat is part of a greater ecological network (Hilty et al. 2019, p.6), and larger patches of habitat have the potential to act as genetic sources to individual smaller patches if there is adequate connectivity. Additionally, multiple small, interconnected patches may collectively provide enough suitable habitat to maintain viable populations, ultimately allowing metapopulations to persist (Mills 2013, Zanon-Martinez et al. 2016, Dellinger et al. 2020).

In response to Dellinger et al.'s (2020) results, the Center for Biological Diversity and the Mountain Lion Foundation submitted a petition to the California Fish and Game Commission to list the Central Coast and Southern California mountain lion populations as threatened or endangered pursuant to the California Endangered Species Act (CESA; Center for Biological Diversity and the Mountain Lion Foundation 2019). Situated between the Santa Cruz and Southern California populations and relatively stable in terms of numbers, genetic diversity, and adequate habitat, the Central Coast population was included to protect “stepping-stone” connectivity between Santa Cruz and Southern California (California Department of Fish and Wildlife 2020, Dellinger et al. 2020, Gustafson et al. 2022).

As a candidate species for threatened or endangered status under CESA, mountain lions in the Central Coast are currently afforded all of the same protections as if they were officially listed (Fish and Game Code §2068); however, this listing does not address the expanding urbanization and habitat fragmentation which has led to the limited genetic variability of populations to the north and south. The Central Coast currently encompasses 16,355 km² of overall suitable habitat and the mountain lion population shows “intermediate” levels of genetic diversity, with an ideal overall allelic richness

value of 3.00 and an adequate observed heterozygosity value (Dellinger et al. 2020). Only 6,780 km² (41.5%) of this suitable habitat is protected from future development, however. The values for allelic richness and observed heterozygosity in the Central Coast Central Coast mountain lion population indicate that there is indeed adequate genetic variation in this region of California (Table 1.1); however, the mixed protection status of the habitat across this region puts the longevity of the mountain lion population at risk.

The Central Coast of California currently has a large amount of continuous wildlife habitat in comparison with many other parts of the state; however, it is one of the fastest-developing regions of California (Thorne et al. 2006, California Department of Finance 2020). The counties which comprise the Central Coast currently have a total human population of 2.1 million, which is projected to increase to about 2.2 million by 2040 (California Department of Finance 2020). Historically, development pressures have largely been localized along the coast, with development primarily in the form of large ranches, agriculture, and small agricultural towns (Ca. Dept. of Fish and Wildlife 2020).

The majority of the Central Coast is under private ownership (Figure 1.2) with the surrounding landscapes largely zoned for agriculture, residential, and commercial purposes. With its Mediterranean climate offering year-round moderate temperatures and plenty of unregulated groundwater which provides some buffer against the effects of drought and climate change, the Central Coast supports a substantial amount of California's agriculture, rangeland, and wildlife habitat. Many other regions of California which have also historically been able to support agriculture have experienced mandated groundwater use restrictions in recent years (Lin and Egerer 2020; Pathak et al. 2018), and the response in many cases has been to expand operations to other areas in the state - potentially with fewer regulations.

While the Central Coast supports a vast range of crops, it is perennial crops such as orchards (e.g. avocados, tree fruits, and nuts) and vineyards in particular which are expected to experience the largest expansion and lead to the greatest conversion of natural and rangeland to cropland and urban growth. Both crop types are expected to put a significant strain on already taxed natural resources through land conversion and year-round watering, in addition to urban expansion (Wilson et al. 2020). Wilson et al. (2020)

predict that the most dramatic increases in urban growth and perennial cropland are expected to take place in Monterey and San Luis Obispo Counties, with a projected increase in developed land by 21.6% and 28-28.5%, respectively, by 2100.

This conversion of natural and rangelands to cropland and urban/suburban lands is a direct threat to wildlife habitat conservation (Gustafson et al. 2021). Without protections in place from potential future expansion or conversion to higher-intensity cropland, the local mountain lion population is left vulnerable to habitat loss and adjacent populations are at risk of losing valuable gene flow in the future (Dellinger et al. 2020). As such, it has become increasingly important to identify high quality habitat, secure protection (i.e. conservation easements, etc.) for the most crucial movement corridors, and maintain the status of that which is currently protected to help ensure the perseverance of not only local and adjacent mountain lion populations but other wildlife species with overlapping habitat requirements.

PART III: CHARACTERIZING WILDLIFE HOME RANGES

Comparing Home Range Analysis Methods: Minimum Convex Polygons and Adaptive-Local Convex Hulls

A home range (HR) can be defined as the area that the individual animal or group routinely traverses during normal activities required for survival and maintenance of fitness (e.g. foraging, reproducing, travel, etc.), defends against conspecifics (Potts and Lewis 2014), and uses at varying intensities based on these activities (Powell and Mitchell 2012). This is typically done by calculating densities of use from location data collected from individual animals or groups of animals.

Two of the most commonly used methods are minimum convex polygons (MCPs) and adaptive-local convex hulls (a-LoCoH). Minimum convex polygons generate the smallest convex polygon (or hull) containing a specified portion of all used locations (Rose 1982, Getz and Wilmers 2004); however, it has been recognized that they are often overly-simplistic, sensitive to outlying points leading to inclusion of areas that the animal seldom or never uses, and incapable of accounting for concave edges (Getz and Wilmers

2004, Burgman and Fox 2003), many modern studies still use them to compare with prior studies and to estimate HRs for species which may be difficult to collect high amounts of location data (Getz et al. 2007). Conversely, while HRs estimated using a-LoCoH also do so by generating the smallest convex polygon containing some specified percentage of all locations used, they adapt with increasing sample sizes, are less sensitive to forays outside of the true HR and are capable of recognizing concave boundaries and impassable features on the landscape which focal animals may be bounded by (Getz et al. 2007, Scull et al. 2012, Karandikar et al. 2023). Through a-LoCoH, all points within an adaptive radius, a , around the root point are used and local hulls (minimum convex polygons) are then constructed such that the sum of the distances between neighboring points and the root is $\leq a$ (Getz et al. 2007, Walter et al. 2015). The area within the union of these convex hulls is known as the LoCoH covering. By ordering these hulls from smallest to largest until some target percentage (e.g. 75%, 95%, etc.) of the total number of points are included, home range boundaries which encompass the target percentage isopleth of the densest group of points in the LoCoH covering begin to form (Getz and Wilmers 2004, Walter et al. 2015). As the density of data points increases, this adaptive method allows for the number of points involved in the construction of the hulls to increase (Getz et al. 2007). As only the radius between nearest-neighbor points is taken into account when calculating the home range, impenetrable landscape features are reflected in patterns of used locations, and thus the HR boundaries. The accuracy of HRs derived via a-LoCoH is largely dependent on large amounts of data points (e.g. > 30 points; Sillero et al. 2021). This makes it highly useful for data collected via GPS collar data, but not as useful for research with sporadic species detections, such as those collected with remote cameras, etc. Using 95% of the locations, as opposed to 100%, excludes peripheral data points which may be excursions outside of the actual home range boundaries and thus not representative of the animal's "normal activities" (Powell 2000).

Comparing Core Area/Utilization Distribution Analysis Methods: Kernel Density Estimators and Adaptive-Local Convex Hulls

Once the boundaries and size of an animal(s)' HR are understood, data on habitat, resource use, and landscape features elucidate core areas, or areas within an HR where use exceeds what would be expected under a uniform distribution (Samuel et al. 1985). While home ranges aid in identifying an animal's, or group of animals', "cognitive map" of its environment (Powell and Mitchell 2012), ecologists often employ KDEs such as kernel density isopleths (KDIs) and nonparametric local convex hulls (LoCoH) to identify utilization distributions (UDs), or core areas of use (Sillero et al. 2021, Getz and Wilmers 2004, Burgman and Fox 2003). UD represents the probability of an individual animal being found at a particular location within its home range at any given time (Powell and Mitchell 2012), and aid in elucidating which resources they select for, and how individuals/groups interact with conspecifics (Fieberg and Kochanny 2005, Powell and Mitchell 2012).

KDEs are often calculated using bivariate Gaussian parametric (i.e. bounded) kernel methods, using grid-based rasters, where an estimate of the mean probability of use is based on the density of points within each grid cell (Hemson et al. 2005). A smoothing parameter (h) used in traditional, or parametric, KDEs adds area around each data point and determines how accurate the isopleth encompassing the desired percentage of data is (e.g. too large and the isopleth will be overestimated, potentially ignoring any impenetrable boundaries the study species is unable to cross; too small and the resulting isopleth may be underestimated and appear as multiple, discontinuous islands). The method by which h is calculated determines the size and shape of the utilization distribution isopleths; however, there are multiple preferred methods for estimating h : the reference smoothing factor ("href"), least-squares cross-validation, and estimating the value based on the researchers' own assessment of the "best value" (i.e. the *ad hoc* method; Hemson et al. 2005). To date, spatial ecologists have not reached consensus about which method is best, and as a result, size estimates often cannot be directly compared between studies.

Using the *ad hoc* method for a bivariate normal kernel (Calenge 2006) involves estimating h given:

$$h = \sigma n^{-1/6}$$

and

$$\sigma^2 = 0.5(\text{var}(x) + \text{var}(y))$$

As described above, adaptive-LoCoH (a-LoCoH) uses all points within an adaptive radius, a , around a root point to create local hulls such that the sum of the distances between neighboring points and the root is $\leq a$ (Getz et al. 2007, Walter et al. 2015). As their accuracy improves with increasing data, core usage areas estimated via a-LoCoH are considered to be more reflective of an animal's true resource selection patterns at varying intensities of use. Additionally, as they do not add area around each root point via a smoothing parameter, they are especially useful for research investigating the impacts that anthropogenic fragmentation and natural impenetrable boundaries have on wildlife resource use (Getz et al. 2007).

PART IV: MODELING WILDLIFE RESOURCE SELECTION

Resource Selection Functions

In order to work toward conserving suitable but currently unprotected mountain lion habitat, the key resources which influence habitat selection within their home ranges (i.e. Johnson's (1980) third order of habitat selection, which relates to how particular resources within home ranges are selected for) must first be identified. Resource selection functions (RSFs) are often used by ecologists to estimate probabilities of use based on variables relevant to the life histories of the focal species.

RSFs, like other statistical models used to identify resource selection such as Euclidean distance analysis and spatially explicit population models, quantify the relative importance of different resources by comparing usage to availability within a habitat (Manly et al, 2002, Koper & Manseau 2010). RSFs are especially useful, however, as they use empirical data specific to both the species and its landscape. This makes for a

more rigorous design which elucidates differences in selection at multiple scales, allowing biologists seeking to conserve and manage wildlife to ask a broader range of questions (Manly et al. 2002, Chetkiewicz and Boyce 2009, Zeller et al. 2017). RSFs are also easy to comprehend visually as they synthesize habitat use across multiple scales, across a single surface (Zeller et al. 2017). Finally, as the hierarchical conditional probabilities collapse into a single logistic equation, RSFs are reasonably straightforward to derive (Manly et al. 2002, Zeller et al. 2017).

The resources being considered, in this case landscape features within individual animal's home range, are comprised of populations of used and available units where the features of each unit such as topography, vegetation class, distance from perennial water, etc. ($X = X_1, X_2, \dots, X_p$) are measured and analyzed (Manly et al. 2002).

Once the relevant data has been extracted from each point, one method of estimating probability of use is logistic regression with a logit link, given by the equation:

$$\text{logit}(x) = \beta_0 + \beta_1 x_1 + \beta_2 x_2 \dots \beta_p x_p$$

where $\text{logit}(x)$ is the log link given by the natural logarithm of the odds of x_1 :

$$\ln(e^x / (1 - e^x))$$

$\text{logit}(x)$ represents the probability of use (for a given location within a mountain lion's home range), β_0 represents the intercept, β_1 represents the coefficient for the habitat variable x_1 . Models are then ranked based upon Akaike information criteria (AIC), an information-theoretic (I-T) approach which often replaces t tests and ANOVA tables due to their limited inferential capabilities. I-T approaches are simple to calculate and comprehend while still providing formal measures to assess support for multiple alternative hypotheses (Burnham et al. 2011). AIC methods measure the amount of information lost in the transition from data to the model and the "best fitting" model is that with the lowest AIC value, as this indicates that the least amount of information was lost. AIC also penalizes models based on excessive parameters, as a more complicated model almost always has a better "fit." Ultimately, the "best supported" model is the most parsimonious model which best explains the effects of the covariates on the probability of use while losing the least amount of information. This model has the

lowest AIC value as compared to other models by a difference of at least 2.0 (Burnham and Anderson 2002, Manly et al. 2002).

Tables & Figures

Table 1.1. Genetic diversity values (expected heterozygosity and allelic richness) and overall/protected habitat values for Santa Cruz, Santa Monica, Santa Ana, and Central Coast mountain lion populations in California (Gustafson et al. 2019, Dellinger et al. 2020).

^aPercent of overall habitat in a given area which is protected.

Population	Observed heterozygosity	Allelic richness	Overall habitat (km²)	Protected habitat (km²)
Western Sierra Nevada	0.51	3.63	40,397	22,183 (55%) ^a
Eastern Sierra Nevada	0.52	3.46	10,241	9,889 (97%)
North Coast	0.40	3.06	27,091	11,624 (43%)
Santa Cruz	0.41	2.62	5,042	1,818 (36%)
Santa Monica Area	0.41	2.63	2,688	1,129 (42%)
Santa Ana	0.34	2.27	2,054	1,081 (53%)
Central Coast	0.45	3.00	16,355	6,780 (41%)
Eastern Peninsular	0.44	3.07	7,683	4,777 (62%)
Transverse Range	0.40	2.75	3,759	2,976 (79%)

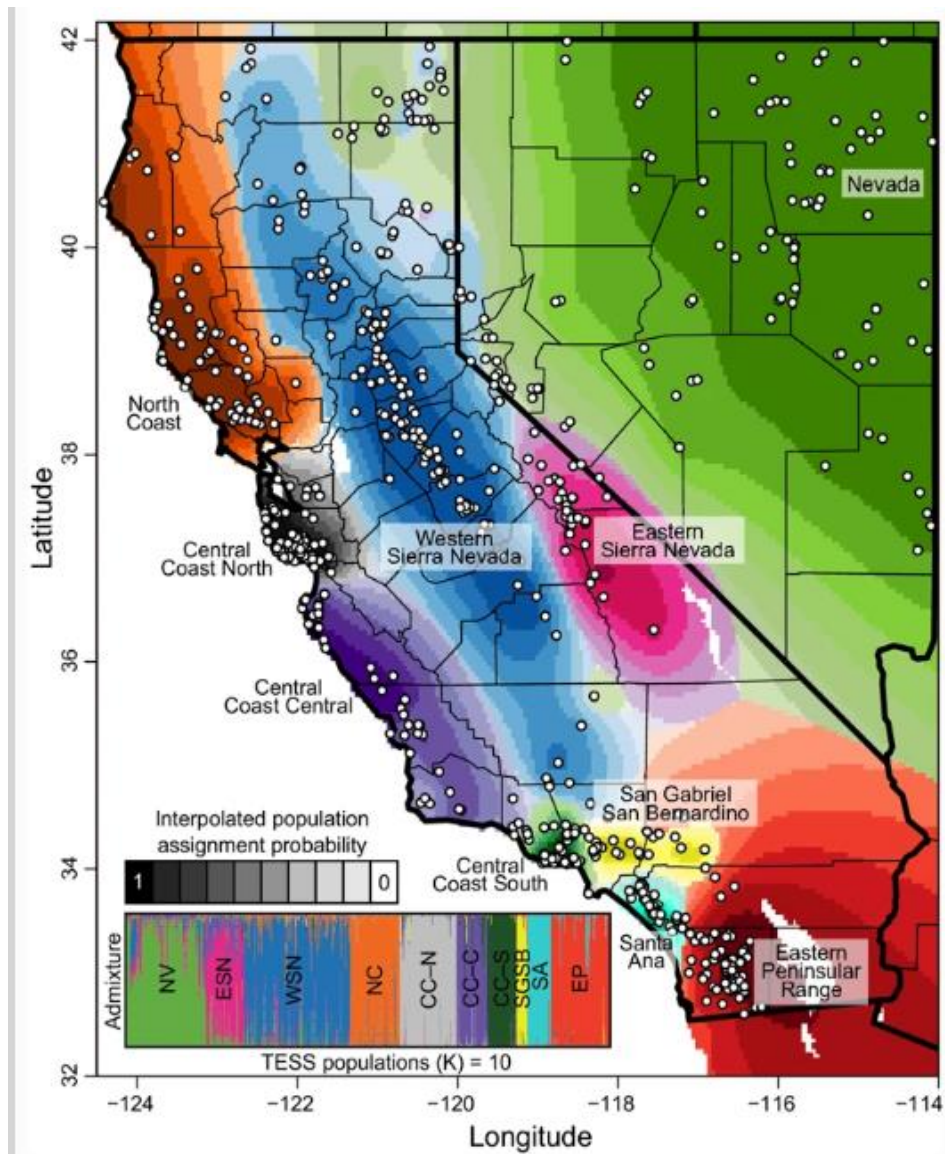


Figure 1.1 Map of the population genetic structure of mountain lions across California and Nevada (Gustafson et al. 2019).

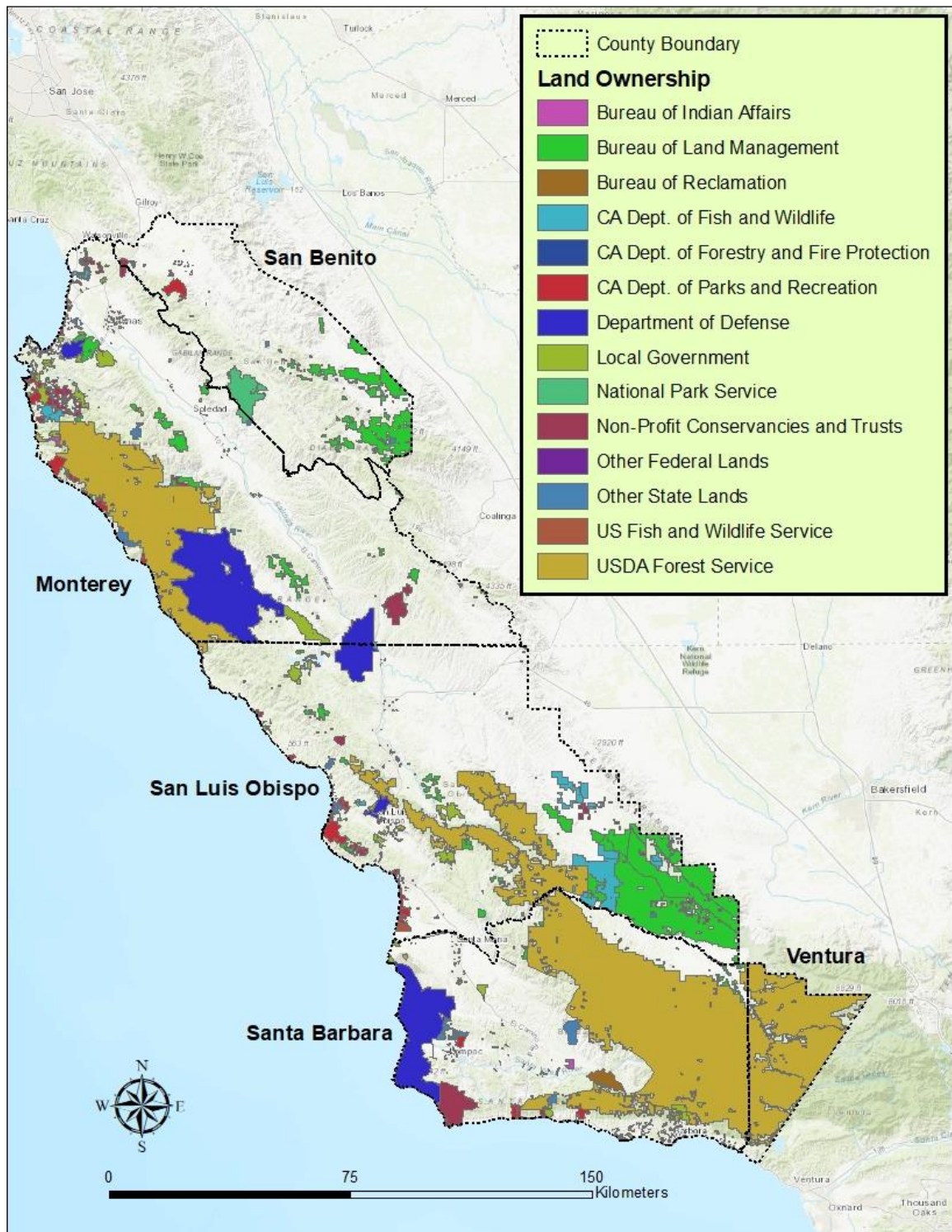


Figure 1.2 Map of land ownership in the California Central Coast Region (California Department of Forestry and Fire Protection 2023). All areas without color-coded polygons represent private ownership.

CHAPTER 2

RESOURCE SELECTION BY MOUNTAIN LIONS IN THE CENTRAL COAST OF CALIFORNIA

INTRODUCTION

Large carnivores are integral to the ecosystems and communities which they inhabit due to their roles as apex predators (Thorne et al. 2006, Barry et al. 2019). As human populations continue to expand and develop into wildlife habitat, however, the futures of many carnivore species throughout the world are in jeopardy primarily due to habitat fragmentation and loss and wildlife-livestock conflict (Ripple et al. 2014). Currently, an estimated 61% of large carnivore species worldwide are categorized as Vulnerable, Endangered, or Critically Endangered by the International Union for Conservation of Nature (IUCN 2019). Due to their extensive home ranges and generalist habitat and dietary selections, large carnivores, especially wild felids, are often considered “umbrella species” for habitat suitability assessments and conservation (Thorne et al. 2006, Barry et al. 2018, Dellinger et al. 2020). With this in mind, a decline in their population numbers impacts not only large carnivore species directly, but many other wild species.

Mountain lions (*Puma concolor*) are the largest wild felid species in the United States and Canada, and though the species as a whole is not threatened in Western North America, the futures of certain local populations are uncertain due to habitat loss, degradation and fragmentation, and restricted gene flow among local populations (Benson et al. 2016, Dellinger et al. 2020, Gustafson et al. 2022). Currently continuously ranging from the Canadian Yukon to near the southern tip of South America (Pierce and Bleich 2003, IUCN 2017), they once also inhabited the eastern United States before lethal removal, loss of prey, and urbanization extirpated them from most of this range (Burdett et al 2010). Presently the only eastern population remaining is that of the highly isolated Florida panthers (*Puma concolor coryi*; Gustafson et al. 2017) listed as Endangered under the Endangered Species Act.

Collectively, mountain lions in California maintain high genetic diversity, however, several coastal populations have the lowest genetic diversity estimates of any genotyped

mountain lion populations in the United States, aside from the Florida panthers (Ernest et al. 2014, Riley et al. 2014, Gustafson et al. 2017). These populations include those in the Santa Ana, Santa Monica, and Santa Cruz Mountains. This is largely attributed to major highways such as interstate I-15 and I-405 in Southern California and US-101 in both Southern California and the Santa Cruz Mountains, and associated development creating nearly impermeable barriers to gene flow and islands of minimally connected habitat. The result has been severely isolated populations with inbreeding depression (Vickers et al. 2015, Riley et al. 2021, Gustafson et al. 2022) as well as low observed heterozygosity (<0.5) and allelic richness values (<3.00 ; Table 2.10; Gustafson et al. 2019). Recent research has called for wildlife managers to incorporate genomic data and connectivity into habitat conservation efforts to avoid leading other currently stable populations down the same path (Dellinger et al. 2020, Gustafson et al. 2022).

Building on Gustafson et al.'s (2017) findings, Dellinger et al. (2020) concluded that a minimum of 14,591 km² of overall suitable habitat is required to maintain an effective population ($N_e \geq 50$) and diminish the negative effects of genetic drift and inbreeding. Only 7,923 km² is necessary if the habitat is protected. Situated between the Santa Cruz and Santa Monica Mountains, Dellinger et al. (2020) found that the area Gustafson et al. (2017) designated as the "Central Coast Central" (hereafter, the Central Coast and encompassing San Luis Obispo, Monterey, San Benito, Santa Barbara, and northern Ventura (north of State Route 118) Counties) currently maintains an estimated 16,355 km² (33%) of suitable mountain lion habitat. Only 6,780 km² (14% of total area, 41% of overall suitable habitat) is currently protected (Dellinger et al. 2020). Presumably because of the large amount of suitable habitat, the mountain lion population in the California Central Coast represents a stable population and provides "stepping-stone" connectivity between the three populations in this part of the state. As humans continue to expand across and develop the landscape (Alcamo et al. 2006, Ellis et al. 2010, Wuebbles et al. 2017, Wilson et al. 2020), conservation of this area has become critical to the persistence of all three populations (Gustafson et al. 2022).

With its Mediterranean climate offering year-round moderate temperatures and plenty of unregulated groundwater, the Central Coast supports a substantial amount of California's

agriculture, rangeland, and wildlife habitat. Monterey County ranked 4th out of the state's 58 counties in terms of gross value of agricultural production for 2019-2020, while Ventura, Santa Barbara, and San Luis Obispo all ranked within the top 15 (California Department of Food and Agriculture 2021). The Central Coast comprises 42,992 km², of which 70% is currently zoned for agriculture, 13% for open space, 8% for public facilities, 3% for zoning which does not fall into a particular major category on its own (e.g. institutional, recreation, etc.), 3% for residential development, 0.20% for industrial purposes, and 0.06% for commercial purposes. While agricultural zoning in and of itself is not necessarily incompatible with wildlife habitat - indeed, livestock grazing can help to shape communities and allow wildlife to disperse and migrate through rangelands due to the preservation of native grasslands and shrublands - sedentarization (with or without crop rotation) and land use change toward more intensive agriculture (e.g. row crops, orchards, vineyards, etc.) in recent years often leads to landscape degradation (Merenlender 2000, Hilty et al. 2019). Perennial crops such as orchards (e.g. avocados, tree fruits, and nuts) and vineyards in particular are expected to experience the largest expansion and lead to the greatest conversion of natural and rangeland to cropland (California Department of Food and Agriculture 2021).

Orchards and vineyards, which thrive in Mediterranean climates, are a direct threat to wildlife habitat conservation as many are being established across foothill grasslands, oak woodlands, and riparian corridors (Hilty and Merenlender 2004, Roehrdanz and Hannah 2016). Research has found that many native species tend to avoid heavily-altered landscapes, as opposed to less-altered landscapes such as pastures and rangelands, which can limit even highly mobile species' abilities to obtain resources such as cover, forage, water, etc. and retain genetic diversity (Machtans et al. 1996, Hilty and Merenlender 2004, Šálek et al. 2009, Wilmers et al. 2013, Vickers et al. 2015, Riley et al. 2021). Hilty and Merenlender (2004) investigated mammalian predator use of vineyards versus riparian corridors in Sonoma County, an area dominated by mixed-oak woodland, and found that detection rates of native predators, including mountain lions, on remote cameras were 11-fold higher in corridors.

In order to effectively conserve mountain lion habitat and populations in the Central Coast, habitat selection patterns needed to be evaluated to assist in determining which areas should be prioritized for conservation. Though intensive research has focused on California populations adjacent to dense human populations such as those in Santa Cruz and several southern California regions (Riley et al. 2021, Dellinger et al. 2019, Zeller et al. 2017, Benson et al. 2016, Riley et al. 2014, Wilmers et al. 2013), to date there have been no published studies of GPS collared mountain lions in the Central Coast.

The objectives of this study were to (1) use location data from collared mountain lions in the Central Coast to characterize home ranges in terms of size, overlap, and composition through comparison of several analysis methods, (2) develop a multi-scale resource selection function at the within-home range scale (i.e. Johnson's (1980) third order habitat selection) to determine which variables contribute to suitable mountain lion habitat in the Central Coast, (3) project this model over the greater Central Coast with land use and ownership GIS layers and predict where the greatest potential development threats and conservation priorities may be.

METHODS

Study Site

The Central Coast of California was defined according to the geographic extent of the “Central Coast Central” population of mountain lions described by Gustafson et al.'s (2019) genetic study (hereafter, the Central Coast; Figure 2.1). Though the extent of this population is not defined by county boundaries, ultimately most land use and conservation decisions would be. Therefore, I included San Benito, Monterey, San Luis Obispo, and Santa Barbara Counties in their entirety, while including only Ventura County north of SR 118 (hereafter Ventura County) due to the presence of the “Central Coast South” mountain lion population in the southern portion of the county.

The Central Coast is characterized by Mediterranean climates with hot dry summers and cool winters with an average of 76-114 cm of precipitation annually (usclimatedata.com). The Central Coast consists of rugged topography with ridges up to 1,780 m running parallel to the coast interspersed with valleys, drainages, and hills (California Department of Fish and Wildlife 2015). Vegetation within these counties primarily consists of mixed native and non-native chaparral and shrub species, savannas, non-native grasslands, oak woodlands, and mixed coniferous forests. Riparian and wetland habitats are also prevalent throughout (California Department of Fish and Wildlife 2005, Thorne et al. 2006), as well as agriculture, vineyards, and pastureland. Urban development in this region has historically been concentrated along coastal lowlands, with crop production focused on valley floors and grazing primarily occurring in surrounding foothills and montane regions. In more recent years, however, human population growth has steadily increased, and pressures from urban and agriculture have sprawled toward the coastline and more interior regions. Large patches of undeveloped natural habitat still remain; however, intensive agriculture, overuse of water resources, and nonnative species invasions represent the greatest threat to these habitats and the biodiversity which relies on them (Ca. Dept. of Fish & Wildlife 2015).

Fort Hunter Liggett

Fort Hunter Liggett (FHL) Army base in southern Monterey County was selected to represent the coastal ecoregion of Central California in terms of climate, topography, and natural habitat. It encompasses approximately 982 km² of nearly contiguous natural habitat (Figure 2.2), and is bordered by the Salinas Valley to the north, the Santa Lucia Mountains to the east, San Luis Obispo County to the south, and the Los Padres National Forest to the west. It is an active Army military base primarily utilized as a training area and airspace for field, live fire, and institutional training for reserve and active troops. The dominant habitats found on FHL include coastal oak woodland including coast live oak, valley oak, and blue oak woodlands, grassland habitats, and Maritime chaparral/coastal scrubland (Integrated Natural Resources Management Plan/Environmental Assessment, 2012). Of the approximately 982 km² that FHL encompasses, chaparral/shrubland habitat comprises approximately 363 km² (Figure

2.3A) with primary species including California sage (*Artemisia californica*), *Salvia spp.* (true sage species), *Arctostaphylos* (manzanitas), *Ceanothus spp.*, coyote bush (*Baccharis pilularis*), among others. Coast oak woodland comprises 320 km² of the Base's area and thrives in both warm, dryer climates as well as cool, moist climates. It is characterized by such species as coast live oak (*Quercus agrifolia*), blue oak (*Q. douglasii*), valley oak (*Q. lobata*), foothill pine (*Pinus sabiniana*), California bay laurel (*Umbellularia californica*), poison oak (*Toxicodendron diversilobum*), and pacific madrone (*Arbutus menziesii*). Grassland comprises 174 km² of the understory and includes needle grass (genus *Nassella*), California melic (*Melica imperfecta*), and June grass (*Koeleria macrantha*). Most native wildlife species are supported and present on the base as well. FHL also has ongoing wildlife habitat improvement and enhancement projects year-round. These include controlled burns and herbicide treatments to manage invasive species, annual planting of 75-100 oak trees from acorn to combat poor regeneration due to drought, disease, and damage from military training, modifications to existing fencing to allow for wildlife movement, and maintenance of natural and man-made water sources (sportsman, n.d.).

Field Methods

Between January 2018 and April 2019, the California Department of Fish and Wildlife caught and placed Vectronic global positioning system (GPS) radio-collars with VHF capabilities on seven (five females, two males) mountain lions on FHL. All seven animals were of adult age (2+ years). Lions were captured using cage traps and trained hounds, then anesthetized with Telazol® (tiletamine HCl and zolazepam HCl; Fort Dodge Animal Health, Fort Dodge, IA, USA) or a combination of ketamine and medetomidine or xylazine. Captures were led with the approval of a CDFW wildlife veterinarian under guidelines of CDFW's animal care and use policy (CDFW Operations Manual Policy 149). Collars were programmed to obtain GPS locations ("fixes") every 3 hours with an expected location accuracy of ± 3 m.

Data Analysis

One female died in June 2020 of unknown causes; two other females' collars failed prematurely (Table 2.1). Telemetry data for the remaining collared lions was truncated on July 11, 2020 to incorporate at least 1 year of data. I removed all fixes which either failed to acquire locations or with a dilution of precision (DOP) value ≥ 7 , a value generally accepted as unreliable (Huihui et al. 2008, Isik et al. 2020). Large DOP values can result from the receiver on the GPS collar receiving information from multiple satellites in close proximity to each other, resulting in inaccurate spatial information. Factors such as dense canopy cover, canyons, mountains, and large man-made structures can also obstruct the satellites' signals, resulting in higher DOP values. To ensure that filtering the points with DOP values ≥ 7 did not introduce bias against habitat with the aforementioned characteristics, I mapped these points in ArcGIS over the 2015 Forest Service CalVeg GIS layer and extracted all habitat type values to the data points *ad hoc* before calculating the number of points which fell into each habitat to determine if there was a disproportionate amount of data from densely vegetated, canyon, and/or habitat within close proximity to large structures being filtered out. I did not detect evidence of such bias, and the number of locations per lion, after filtering, ranged from 1,647-4,287 for the females and 4,159-4,204 for males (Table 2.1).

Defining Home Ranges and Utilization Distributions

I calculated and compared 95% minimum convex polygons (MCPs) with adaptive local convex hulls (a-LoCoH), which both generate the smallest convex polygon (or hull) containing a specified portion of all used locations (Getz and Wilmer 2004, Rose 1982) to provide an estimate of the HR's outer boundary as well as the total area being used. All mountain lions included were > 2 years old (i.e. adults established in their home ranges), and thus I expected all location data to be representative of their true home ranges. MCPs and HRs calculated via a-LoCoH were done using the *adehabitatHR* package in R (v3.6.1).

Once HRs had been estimated, I fit utilization distributions (UDs) at 95%, 75%, 50%, and 25% use intensity using parametric KDEs. KDEs estimate the underlying probability

density functions from the data by first placing a kernel over each observation point using a smoothing parameter, h , which is the width of each kernel and may be modified based on distances between individual points (Silverman 1986, Horne and Garton 2006), or may be a “best” fixed value at each point (Getz and Wilmers 2004). To estimate h , I used the *ad hoc* method for a bivariate normal kernel (Calenge 2006), wherein:

$$h = \sigma n^{-1/6}$$

and

$$\sigma^2 = 0.5(\text{var}(x) + \text{var}(y))$$

These polygons were overlaid onto my various covariate layers in ArcMap to quantify and visualize each mountain lion’s resource use.

Finally, I also estimated 95%, 75%, 50%, and 25% isopleths using adaptive-local convex hulls (a-LoCoH), wherein a was the maximum distance between any two points in the dataset, irrespective of time stamps of locations (Getz et al. 2007). As data collection technology continues to advance, a-LoCoHs are becoming more widely regarded as more accurate for analyses of both home ranges and UD, particularly for large mammals (Getz et al. 2007, Dellinger et al. 2019). Additionally, as Dellinger et al. (2019) employed this method to estimate 95% home ranges for mountain lions across California, and my research is intended to expand upon a subsection of their results, including their methods was key.

Covariates

I assessed the composition of the total HR and core areas using environmental variables which numerous studies have found represent mountain lion movement patterns, hunting style, and presence of prey in the western United States (Table 2.2; Pierce et al. 1999, Burdett et al. 2010, Wilmers et al. 2013, Benson et al. 2016, Dellinger et al. 2018, Dellinger et al. 2019). Mountain lions strongly select for habitats with low vegetation (e.g. shrubs and chaparral), tree cover, and riparian areas, which provide concealment for both hunting and for the prey itself. They generally avoid barren areas (i.e. habitat with

<2% total vegetation cover by herbaceous, desert, or non-wildland species; Parker and Maytas 1981) and grasslands which do not provide adequate cover for stalking and/or escape; however, they will sometimes use them as travel corridors. Further, adult mountain lions generally select for moderate slopes, avoiding gentle and steep slopes (Dellinger et al. 2020, Dunford et al. 2020, Riley et al. 2021), and select for habitat nearer to tertiary roads (i.e. dirt roads) when available as traveling via road is more energetically-efficient for lions as well as their prey.

I assessed 5 biotic variables: distance to tree cover (m), distance to oak woodland (m), distance to shrub/chaparral cover (m), distance to grassland/barren habitat (m), and habitat suitability for deer, lions' primary prey (1-3 index). Vegetation types were defined according to the U.S. Forest Service CalVeg GIS layer (Table 2.2). I used a data layer of the suitability of deer habitat in California collected by the CDFW as a proxy for presence of prey.

I aggregated all tree habitats (e.g. woodland, forest, etc.) into one variable for "tree covered" habitat and did similarly for oak species only into "oak woodland" habitat. Distance to tree cover and distance to oak woodland were both included as covariates because while oak woodland makes up a significant portion of the habitat types on FHL and my focus was centered on oak woodlands in the Central Coast, many other arboreal species also contribute to the overall canopy cover on the base. I similarly aggregated shrub and chaparral species into "shrubs/chaparral" habitat, as the differences between the two types of vegetation were negligible for the purposes of this study. I combined grassland and barren land habitat types into one variable ("barren/grassland") as both represented sparsely vegetated areas on FHL. Unlike many sites where mountain lion resource selection has been studied in California, FHL contains very little anthropogenic development, urban or agricultural (Figure 2.3A,B). Thus, neither of these habitat types were included in analyses.

I also assessed 3 abiotic variables: slope (%), distance (m) to perennial water sources, and distance (m) to tertiary roads. I derived slope from the digital elevation model using the Slope tool in ArcMap (v10.8.1; Table 2.2). I used GIS layers from the U.S. Geological Survey for both perennial water sources and tertiary roads, respectively. Primary and/or

secondary roads did not occur within any of the home ranges of the study lions so these road types were not included in my analyses.

Resource Selection Functions

To quantify resource selection patterns of mountain lions in the Central Coast, I developed resource selection functions (RSFs) in a use-availability framework (Manly et al. 2002). RSFs are commonly used in resource selection analyses for wildlife because they are one of the simplest methods of estimating resource selection and are applicable at multiple scales and orders of habitat selection (Boyce 2006, Zeller et al. 2014, Zeller et al. 2017, Dellinger et al. 2019). They are executed by comparing used and randomly generated available units (in this case, habitat features corresponding to GPS locations and random points within the HR).

I defined locations from my study animals' GPS collars as "used" and randomly generated points as "available" for each mountain lion. Under a systematic random sampling framework (Lashley et al. 2018, Mekonen 2020) to avoid systematic and random sampling biases, respectively, I created a grid of 100 m² hexagons using the Generate Tessellation tool in ArcMap (v10.8.1) within each lion's home range, and then randomly placed one data point per cell, using the Create Random Point tool.

I extracted distances to biotic and abiotic covariates from each used and available location at the within-home range scale using the Euclidean Distance tool in ArcMap. This corresponded with Johnson's (1980) third order of selection, which addresses relative use of resources within the home range. Polygons and polylines were created for each habitat type, perennial waterway, and tertiary road, respectively, before I calculated distances between each location and each covariate in a Euclidean distance analysis framework (Benson 2013). Deer habitat suitability index values and slope percentages were extracted at the exact point. All variables were analyzed at a 30 x 30 m resolution (Dellinger et al. 2019).

Once the relevant data had been extracted from each applicable unit, a logistic regression with a logit link was used given by the equation:

$$\text{logit}(x) = \beta_0 + \beta_1 x_1 + \beta_2 x_2 \dots \beta_p x_p$$

where $\text{logit}(x)$ is the logit link given by the natural logarithm of the odds of x_1 :

$$\ln(e^x/(1-e^x))$$

$\text{logit}(x)$ represents the relative probability of use for a given location within a mountain lion's home range, β_0 represents the intercept, and β_i represents the coefficient for the covariate $x_1, x_2 \dots x_p$.

I standardized all variables except for deer habitat suitability to be able to compare their coefficients within my final models. Standardization transforms data so that it is normally distributed with a mean of 0 and a standard deviation of 1 and allows coefficient estimates derived from data on different scales to be more easily interpretable and comparable to each other.

I standardized the variables using:

$$x' = \frac{x - x_{\text{mean}}}{\text{std}(x)}$$

where x' is the standardized feature value, x is the original feature value, x_{mean} is the mean of the original feature value, and $\text{std}(x)$ is the standard deviation of the original feature value. I did not perform this on the deer habitat suitability index as the values (i.e. 1-3, or low, medium, high; Gogol-Prokurat 2017) were already on a scale comparable to that of the other variables after standardization.

I used logistic regression and univariate models using the *lme4* package to assess the need for quadratic variables. The purpose of this was to detect possible thresholds and/or nonlinear relationships in the relationships between predicted probability of use and predictor variables. I then checked for collinearity among the covariates using the Spearman correlation method. When two variables were highly correlated ($|r| > 0.60$; Dellinger et al. 2019), I examined the global model, first removing one of the correlated variables from the model, then replacing it and removing the other, comparing Akaike's Information Criterion corrected for small sample sizes (AIC_c) for each. The variable in the best fitting model (i.e., the one with the lowest AIC_c value by a ΔAIC of at least 2.0) was retained (Burnham and Anderson 2002) and the other variable was dropped. Once all

covariates with high collinearity had been removed, I used the *car* package to check for high variance inflation factor (VIF) values, an indication of high multicollinearity between variables. Variables with VIF values >4 were regarded as highly correlated with other variables. I again compared AIC_c values of the models with each of the correlated values and retained the variable in the model with the lowest score. Collinearity among all covariates was reassessed until high collinearity was no longer detected.

I then began building RSFs at the within home range scale using logistic regression and mixed effects models (Boyce et al. 2002, Burnham and Anderson 2002, Manly et al. 2002, Dellinger et al 2019). I used mixed-effects models to account for differences in used and available data for the home ranges of my study mountain lions and included a random effect for individual study animals to account for the unequal number of GPS locations for each lion. Being that GPS locations are sequential spatially as well as temporally, I attempted to control for Type I errors stemming from autocorrelation by selecting and interpreting results from the most parsimonious models rather than the statistical significance of variables within the models (Boyce 2006, Dellinger et al. 2019). The location error of my GPS data was ± 3 m, less than the resolution used for any of the covariates (30 x 30 m). Location error had little impact on my analyses.

Model Evaluation

I used k -fold cross-validation to evaluate the predictive performance of my RSFs (Fielding and Bell 1997, Hastie et al. 2001, Johnson et al. 2006, Dellinger et al. 2019). Boyce et al. (2002) proposed that k -fold cross-validation is a more appropriate assessment of model predictive performance for RSFs in a use-availability framework over other methods for validating logistic regression such as ROC, Hosmer-Lemeshow goodness-of-fit, or percent correctly classified. I randomly selected 80% of the used and available location data (model-training data) for all seven study lions to create the RSF and withheld the remaining 20% (model-testing data) to evaluate the predictive performance. I repeated this process a total of five times to ensure that the entirety of the data could be used to not only train the RSF, but also assess the model predictions. This resulted in 5-fold cross-validation.

Typically, model-testing data sets can be assessed against the RSF predictions from the model-training data sets using the correlations between the RSF values' bin rank and the frequency of independent, withheld observations falling into the same bin rank (standardizing for area). Johnson et al. (2006), modified this approach to improve the precision of evaluation and assess the assumption that the model is reasonably proportional to probability of use.

This modified approach is performed for each of the respective training/testing data sets as follows:

1. Use logistic regression to calculate a RSF with the model-training data.
2. Project RSF coefficients in a GIS using the logit link function. This results in raster pixel values ranging between 0 and 1.
3. Reclassify each pixel into ordinal bins ranging from 1 to 9 (e.g. 0.0-0.1 becomes 1, 0.10001-0.2 becomes 2, and so on), representing low (i.e. 1) to high (i.e. 9) relative probability of habitat use for mountain lions (Holbrook et al. 2017, Dellinger et al. 2019).
4. Obtain the midpoint value of raw RSF scores for each RSF bin.
5. Calculate the utilization value $U(x_i)$ for each bin i :

$$U(x_i) = w(x_i)A(x_i) / \sum w(x_j)A(x_j)$$

where $w(x_i)$ equals the midpoint RSF of bin i and $A(x_i)$ is the area of bin i (Boyce and McDonald 1999).

6. Overlay withheld testing-data onto the projected RSF and total the number of used observations that fall into each RSF.
7. Using the following formula, quantify the expected number of validation observations (N_i) within each ordinal bin:

$$N_i = N \times U(x_i)$$

where N is the total count of used observations in the testing-data and $U(x_i)$ is the utilization value from step 4.

8. Use linear regression to compare expected number of used points (N_i) for each bin from step 7 to observed number from step 6. Determine if the slope of the regression line significantly differs from zero (use equals availability and the model does not differ from a random model) or 1.0 (the model is proportional to the probability of use). Finally, determine if the intercept is significantly different from zero (an intercept of zero would be expected for a model proportional to the probability of use).
9. Perform a Spearman rank correlation (r_s) test to estimate correlation between expected and observed number of used observations in each ordinal bin. A correlation coefficient of $|r_s| > 0.6$ as well as a similarly strong R^2 suggest that the model has a strong capability of predicting relative probability of habitat use by a mountain lion (Johnson et al. 2006, Dellinger et al. 2013).

To summarize, a model with a slope significantly different than 0 but not 1.0, with an $R^2 > 0.6$ and Spearman rank correlation (r_s) between number of observed and expected used locations also > 0.6 indicates a model with good predictive capability (Johnson et al. 2006).

Once I had determined if my final RSF model had strong predictive capability, I projected it over FHL and characterized habitat as either suitable or unsuitable. I first estimated the threshold relative probability of use of my final RSF model which captured 90% of all used locations and considered any value for probability of use above that threshold suitable, while anything below was unsuitable (Hebblewhite et al. 2014, Holbrook et al. 2017, Dellinger et al. 2019).

Projection of Model to the Central Coast Region

I then projected my model within the bounds of the greater Central Coast region and used the threshold probability value to determine the suitable/unsuitable habitat across it. To visualize areas of suitable mountain lion habitat most likely to be developed in the future, I overlaid my habitat suitability raster with zoning GIS layers, including agriculture, commercial, residential, industrial, open space, and public facilities, for each Central

Coast county as well as a layer specifying protection status and land ownership (i.e. public and land currently under conservation easements; California Protected Areas Database 2022). The Channel Islands were excluded from this exercise as they are not a part of mountain lions' natural range. I then calculated the area of each zoning category and compared this with the amount of suitable habitat and land protection status. Lastly, I compared these values to the amount of habitat Dellinger et al. (2020) found to be required to alleviate the negative impacts of genetic drift and inbreeding.

RESULTS

Characterization of Home Ranges: Overall Area and Within-Home Range Core Use Areas

Overall Area

Home range size varied between sex and method used, with areas calculated with a-LoCoH being larger than those calculated with MCPs for all study lions.

Using the 95% MCP method, the five female lions' home ranges averaged 187.07 km² (SE = 19.61) while the two males' home range sizes averaged 1,158.25 km² (SE = 133.54; Table 2.3). Using 95% a-LoCoH, female home range sizes averaged 210.26 km² (SE = 27.98; Table 2.3). Males' home range sizes averaged 1,185.05 km² (SE = 209.97; Table 2.3).

On average, FHL is comprised of 42.3% shrubland/chaparral habitat, 37.3% tree-covered habitat, 20.3% grassland/barren, and 0.02% habitat classified as "other" (urbanized facilities for Army personnel, etc.). Vegetation composition proportions within the mountain lions' home ranges reflected this and were extremely similar, regardless of HR estimation method (Table 2.4). HRs were, on average, composed of 36.6% shrubland and/or chaparral, 35.0% tree-covered habitat, and 26.0% grassland/barren habitat. "Other" habitat types comprised 0.17%.

Overlap Using the results for HR boundaries estimated with 95% a-LoCoH, the female mountain lions' HRs overlapped between 0% (no overlap) to 46% (SF1 with SF3; Table 2.7; Figure 2.8), while overlap between the two males was between 54% (SM3 with SM4) and 79% (SM4 with SM3; Figure 2.9). All females' HRs overlapped with both males', with overlap ranging from 16% (SF2 with SM4) to 99% (SF5 with SM3; Figure 2.10A).

Core Use Areas

For females and males, respectively, the average combined areas of the 75% isopleths were very similar. Regardless of whether KDI or a-LoCoH was used, for female lions, there was <1% difference in isopleth area when estimated by KDI versus a-LoCoH (Table 2.5), and there was a 1% difference for males. Both methods also confirmed similar habitat composition patterns. Female isopleths were dominated by shrubland/chaparral, followed by tree-covered habitat, grassland/barren habitat, then habitat classified as "other" (Table 2.6, Figure 2.7). The largest portions of male isopleths were tree-covered habitat, followed by grassland/barren habitat, shrubland/chaparral habitat, and then "other" habitat.

Average combined areas for 50% isopleths were also similar for females and males, respectively, with KDIs 7-8% larger than isopleths estimated with a-LoCoH (Table 2.5). Females maintained one to three core areas at this level for both methods (Figure 2.6). These were dominated by tree-covered habitat, closely followed by shrubland/chaparral, grassland/barren habitat, and "other" (Table 2.6, Figure 2.7). 50% of SM3's locations were spread between two core areas, while SM4's points were contained within one area (Figure 2.6). The largest proportions of habitat for both males within these areas, in descending order, were grassland/barren, tree-covered, shrubland/chaparral, and "other" (Table 2.6, Figure 2.7). There was no female-female overlap at this level (Figure 2.8); however, four of the females overlapped with SM3's 50% core areas, and two overlapped with SM4's (Figure 2.10B). SM4 overlapped almost entirely with SM3 (Figure 2.9).

For both female and male lions, the average combined areas for 25% isopleths were about 30% larger when calculated with KDIs versus a-LoCoH (Table 2.5). Females used

between one and four core areas at this level (Figure 2.6). When isopleths were estimated using KDIs, the largest proportion was tree-covered habitat, followed by shrubland/chaparral, grassland/barren, and “other” (Figure 2.7). a-LoCoH isopleths also followed this pattern but with shrubland/chaparral as the highest proportion and tree-covered habitat next (Table 2.6). There was no female-female overlap at the 25% level (Figure 2.9); however, SF3 and SF5’s core areas overlapped almost completely with SM3’s (Figure 2.10B). Similar to their 50% core use areas, the majority of SM4’s 25% isopleth overlapped with SM3’s (Figure 2.9).

Within Home Range Resource Selection

The collared mountain lions on Fort Hunter Liggett exhibited negative relationships with increasing distance from all variables where distance was measured, and they exhibited a positive relationship with increasing suitability of deer habitat (Table 2.8 and 2.9).

Distance from shrub cover, if there was also a perennial water source nearby, had the largest negative impact on the predicted probability of a mountain lion using a given location within their home range ($\beta_{\text{shrub}*\text{water}} = -0.780$, $\text{SE} = 0.072$; Table 2.9). Distance to vegetative cover (i.e. shrub and/or tree cover; $\beta_{\text{shrub}} = -0.619$, $\text{SE} = 0.035$, $\beta_{\text{trees}} = -0.496$, $\text{SE} = 0.015$) as well as distance to perennial water sources ($\beta_{\text{water}} = -0.583$, $\text{SE} = 0.050$) had similarly negative effects on predicted probability of use (Figure 2.11B, C, E). Mountain lions were less likely to use areas of increasing slope or sites further from tertiary roads (Figure 2.11A, D), though neither of these had a strong influence on selection ($\beta_{\text{slope}} = -0.070$, $\text{SE} = 0.008$, $\beta_{\text{tertiaryroad}} = -0.037$, $\text{SE} = 0.009$, respectively). Predicted probability of use increased as the suitability of habitat for deer increased from low to high ($\beta_{\text{deer}} = 0.360$, $\text{SE} = 0.010$; Table 2.8, Figure 2.11F).

Distance to grassland/barren habitat was not included in the final model as it was strongly correlated with distance to shrub cover ($r_s = -0.62$), and the model containing distance to shrub cover yielded a lower AIC_c . Distance to oak woodland was also not included in any of the final models as it was, predictably, strongly positively correlated with distance to tree cover (which included all local tree species; $r_s = 0.96$). The model with distance to

tree cover resulted in a lower AIC_c. Quadratic variables were necessary for distance to both tree and shrub cover (Table 2.8). The ΔAIC_c between the top model and all others was >2 (Table 2.8), thus it was reasonable to conclude that this was the best fitting model.

Using 5-fold cross validation, I confirmed that the slope of the final model was significantly different from 0 but not 1.0. The R^2 value of the final top model was 0.96 and the Spearman rank correlation value between the expected and observed number of used observations was $r_s = 0.96$, giving further support to my best fitting model (Johnson et al. 2006).

Applying Resource Selection Function

Fort Hunter Liggett

Predicted habitat suitability values within FHL boundaries ranged between 0.009-0.86 (Figure 2.12). Using a 90% threshold relative probability of use of 0.75 (Dellinger et al. 2019), I estimated 682 km² of suitable habitat on the Base. This constitutes 69% of the area within FHL boundaries with almost continuously suitable habitat, which was located in the northern and southwestern portions of the Base.

California Central Coast

Using a 90% threshold relative probability of use, I predicted that habitat with a relative probability of use ≥ 0.75 is suitable mountain lion habitat and estimated that 17,685 km² can be found throughout the greater Central Coast. Habitat suitability values ranged from 1.11×10^{-32} to 0.93 (Figure 2.13). The Salinas Valley was omitted from the projection as the predicted deer habitat GIS layer did not include data for this area (Gogol-Prokurat 2017).

I estimated 17,685 km² of suitable mountain lion habitat throughout the Central Coast (Table 2.12). Of this, 11,338 km² (64%) of suitable habitat fell within agricultural-zoned land, 2,504 km² (14%) was within land zoned as open space, 2,000 km² (11%) was

within land zoned for public facilities, 290 km² (2%) was within land which fell under the category of “other” (e.g. specially-planned projects, etc.), 497 km² (3%) is within residentially-zoned land, 14 km² (0.08%) is within land zoned for industrial purposes, and 5 km² (0.03%) falls within commercially-zoned land (Table 2.12).

Monterey County represented the largest amount of suitable habitat, followed by San Luis Obispo, Santa Barbara, San Benito, and northern Ventura Counties (Table 2.12); however, many areas of highly suitable habitat are already fragmented to varying degrees by human development. For example, highly suitable habitat can be found in the majority of western San Luis Obispo and Santa Barbara Counties, but several state highways as well as medium-highly developed towns such as Atascadero, San Luis Obispo, Santa Maria, Lompoc, etc. break many of these areas up (Figure 2.14). Highway 101 also represents a formidable barrier between habitat in these counties, but it, as well as towns such Salinas and Monterey proper, are of particular concern where northern Monterey County borders Santa Cruz County and highly-suitable habitat exists. Though eastern and western San Benito and northern Ventura Counties encompass suitable habitat and currently have lower levels of development due to each county being mainly agriculture or public land, respectively, each county has at least one state highway bisecting the habitat. San Juan Bautista and Hollister are also situated on San Benito’s northern border with Santa Cruz.

A total of 7,367 km² (41%) of this overall suitable habitat is currently protected from development (Table 2.13). Of this protected habitat, 2,691 km² (37%) is zoned for agriculture, 2,294 km² (31%) falls under open space zoning, 1,177 km² (16%) is zoned for public facilities, 159 km² (2%) is land zoned for “other” purposes, 9 km² (0.12%) falls within residentially-zoned land, and 0.03 km² (~0%) is zoned for commercial uses. Conserved land in Monterey County accounts for 1,037 km² (14%). In addition to containing the most overall suitable habitat, Monterey County also has the largest amount of protected habitat, followed by San Luis Obispo, Santa Barbara, northern Ventura, and San Benito Counties (Table 2.13).

Subtracting the area of protected suitable habitat from overall suitable habitat gives the amount of suitable habitat for mountain lions which is currently unprotected. I estimated

that 10,316 km² of the Central Coast is suitable for mountain lions but unprotected (Table 2.14), with 8,647 km² (84%) agriculturally-zoned, 210 km² zoned as open space (2%), 823 km² (8%) zoned for public facilities, 129 km² (1%) zoned for “other” purposes, 488 km² (5%) zoned as residential, 14 km² (0.14%) zoned as industrial, and about 5 km² (0.04%) is under commercial zoning. With the largest area of overall and protected suitable habitat, Monterey County also possesses the most unprotected habitat, followed by San Luis Obispo, Santa Barbara, San Benito, and northern Ventura Counties (Table 2.14).

DISCUSSION

Home Ranges and Utilization Distributions

Ecologists have several motives for defining and analyzing home ranges for wildlife. Mapping home ranges can elucidate the spatial extent of movement patterns, boundaries, and/or overlap between individuals or groups of animals, all of which may aid in estimating carrying capacity for a given habitat or location. Ecologists may also map home ranges to analyze the utilization distributions (UDs), or estimates of the probability of an animal or group of animals being found in any part of their home range at a given time (Getz and Wilmer 2004). By investigating UD, ecologists are able to better understand resource use, and thus how animals use the landscape, within the extent of the HR.

Home Ranges

Size HR area measurements resulting from 95% a-LoCoH yielded 12.4% larger HR areas for female lions than those calculated with MCPs and 2.3% larger HR areas for male lions, while also allowing for more detailed boundaries for all study mountain lions (Table 2.3). Visualizations of the data also showed oversimplified HR perimeters for MCPs, encompassing both large areas where there were no GPS fixes even at a relatively coarse fix rate of 3 hours, as well as groups of outlying points likely representing exploratory movements (Figure 2.4).

My results for HR size between adult male mountain lions versus adult females, for both MCP and a-LoCoH methods, were generally consistent with research from other lion populations in the western United States (Criffield et al. 2018, Dellinger et al. 2018, Karelus et al. 2021, Riley et al. 2021). For mountain lions, sex plays an especially critical role (Grigione et al. 2002) in HR size, with males tending to have larger ranges than females, which can provide more hunting opportunities to sustain their greater body masses but may also be a reproductive strategy to allow for overlap with multiple females (Sillero et al. 2021). Smaller HR sizes for females may be reflective of fewer resources needed to sustain smaller body sizes, less need to overlap with potential mates, and periods when they are producing and caring for young (Grigione et al. 2002, Elbroch et al. 2016). Indeed, this pattern has been observed in home range analyses for mountain lions within the Trans-Pecos region of Texas (average male HR: $1,078 \pm 219 \text{ km}^2$; average female HR: $284 \pm 38 \text{ km}^2$; Karelus et al. 2021), Santa Monica Mountains and surrounding areas in Southern California (average male HR: $372 \pm 103 \text{ km}^2$; average female HR: $134 \pm 22 \text{ km}^2$), southwestern Idaho/northwestern Utah (average male HR: 100-500+ km^2 ; average female HR: 30-100+ km^2 ; Laundré and Loxterman 2007), and in the central Sierra Nevadas of California regardless of season (average male HR: 249 km^2 (summer), 121 km^2 (winter); average female HR: 131 km^2 (summer); 56 km^2 (winter); Dellinger et al. 2018).

Male and female mountain lions on FHL had larger average home range sizes than the aforementioned populations, with the exception of those found in West Texas (Table 2.3). HR size can vary widely across mountain lion populations due to multiple factors including habitat type and location, resource availability, body mass, season, and population density (Nunez-Perez and Miller 2019). For example, Riley et al. (2021) attributed the HR size of mountain lions in the Santa Monica Mountains area to the risks posed by numerous freeways and urbanization throughout the area, while Dellinger et al. (2018) and Laundré and Loxterman (2007) speculated that home range sizes in the central Sierra Nevada and southern Idaho/northwestern Utah, respectively, may be explained by seasonal deer density, migration patterns, and hunting opportunities. The majority of the land in West Texas is either public land or privately owned ranches with a low human population density. Further, seasonal weather patterns are relatively mild,

resulting in ample year-round prey with limited shifts in seasonal distribution (Karels et al. 2021). Without as many risks imposed by human populations or challenges associated with following migrational prey in variable seasonal weather, mountain lions are less likely to restrict their ranges (Logan 2019). Due to the fact that this also closely describes conditions on FHL, it is likely that the similarity in home range sizes reflects this.

Habitat Composition Though area differed between male and female lions, the proportions of habitat types found within each were relatively similar regardless of sex. On average, shrubland/chaparral habitat made up the largest proportion of all HRs, followed by tree-covered, and grassland/barren habitat (Table 2.4). This also reflects the overall composition of FHL (Figure 2.3A, B). Previous research has found that mountain lion habitat selection is largely consistent across the Western United States. At the within-home range scale (Johnson's (1980) third order selection), mountain lions and their prey rely on vegetative cover for movement as well as concealment for and from ambush, respectively, and therefore select for habitat types with vegetation which aids in this (Logan and Irwin 1985, Wilmers et al. 2013, Nicholson et al. 2014, Dennison et al. 2016, Zeller et al. 2017, Dellinger et al. 2019, Gigliotti et al. 2019, Riley et al. 2021). Deer are often the primary prey for adult male and female lions when available, though male diets typically also include larger ungulates (e.g. elk, large wild boars (*Sus scrofa*), etc.; Wilckens et al. 2016, Knopff et al. 2010).

Other habitat, such as that urbanized by FHL Army personnel, made up a very small amount of the overall area (0.02%). Though Fort Hunter Liggett is a military field training base, and therefore comprises less human development than those in either the Santa Cruz Mountains or Southern California, there are still examples of this on site (e.g. commissary, lodging for staff and visitors, parking lots, etc.). HRs for all lions included in this study contained <1% "other" habitat, suggesting that if sufficient habitat undisturbed by humans is available, mountain lions will avoid that which is disturbed (Wilmers et al. 2013).

Utilization Distribution

Size The average area of the 95% HR estimated with a-LoCoH for female mountain lions was only about 1% larger than that of the 95% home range calculated using KDI methods, while the average area for males was about 2% larger when calculated using a-LoCoH versus KDIs (Table 2.3). Similarly, 75% isopleths calculated from a-LoCoH and parametric KDE methods were also very similar in terms of size with a < 2% difference between the methods for males and females, respectively (Table 2.5).

Discrepancies between the results from a-LoCoH and KDE methods were slightly larger when it came to 25% and 50% isopleths (i.e. core use areas). Female and male 50% isopleths calculated with KDIs were 7-8% larger than when calculated with a-LoCoH, respectively, while 25% isopleths were 27-29% larger when calculated with KDIs versus a-LoCoH (Table 2.5). In some instances where the a-LoCoH method estimated multiple smaller core use isopleths, the parametric KDIs connected them into a single larger isopleth of core use (Figure 2.6 SF2, SF4, SM3). The differences in isopleth size as well as multiple smaller isopleths versus one larger one were very likely due to the smoothing parameter. Ultimately, the utilization distributions calculated using a-LoCoH were more accurate to the lions' use of the landscape within their home ranges based on tighter boundaries around data point distributions (Figure 2.6); however, as FHL represents a large amount of habitat mostly unfragmented by anthropogenic and/or natural hard boundaries such as hydrographic or geological features, resulting UD's were very similar and selecting one method over the other was not critical. Because a-LoCoH adapts with increasing densities of points, allowing for concave edges, disjoint regions, and holes, while reflecting hard boundaries (Sillero et al. 2021), this method would be especially important over parametric KDEs for research investigating the impacts that anthropogenic fragmentation and natural impenetrable boundaries have on wildlife resource use at the within-home range level (Getz et al. 2007). It's also important to keep in mind that even with an elusive study species which exists in low densities, this sample size for comparative home range analyses was relatively small. Similar analyses done with data from more study animals, perhaps collared in areas with more topographic variation (anthropogenic or natural), would bolster conclusions.

Resource Selection

Proximity to tree cover, shrubland/chaparral, and habitat suitability for deer strongly influenced resource selection of mountain lions on Fort Hunter Liggett within their home ranges. My results are consistent with previous studies that have found that mountain lions inhabit almost any ecosystem in the western United States if there is adequate cover and prey (Nicholson et al. 2014, Zeller et al. 2017, Dellinger et al. 2019, Riley et al. 2021). Specifically in the context of this study, mountain lions in the Santa Cruz Mountains, a heavily forested region of California to the north of the Central Coast, have been documented selecting conifer forests and shrubland for movement (Wilmer et al. 2013). Mountain lions in the Santa Monica Mountains and Peninsular Ranges (i.e. Coastal Southern California to the south of the Central Coast) inhabiting habitats also dominated by mixed conifer species, oak woodland, and shrubland/chaparral, exhibited an especially strong affinity for shrubland/chaparral within their HRs (Dickson and Beier 2002, Burdett et al. 2010, Jennings et al. 2016, Zeller et al. 2017, Riley et al. 2021). Further, Dellinger et al. (2020) reported consistent selection of shrub cover in their analysis of California mountain lions resource selection as a whole, which suggests that there is a need for stronger recognition of the importance of this vegetation type for mountain lions.

In agreement with the aforementioned studies, 50-95% isopleths for male lions encompassed shrub/chaparral and tree-covered habitat (Table 2.6), but it was most notable that there was a considerable amount of edge habitat within the 25% isopleths for both males (Figure 2.7). Furthermore, these areas where ample tree and shrubland/chaparral border grassland/barren habitat coincided with medium-high quality deer habitat (Figure 2.9). Mountain lions typically avoid open areas in favor of cover, effectively selecting for areas where probability of a successful hunt is greatest over areas where encounter rates are high (Dellinger et al. 2020, Riley et al. 2021). Additionally, it is well-documented that wild carnivores often select for edge habitat, or the interface between covered and open habitat (Šálek et al. 2014, Takahata et al. 2014, Pywell et al. 2015), when hunting. Mule deer inhabiting rural, mostly undisturbed areas of

northwestern Colorado select for shrubland and forested, as well as barren and herbaceous, habitats (Lendrum et al. 2012), while female mountain goats (*Oreamnos americanus*), a prey source for mountain lions in Alberta, Canada, prefer to forage along interfaces between covered escape habitat and open habitat with abundant food (Hamel and Côté 2007). Shrubland/chaparral offers nutritious browse for deer, though it also leaves them more exposed to predators (Pierce et al. 2010), while forested habitats provide microclimates (Parker and Gillingham 1990) as well as concealment from predation for prey species, particularly while traveling (Bowyer 1986). Alternatively, areas of heavy vegetative cover also provide concealment for predators; thus open habitat (i.e. grassland and/or barren habitat) becomes an attractive option as deer are able to forage but also alert to potential risks (Lendrum et al 2012).

Female lions showed similar patterns of shrubland/chaparral and ample tree cover within their 50-95% isopleths, but whereas over half of the area of the males' 25% isopleths contained open habitat surrounded by vegetative cover, the average female 25% isopleth was almost entirely comprised of cover (Table 2.6, Figure 2.7). Mountain lions are opportunistic hunters and heavy vegetation also provides concealment and forage for small prey young mountain lions can consume (Yovovich et al. 2020).

These core areas of use also coincided with moderate-highly suitable deer habitat (Figure 2.9). Fort Hunter Liggett represents a large, contiguous piece of wildlife habitat, supporting a large population of ungulate prey (primarily mule deer, wild boar, and, to a lesser degree, elk) for the mountain lion population inhabiting it, and the high degree of female-female (0-46%) and male-male (54-79%) overlap between mountain lion home ranges is likely explained by the amount of resources on the landscape. While a majority of large carnivore species are considered solitary (Sandell 1989), prior research investigating mountain lion spatial organization has found that when prey densities increase, lion populations increase and individuals may be more tolerant of conspecifics within their established home ranges (Elbroch et al. 2014, 2016, Logan 2019). For example, Pierce et al. (1999, 2000) observed that the distribution of deer killed by lions was identical between areas occupied by one lion and areas of overlapping use and concluded that lions “most likely were limited by prey availability, and not territoriality.”

Laundré et al. (2007) found that populations in southern Idaho and northwestern Utah increased exponentially in response to increases in mule deer abundance (and declined after the deer population declined with a four-year time lag).

Individuals and family groups typically avoid other lions temporally, however (Bailey 1981). The majority of mule deer in the Central Coast are resident herds (California Department of Fish and Wildlife 2015), and it is likely that lion home range overlap can also be explained by areas of year-round abundant hunting opportunities. It was not surprising, then, that female-female and male-male HR overlap on FHL occurred in areas of medium-higher quality deer habitat (Figure 2.8, Figure 2.9) with the two males avoiding each other temporally. For example, time stamps in the GPS data confirmed that between late morning to late evening, SM3 could often be found in core areas of his range, while SM4 was several kilometers away, and vice versa. This suggests that there was enough prey to decrease territoriality but not remove it completely.

Increasing distance from perennial water, particularly if the source was also in proximity to shrubland/chaparral habitat, had a strong negative influence on the probability of utilization for Central Coast mountain lions. Year-round water is relatively scarce on Fort Hunter Liggett, with rain averaging 30.63 cm annually (National Oceanic and Atmospheric Administration 2022). Riparian habitats offer plentiful vegetative cover, cooler temperatures, fresh water, and vegetational complexity. Mountain lions in other water-limited regions such as the Northern Great Plains selected for riparian areas when selecting at the home range level (Gigliotti et al. 2019), while several southern California studies have also found that mountain lions strongly select for riparian habitat (Burdett et al. 2010, Zeller et al. 2017, Riley et al. 2021). This is likely due to their utilization of dense vegetation which supports large herbivorous prey species while allowing for cover when hunting (Atwood et al. 2007), as well as thermoregulation. As temperatures increase due to climate change, riparian areas are more likely to maintain their structural connectivity and ecological role as movement corridors as well as become refuges for species with limited adaptive capabilities (Krosby et al. 2018). For these reasons, ecologists frequently advise that they be highly prioritized in conservation efforts,

particularly in the face of not only climate change but increasing anthropogenic development (Hilty and Merenlender 2004).

The impact of slope percentage was minimal compared to the previous variables. This was likely due to the fact that FHL simply does not feature many slope gradients > 38%. 75-95% isopleths for all lions showed activity along gentle to moderately-steep slopes, but 25-50% isopleths reflected activity in primarily gentle to intermediate slopes. Riley et al. (2021) and Yovovich et al. (2020) also reported that adult mountain lions in analogous habitats found in the Santa Monica and Santa Cruz Mountains, respectively, generally avoid steeper slopes (>38.4%). Dellinger et al.'s (2020) statewide study found that mountain lions at lower elevations select for increasing slopes, but that they begin to show avoidance once slopes become steeper. This is very likely due to prey access and increased hunting success rates as mule deer generally prefer gentle-moderate slopes when balancing the benefits of forage and risk of predation (Nicholson et al. 1997, Lendrum et al. 2012, Peterson et al. 2021).

Distance from tertiary roads also did not have a large negative impact on probability of use. Tertiary roads are not only more energy-efficient for traversing elevation changes, they provide an elevated vantage point from which mountain lions may be able to scan the habitat below them for prey while still moving. They also facilitate early-successional plant communities which deer take advantage of (Dellinger et al. 2020). While these still apply on FHL, and tertiary roads feature heavily throughout the Base in part due to anthropogenic use, they are not as important of a resource within lion home ranges being that the elevation gradient is relatively minimal. While my results support previous research conclusions on habitat use and resource utilization, I again acknowledge the small sample size of study animals and urge future research in the Central Coast to continue prioritizing collaring more mountain lions in order to make stronger conclusions.

Central Coast Mountain Lion Population and Recommendations for Conservation

The within-home range resource selection results of the mountain lions on FHL were consistent with the majority of mountain lion studies in western North America within

the last several decades, with the main difference being that FHL represents a large amount of contiguous wildlife habitat with minimal anthropogenic interference. Regardless, it is a good representation of the largely unfragmented suitable habitat throughout the Central Coast. FHL is owned by the Department of Defense, however, and as such faces a much smaller risk of future fragmentation or habitat destruction than the rest of this area of California.

The counties comprising the Central Coast encompass 42,992 km² (Table 2.11). Of this, 41% is suitable mountain lion habitat (Table 2.12), of which area 41.7% is currently protected. Though Dellinger et al. (2020) and my results for area of overall suitable and protected habitat were very similar (their analyses found 16,355 km² and 6,780 km², respectively while mine found 17,685 and 7,367, respectively; Table 2.10), discrepancies between Dellinger et al.'s (2020) results and mine can likely be explained by slight differences in definitions of covariates and/or differences in models. Of this protected habitat, 4,989.94 km² is publicly owned, while conservation easements comprise 2,038.50 km².

As the amount of overall suitable habitat within the Central Coast exceeds the estimated amount of overall suitable habitat necessary to mitigate negative effects of low genetic diversity and support a sufficient breeding population (i.e. 14,591 km²; Table 2.12; Dellinger et al. 2020), my results demonstrate that the majority of this habitat is suitable for mountain lions, and thus many other species. Further, the amount of available protected suitable habitat very nearly meets the minimum threshold to maintain intermediate levels of both observed heterozygosity and allelic richness as well as an effective population > 50 (i.e. 7,923 km²; Gustafson et al. 2019, Dellinger et al. 2020). Wide-ranging species typically have resource requirements which cause them to move beyond these protected boundaries, though. If the surrounding areas are not also protected from eventual high levels of human development, the population may still be at risk. While the intuitive solution to this would seem to be focusing more effort into the stewardship of working lands surrounding protected habitats (Kremen and Merenlender 2018), particular county zoning categories and stakeholder ownership status can complicate such efforts. For example, land zoned for agriculture surrounds the majority

of protected habitat in the Central Coast (Figure 2.16). Much of this is under private ownership and as the human population continues to expand, it is at the greatest risk of habitat conversion.

It is not enough for habitat to be protected, however, if there is not only limited connectivity to other viable habitat but also inadequate area to support the life histories of the wildlife inhabiting it (Gigliotti et al. 2022). Monterey, San Luis Obispo, and Santa Barbara Counties each currently encompass the largest amount of suitable habitat for mountain lions in the Central Coast, with 25-28% each (Table 2.12), while San Benito has 12%, and northern Ventura encompasses 4%. Several networks of highways fragment this habitat, with U.S. Highway 101 dividing it into what may be viewed as three main areas of conservation focus.

The largest area of high-quality habitat spans from northern Monterey County to just south of State Route 46 in San Luis Obispo and includes the Sierra de Salinas and Santa Lucia Ranges (Figure 2.14). Despite already being heavily conserved through public land status (e.g. state and federal lands) and conservation easements (Figure 2.15), it is surrounded on almost all sides by land zoned for residential and agriculture (Figure 2.16). The cities of Salinas and Monterey are situated at the northern end of the habitat, the cities of Paso Robles and San Luis Obispo are to the south, while the Salinas Valley is at the bases of both mountain ranges to the east. Thorne et al. (2006) identified potential linkages between this habitat and that within the Santa Cruz, Gabilan, and Diablo Mountain Ranges at the northern end of this habitat as well as the southern end near Paso Robles, and the Transverse Ranges. All potential linkages are still minimally developed despite Highway 101 bisecting them. Wilson et al. (2020) predicted that the most dramatic increases in urban and perennial cropland growth are expected to take place in Monterey and San Luis Obispo Counties, however, with developed land projected to increase by 21.6% and 28-28.5%, respectively, by 2100. Given the orchards and vineyards already heavily established, expected increase in developed land, and the amount of agricultural zoning already in place nearby to towns and markets, these linkages are likely most at risk for loss due to development. As much of the land within proximity to these is privately owned ranches, priority should be given to obtaining

conservation easements on them. Several privately owned ranches (e.g. Eagle and Santa Margarita Ranches) near Templeton and Atascadero in San Luis Obispo County, for example, would contribute very valuable connectivity for wildlife in that region.

Secondly, habitat along the Transverse Ranges in eastern Santa Barbara County and northern Ventura County is largely conserved as public land (e.g. Los Padres National Forest; Figure 2.15); however connectivity between this habitat and that of western Santa Barbara is significantly hindered by human development. While the majority of the habitat in the aforementioned counties is conserved, it is isolated from the habitat to the west by Highway 101. This western habitat is also fragmented by highly trafficked state routes associated with Santa Maria, Lompoc, and Santa Barbara proper such as SRs 1, 135, 246, and 154. Additionally, this habitat is almost entirely zoned for agriculture and residential (Figure 2.16). Historically, most of the county zoned for agriculture has been used as rangelands, but low-medium density housing and agriculture in the form of row crops and vineyards have been increasing since 1940 (Syphard et al. 2018, Shapero et al. 2022). Sprawling development is the lead driver of land conversion and fire risk as it increases habitat fragmentation and removal (Syphard et al. 2018), increases road densities which are often the source of wildfires (Syphard and Keeley 2015), and increases the spread of non-native species (Bar-Massada et al. 2014). Impacts on biodiversity and ecological resilience may be minimized, however, through private land acquisition and clustering future development in areas of lower fire risk and biodiversity (Syphard et al. 2016). Further, mitigation measures to reduce mortality for wildlife attempting to travel between habitats on the eastern and western sides of Highway 101 would likely improve connectivity for populations impacted by fragmentation.

Finally, high quality habitat can be found throughout both the Gabilan and Diablo Ranges in San Benito County. The vast majority of the county is zoned for agriculture supported by limited groundwater basins, and what is not actively being used for privately owned crops and/or grazing is undeveloped natural land (Wilson et al. 2020). San Juan Bautista and Hollister to the north represent the largest sources of human development (Figure 2.14) and as the population continues to increase, development is likely to radiate from both towns. Recommendations for conservation include obtaining conservation

easements on privately owned farmland, particularly those on either side of Highway 101 and SR 156 so as to preserve the already tenuous linkage between the Gabilan, Diablo, and Santa Cruz Mountains (Diamond et al. 2022). Infilling low-medium density urban areas so as to be higher density and/or targeting lower quality wildlife habitat for development could also serve to direct pressure away from undeveloped areas throughout the rest of the county.

The US-101 corridor is also predicted to continue to experience pressure from urbanization into the future (California Department of Finance 2020); however, wildlife crossing structures are an extremely effective tool to maintain connectivity for wildlife (Thorne and Huber 2011, Zeller et al. 2017, Dellinger et al. 2020). Crossing structures are particularly effective when in proximity to vegetative cover and riparian corridors (Clevenger and Waltho 2000, Riley et al. 2006, Thorne et al. 2006, Craveiro et al. 2019, Jensen et al. 2022) with suitable habitat present on each side (Ng et al. 2004). Thorne and Huber (2011) identified areas of connectivity via undercrossings along Cuesta Grade where the Santa Lucia Range meets Highway 101 in San Luis Obispo County, while Jensen et al. (2022) also detected high levels of species diversity utilizing several others along the highway throughout the county. Protection of the habitat within the Cuesta Grade area would also maintain connectivity and reduce fragmentation between large areas of protected land in Monterey and San Luis Obispo Counties. Similar to protected habitat, much of the privately-owned land within and on either side of the US-101 corridor is zoned for agriculture (Figure 2.16), and conservation of habitat along this corridor should be prioritized. Additionally, conserving habitat in northern San Benito County would aid in maintaining connectivity between the Santa Cruz Mountain and Central Coast populations primarily on the eastern side of US-101.

Management Implications

As the human population in the California Central Coast continues to rapidly expand and develop, consideration from wildlife managers should not only be given to protecting and maintaining wildlife populations, but fostering relationships with large-scale private landowners (e.g. timber companies, ranching operations; Dellinger et al. 2020), other

government agencies at the state and federal level, and conservation entities. This serves to keep all key players in coordination and engaged, while increasing potential to make conservation beneficial for all involved.

Conservation of wild felids and other large carnivore populations is more likely to succeed when wildlife managers incorporate genomic data and connectivity into conservation efforts (Gustafson et al. 2022). Other fragmented large cat populations have benefited significantly from genetic rescue, including natural occurrences such as a single male mountain lion, M86, emigrating from the Eastern Peninsular population to the adjacent, isolated population in the Santa Ana Mountains and introducing novel genes in Southern California (Gustafson et al. 2017), as well as human intervention in the form of translocation such as that seen with African lions (*Panthera leo*) in South Africa (Miller et al. 2020) and Florida panthers (*Felis concolor coryi*; Whitely et al. 2015). While these examples illustrate successes for imperiled populations, it is critical that wildlife managers also view these as cautionary tales. Isolated mountain lion populations such as those in Southern California and the Santa Cruz Mountains illustrate the risks to the species as a whole if currently contiguous suitable habitat is similarly destroyed and/or fragmented. Conservation of suitable habitat, particularly corridors, will allow currently thriving populations not only to persist, but increase the likelihood of adjacent populations to recover. As apex predators are essential to the ecosystems in which they inhabit, ensuring their survival also ensures the persistence of many other species and ecosystems (Thorne et al. 2006, Ripple et al. 2014, Dellinger et al. 2020).

Tables & Figures

Table 2.1. Summary of location data for seven GPS collared mountain lions in the California Central Coast from 2018-2020, by animal ID, location, and dates of data collection

Animal ID	Sex	Start Date	End Date	Data Days	Number of Used Locations (at cut-off 7/11/2020)	Comment
SM3	Male	12/8/2018	7/11/2020	581	4,204	
SM4	Male	1/10/2019	7/11/2020	548	4,159	
SF1	Female	12/3/2018	7/11/2020	586	4,287	
SF2	Female	1/7/2019	6/13/2020	523	3,655	Animal died
SF3	Female	1/27/2019	1/16/2020	355	2,632	Collar failed
SF4	Female	2/21/2019	7/11/2020	519	3,833	
SF5	Female	4/9/2019	11/5/2019	198	1,647	Collar failed

Table 2.2. Predictor variables and sources used in resource selection analyses for mountain lions in the Central Coast of California (2018-2020).

Variable	Units	Source/Derivation	Year
Distance to tree cover	m	Aggregated tree classes from CalVeg	2015
Distance to oak woodland	m	Aggregated oak woodland classes from CalVeg	2015
Distance to shrub/chaparral cover	m	Aggregated shrubland and chaparral classes from CalVeg	2015
Distance to grassland/barren habitat	m	Aggregated grassland and barren classes from CalVeg	2015
Suitability of deer habitat	index	California Department of Fish and Wildlife	2017
Slope	%	ArcMap10.8.1 (Surface toolbox, Slope tool)	
Distance to perennial water	m	U.S. Geological Survey	2017
Distance to tertiary road	m	U.S. Geological Survey	2017

Table 2.3. Home range size (km²) of GPS collared mountain lions, by sex and HR delineation method, in the Central Coast of California from 2018-2020.

	Home Range Area (km²)		
Animal	95% MCP	95% KDI	95% a-LoCoH
SF1	168.02	186.91	189.51
SF2	233.25	301.06	307.04
SF3	219.35	211.03	212.64
SF4	122.83	132.21	133.80
SF5	191.88	206.95	208.30
Female: Mean; SE	187.07 ± 19.61	207.63 ± 27.26	210.26 ± 27.98
SM3	1291.79	1363.80	1395.01
SM4	1024.71	950.93	975.08
Male: Mean; SE	1158.25 ± 133.54	1157.37 ± 206.44	1185.05 ± 209.97
Overall: Mean; SE	187.07 ± 181.97	478.98 ± 181.83	488.77 ± 172.53

Table 2.4. Comparison of area (km²) and proportion (%) of habitat types within home ranges of GPS collared mountain lions in the Central Coast of California from 2018-2020 calculated using 95% MCP and 95% a-LoCoH.

Area (km²) and Proportion (%) of Habitat Type within Mountain Lion HRs by Method								
	95% MCP				95% a-LoCoH			
Animal	Shrubland/ Chaparral	Tree- Cover	Grassland/ Barren	Other	Shrubland/ Chaparral	Tree- Cover	Grassland/ Barren	Other
SF1	28.5; 17.0%	74.76; 45.7%	64.68; 38.5%	0.004; 0.002 %	37.47; 19.8%	83.05; 43.8%	68.93; 36.4%	0.004; 0.002 %
SF2	131.14; 56.2%	81.21; 34.8%	20.16; 8.6%	0; 0%	162.17; 52.8%	115.94; 37.8%	27.27; 8.9%	0.04; 0.01%
SF3	22.71; 10.1%	81.12; 37.0%	113.49; 51.7%	1.77; 0.8%	25.00; 11.8%	81.46; 38.3%	105.33; 49.5%	0.59; 0.28%
SF4	58.08; 47.3%	40.54; 33.0%	24.10; 19.6%	0.09; 0.07%	60.52; 45.2%	42.75; 32%	30.43; 22.7%	0.09; 0.07%
SF5	102.75; 53.5%	72.21; 37.6%	16.80; 0.09%	0.02; 0.01%	108.69; 52.2%	79.44; 23.7%	20.04; 9.8%	0.02; 0.01%
Female Mean	68.64 (SE = 21.10); 36.8%	69.97 (SE = 7.57); 37.6%	47.85 (SE = 18.56); 23.7%	0.471 (SE = 0.43); 0.22%	78.77 (SE = 25.27); 36.4%	80.53 (SE = 11.60); 35.1%	50.4 (SE = 16.14); 25.5%	0.15 (SE = 0.11); 0.07%
SM3	567.14; 43.9%	446.21; 34.5%	274.24; 21.2%	2.94; 0.23%	616.18; 44.2%	477.58; 34.2%	296.60; 21.3%	3.32; 0.24%
SM4	343.70; 33.5%	344.50; 33.6%	327.61; 32.0%	8.53; 0.83%	291.62; 29.9%	346.25; 35.5%	328.58; 33.7%	5.83; 0.60%
Male Mean	455.42 (SE = 111.72); 38.7%	395.36 (SE = 50.86); 38.7%	300.93 (SE = 26.69); 26.6%	5.74 (SE = 7.80); 0.53%	453.9 (SE = 162.28); 37.1%	411.92 (SE = 65.67); 34.9%	27.50 (SE = 6.2); 27.5%	4.58 (SE = 1.26); 0.42%
Overall Mean	179.15 (SE = 76.78); 37.4%	162.94 (SE = 61.25); 36.6%	120.15 (SE = 48.75); 24.5%	2.23 (SE = 1.35); 0.32%	185.95 (SE = 79.65); 36.6%	175.21 (SE = 63.28); 35.0%	125.31 (SE = 49.75); 26.0%	1.41 (SE = 0.86); 0.17%

Table 2.5. Comparison of area (km²) encompassed in varying utilization intensities for GPS collared mountain lions in the Central Coast of California from 2018-2020 calculated using parametric kernel density isopleth and a-LoCoH methods.

Areas (km²) of Kernel Density Isopleths by Method								
	Parametric KDI				a-LoCoH			
Animal	95%	75%	50%	25%	95%	75%	50%	25%
SF1	186.91	96.97	50.80	20.91	189.51	99.85	46.41	13.60
SF2	301.06	172.65	98.28	43.92	307.04	174.75	95.37	34.49
SF3	211.03	99.69	48.31	16.72	212.64	100.81	41.09	12.76
SF4	132.21	68.79	36.30	13.67	133.80	65.58	33.89	10.61
SF5	206.95	107.90	55.36	22.41	208.30	108.19	53.04	19.42
Female: Mean; SE	207.63 ± 19.61	109.20 ± 17.18	57.81 ± 10.60	23.53 ± 5.33	210.26 ± 27.98	109.84 ± 17.83	53.96 ± 10.82	18.18 ± 4.33
SM3	1363.80	704.08	372.40	152.61	1395.01	714.69	343.73	115.18
SM4	950.93	465.63	211.08	74.67	975.08	440.73	196.80	63.67
Male: Mean; SE	1157.37 ± 206.44	584.86 ± 119.23	291.74 ± 80.66	113.64 ± 38.97	1185.05 ± 209.97	577.71 ± 136.98	270.27 ± 73.47	89.43 ± 25.76
Overall: Mean; SE	478.98 ± 181.83	245.10 ± 95.27	124.65 ± 46.68	49.27 ± 19.03	488.77 ± 172.53	243.51 ± 92.14	115.76 ± 43.64	38.53 ± 14.60

Table 2.6. Comparison of area (km²) of different habitat types encompassed in varying utilization intensities for GPS collared mountain lions in the Central Coast of California from 2018-2020 calculated using parametric kernel density isopleths and a-LoCoH methods.

Areas (km ²) of Habitat Type Within Core Use Isopleths by Method									
	Parametric Kernel Density Isopleths					Adaptive-Local Convex Hulls			
	Utilization Intensity	Shrubland/ Chaparral	Tree-Cover	Other	Grassland/ Barren	Shrubland/ Chaparral	Tree-Cover	Other	Grassland/Barren
Female Average	75%	43.14 ± 15.76; 37.9%	40.41 ± 6.78; 36.6%	0.03 ± 0.01%; ; 0.03%	25.42 ± 8.36; 25.3%	43.33 ± 15.75; 38.5%	40.80 ± 7.05; 36.7%	0.03 ± 0.01; 0.03%	26.04 ± 8.69; 25.7%
	50%	23.73 ± 9.06; 38.7%	21.74 ± 3.95; 37.8%	0.01 ± 0.004; 0.02%	12.09 ± 3.68; 22.6%	19.97 ± 9.49; 31.0%	20.10 ± 3.96; 37.4%	0.01 ± 0.004; 0.02%	10.86 ± 3.36; 23.0%
	25%	9.62 ± 3.47; 39.0%	9.79 ± 2.09; 41.8%	0.01 ± 0.004; 0.03%	4.00 ± 1.11; 18.8%	7.69 ± 2.79; 40.4%	7.64 ± 1.75; 35.3%	0.01 ± 0.004; 0.1%	2.78 ± 0.72; 17.3%
Male Average	75%	152.24 ± 72.50; 24.5%	232.23 ± 45.07; 38.0%	1.78 ± 0.53; 0.3%	198.03 ± 2.07; 35.3%	151.62 ± 80.36; 24.3%	225.99 ± 50.27; 39.3%	1.92 ± 0.41; 0.4%	197.61 ± 6.66; 36.0%
	50%	54.65 ± 37.03; 16.5%	118.82 ± 41.42; 39.9%	0.39 ± 0.08; 0.2%	117.51 ± 2.24; 43.4%	48.40 ± 34.15; 15.6%	106.82 ± 37.86; 38.6%	0.36 ± 0.04 ; 0.2%	114.18 ± 1.33; 45.5%
	25%	15.44 ± 7.67; 12.8%	35.31 ± 29.91; 25.0%	0.16 ± 0.03; 0.6%	54.95 ± 8.99; 51.8%	11.00 ± 4.37; 11.9%	31.35 ± 14.00; 33.0%	0.14 ± 0.01; 0.2%	46.84 ± 7.31; 54.6%
Overall Average	75%	74.32 ± 27.81; 34.1%	95.21 ± 37.01; 37.5%	0.52 ± 0.34; 0.1%	74.74 ± 35.36; 28.1%	74.27 ± 28.71; 34.4%	93.72 ± 36.20; 37.4%	0.57 ± 0.36; 0.1%	75.06 ± 32.24; 28.6%
	50%	32.57 ± 11.70; 32.3%	49.55 ± 20.22; 38.4%	0.11 ± 0.07; 0.1%	42.21 ± 19.61; 28.5%	28.09 ± 11.22; 26.6%	44.88 ± 18.21; 37.7%	0.11 ± 0.07; 0.01%	40.38 ± 19.20; 29.4%
	25%	11.28 ± 3.11; 31.5%	17.08 ± 8.17; 37.0%	0.05 ± 0.03; 0.1%	18.57 ± 9.64; 28.2%	8.63 ± 2.23; 32.2%	14.41 ± 5.47; 34.7%	0.04 ± 0.02; 0.1%	15.37 ± 8.30; 27.9%

Table 2.7. Percentage of area (km²) overlap between 95% a-LoCoH home ranges of seven GPS collared mountain lions in the Central Coast of California. This table should be read as row in relation to column (e.g. 14% of SF5's home range overlapped with SF1's, 18% of SM4's home range overlapped with SF1's, and so forth).

Proportion of Overlap Between 95% Home Ranges							
	SF1	SF2	SF3	SF4	SF5	SM3	SM4
SF1	1	0	0.46	0	0.15	1	0.92
SF2	0	1	0	0.007	0	0.81	0.16
SF3	0.42	0	1	0.13	0.13	1	1
SF4	0	0.015	0.2	1	0	0.84	0.85
SF5	0.14	0	0.13	0	1	0.99	0.69
SM3	0.14	0.17	0.15	0.08	0.15	1	0.54
SM4	0.18	0.03	0.22	0.12	0.15	0.79	1

Table 2.8. Change in Akaike's Information Criterion corrected for small sample sizes (ΔAIC_c) and number of parameters (K) of top three most parsimonious resource selection function models for quantifying habitat selection of seven GPS collared mountain lions from 2018-2020 at the within-home range scale (third-order resource selection; Johnson 1980).

Model	ΔAIC_c	K
Trees + trees ² + shrubs + slope + deer + water + tertiaryrd + (shrubs*water)	0.00	8
Trees + trees ² + shrubs + slope + deer + water + tertiaryrd + (tertiaryrd*deer)	34.39	8
Trees + trees ² + shrubs + deer + water + tertiaryrd + (tertiaryrd*deer)	104.04	7

Table 2.9. Coefficient estimates (β), standard errors, and p-values for the best fitting resource selection function model for quantifying the resource selection of seven GPS collared mountain lions in the Central Coast of California from 2018-2020 at the within-home range scale.

Variable	β	Std. Error	p-value
Distance to tree cover	-0.496	0.015	$<2 \times 10^{-16}$
Distance to tree cover ²	0.048	0.006	$<2 \times 10^{-16}$
Distance to shrub cover	-0.619	0.035	$<2 \times 10^{-16}$
Slope	-0.070	0.008	$<2 \times 10^{-16}$
Deer habitat suitability	0.360	0.010	$<2 \times 10^{-16}$
Distance to perennial water source	-0.583	0.050	$<2 \times 10^{-16}$
Distance to tertiary road	-0.037	0.009	$<3.07 \times 10^{-5}$
Interaction between distance to shrub cover and perennial water source	-0.780	0.072	$<2 \times 10^{-16}$

Table 2.10. Genetic diversity values (expected heterozygosity and allelic richness; Gustafson et al 2019) and overall/protected habitat values for the Central Coast mountain lion population in California (Dellinger et al. 2020).

Population	Observed heterozygosity	Allelic richness	Overall habitat (km ²)	Protected habitat (km ²)
Santa Cruz	0.41	2.62	5,042	1,818 (36%)
Central Coast	0.45	3.00	16,355 (33%)	6,780 (41%)
Santa Monica	0.41	2.63	2,688	1,129 (42%)
Santa Ana	0.34	2.27	2,054	1,081 (53%)
Transverse Range	0.40	2.75	3,759	2,976 (79%)

Table 2.11. Overall area (km²) of the California Central Coast and zoning categories within (Channel Islands excluded).

<i>Overall Area (km²)</i>						
	San Luis Obispo	Monterey	Santa Barbara	San Benito	Ventura	Total Area
Agriculture	10,338.30 (80.2%)	7,542.06 (58.7%)	6,660.37 (67.0%)	5,424.24 (97.4%)	36.54 (2.1%)	30,001.51; 69.8%
Open Space	1,307.19 (10.1%)	24.80 (0.2%)	2,523.46 (25.4%)	0 (0%)	1,707.44 (97.7%)	5,572.89; 13.0%
Public Facilities	197.96 (1.5%)	3,306.58 (25.7%)	3.37 (0.03%)	23.89 (0.4%)	0 (0%)	3,531.80; 8.2%
Other	481.51 (3.7%)	88.82 (0.7%)	559.70 (5.6%)	92.38 (1.7%)	0 (0%)	1,226.41; 2.9%
Residential	541.74 (4.2%)	431.13 (3.4%)	177.36 (1.8%)	20.79 (0.4%)	0.53 (0.03%)	1,171.55; 2.7%
Industrial	11.77 (0.09%)	53.72 (0.4%)	13.74 (13.8%)	6.09 (0.1%)	2.18 (0.1%)	87.50; 0.20%
Commercial	5.10 (0.04%)	14.35 (0.1%)	5.50 (0.06%)	1.71 (0.03%)	0.08 (0.005%)	26.74; 0.06%
Conservation (Monterey Co.)	0 (0%)	1,387.71 (10.8%)	0 (0%)	0 (0%)	0 (0%)	1,387.71; 3.2%%
Grand Totals	12,883.57	12,849.17	9,943.50	5,569.10	1,746.77	42,991.63

Table 2.12. Suitable mountain lion habitat within different zoning categories in the Central Coast (Channel Islands excluded).

<i>Suitable Habitat (km²)</i>						
	Monterey	San Luis Obispo	Santa Barbara	San Benito	Ventura	Total Area
Agriculture	2,652.64 (44.5%)	3,522.31 (74.0%)	3,164.00 (75.3%)	1,986.37 (97.0%)	12.46 (1.7%)	11,337.78; 64.11%
Open Space	12.19 (0.2%)	867.36 (18.2%)	923.29 (22.0%)	0 (0%)	701.41 (98.0%)	2,504.25; 14.16%
Public Facilities	1,908.23 (32.0%)	69.91 (1.5%)	5.31 (0.1%)	17.20 (0.8%)	0 (0%)	2,000.65; 11.31%
Other	50.82 (0.9%)	174.60 (3.7%)	61.82 (1.5%)	2.94 (0.1%)	0 (0%)	290.18; 1.64%
Residential	284.81 (4.8%)	125.79 (2.6%)	45.82 (1.1%)	39.92 (1.9%)	0.26 (0.04%)	496.60; 2.81%
Industrial	5.78 (0.1%)	1.28 (0.03%)	4.19 (0.1%)	1.60 (0.08%)	1.54 (0.2%)	14.39; 0.08%
Commercial	3.48 (0.06%)	0.40 (0.008%)	0.49 (0.01%)	0.11 (0.005%)	0.07 (0.01%)	4.55; 0.03%
Conservation (Monterey Co)	1,036.92 (17.4%)	0 (0%)	0 (0%)	0 (0%)	0 (0%)	1,036.92; 5.86%
Grand Totals	5,954.87	4,761.65	4,204.43	2,048.14	715.74	17,684.83

Table 2.13. Protected suitable mountain lion habitat within different zoning categories in the Central Coast (Channel Islands excluded).

<i>Protected Suitable Habitat (km²)</i>						
	Monterey	San Luis Obispo	Santa Barbara	Ventura	San Benito	Total Area
Agriculture	253.88 (10.2%)	981.92 (52.1%)	949.78 (51.1%)	0.16 (0.03%)	504.9 (96.8%)	2,690.64; 36.52%
Open Space	1.66 (0.1%)	824.62 (43.7%)	867.32 (46.6%)	600.82 (99.9%)	0 (0%)	2,294.42; 31.15%
Public Facilities	1,158.98 (46.4%)	1.32 (0.07%)	0.23 (0.01%)	0 (0%)	16.81 (3.2%)	1,177.34; 15.98%
Other	45.54 (1.8%)	75.6 (0.04%)	37.39 (2.01%)	0 (0%)	0 (0%)	158.53; 2.15%
Residential	2.02 (0.08%)	2.13 (11.3%)	4.73 (0.23%)	0 (0%)	0 (0%)	8.88; 0.12%
Industrial	0 (0%)	0 (0%)	0 (0%)	0 (0%)	0 (0%)	0; 0%
Commercial	0 (0%)	0.03 (0.002%)	0 (0%)	0 (0%)	0 (0%)	0.03; 0%
Conservation (Monterey Co)	1,036.92 (41.5%)	0 (05%)	0 (0%)	0 (0%)	0 (0%)	1,036.92; 14.08%
Grand Totals	2,499.00	1,885.62	1,859.45	600.98	521.71	7,366.76

Table 2.14. Unprotected suitable mountain lion habitat within different zoning categories in the Central Coast (Channel Islands excluded).

<i>Unprotected Suitable Habitat (km²)</i>						
	Monterey	San Luis Obispo	Santa Barbara	San Benito	Ventura	Total Area
Agriculture	2,398.76 (69.4%)	2,540.39 (88.3%)	2,214.22 (94.4%)	1,481.47 (97.2%)	12.3 (10.7%)	8,647.14; 83.83%
Open Space	10.53 (0.3%)	42.74 (1.5%)	55.97 (2.4%)	0 (0%)	100.59 (87.7%)	209.83; 2.03%
Public Facilities	749.25 (21.7%)	68.59 (2.4%)	5.08 (0.2%)	0.39 (0.03%)	0 (0%)	823.31; 7.98%
Other	5.28 (0.15%)	99.00 (3.4%)	24.43 (1.04%)	0 (0%)	0 (0%)	128.71; 1.25%
Residential	282.79 (8.2%)	123.66 (4.3%)	41.09 (1.8%)	39.92 (2.6%)	0.26 (0.23%)	487.72; 4.73%
Industrial	5.78 (0.2%)	1.28 (0.04%)	4.19 (0.2%)	1.6 (0.1%)	1.54 (1.3%)	14.39; 0.14%
Commercial	3.48 (0.1%)	0.37 (0.01%)	0.49 (0.02%)	0.11 (0.007%)	0.07 (0.06%)	4.52; 0.04%
Conservation (Monterey Co)	0 (0%)	0 (0%)	0 (0%)	0 (0%)	0 (0%)	0.00; 0%
Grand Totals	3,455.87	2,876.03	2,345.47	1,523.49	114.76	10,315.62

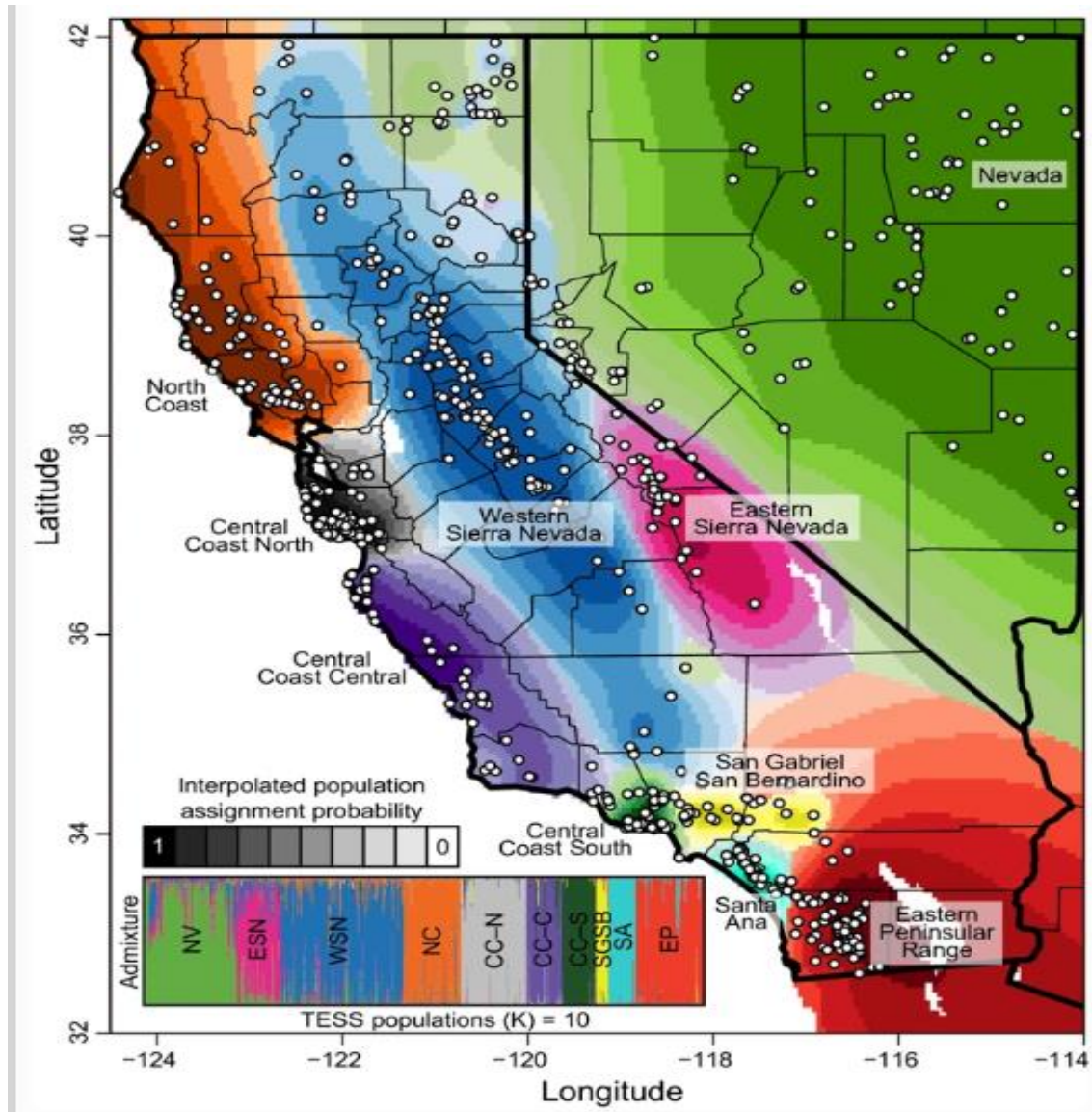


Figure 2.1 Map of the population genetic structure of mountain lions across California and Nevada from Gustafson et al. (2019). I defined the Central Coast according to the geographic extent of Gustafson et al.’s “Central Coast Central” population, including Monterey, San Benito, San Luis Obispo, Santa Barbara, and Ventura (north of SR118) Counties.

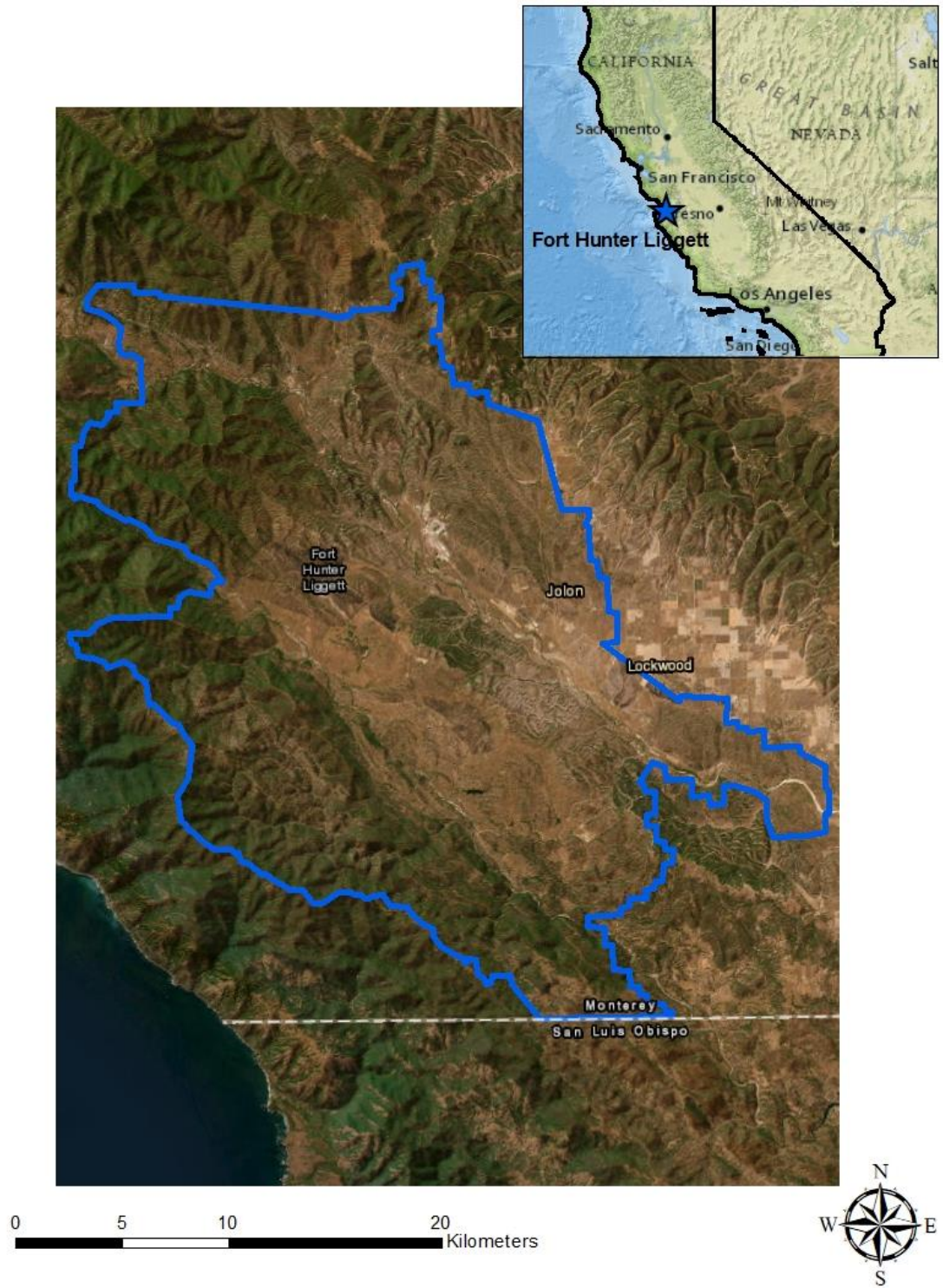


Figure 2.2 Map of Fort Hunter Liggett boundary.

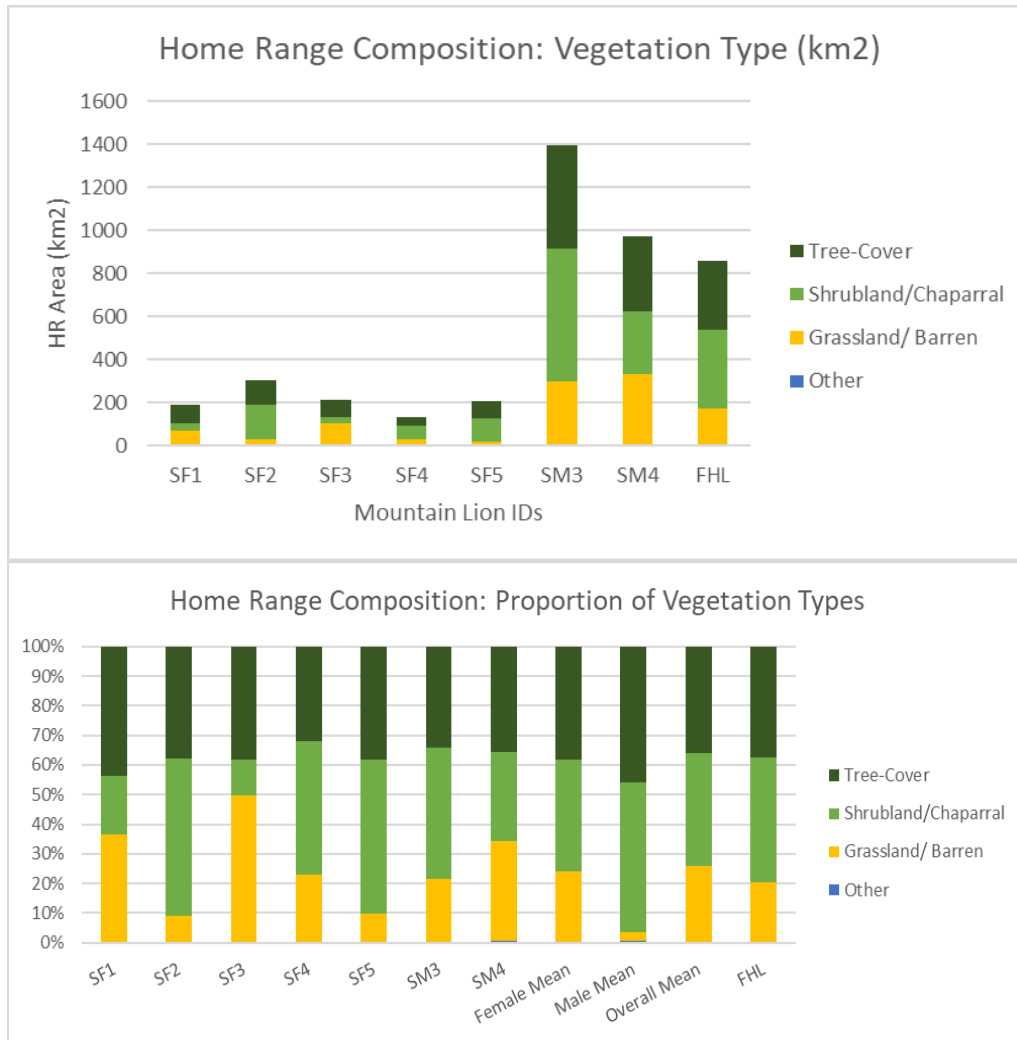


Figure 2.3 A Area (km²) **B** proportion of vegetation types found within the home ranges of seven GPS collared mountain lions within the Central Coast of California from 2018-2020 (USDA Forest Service Remote Sensing Lab 2015). All areas calculated using 95% a-LoCoH.

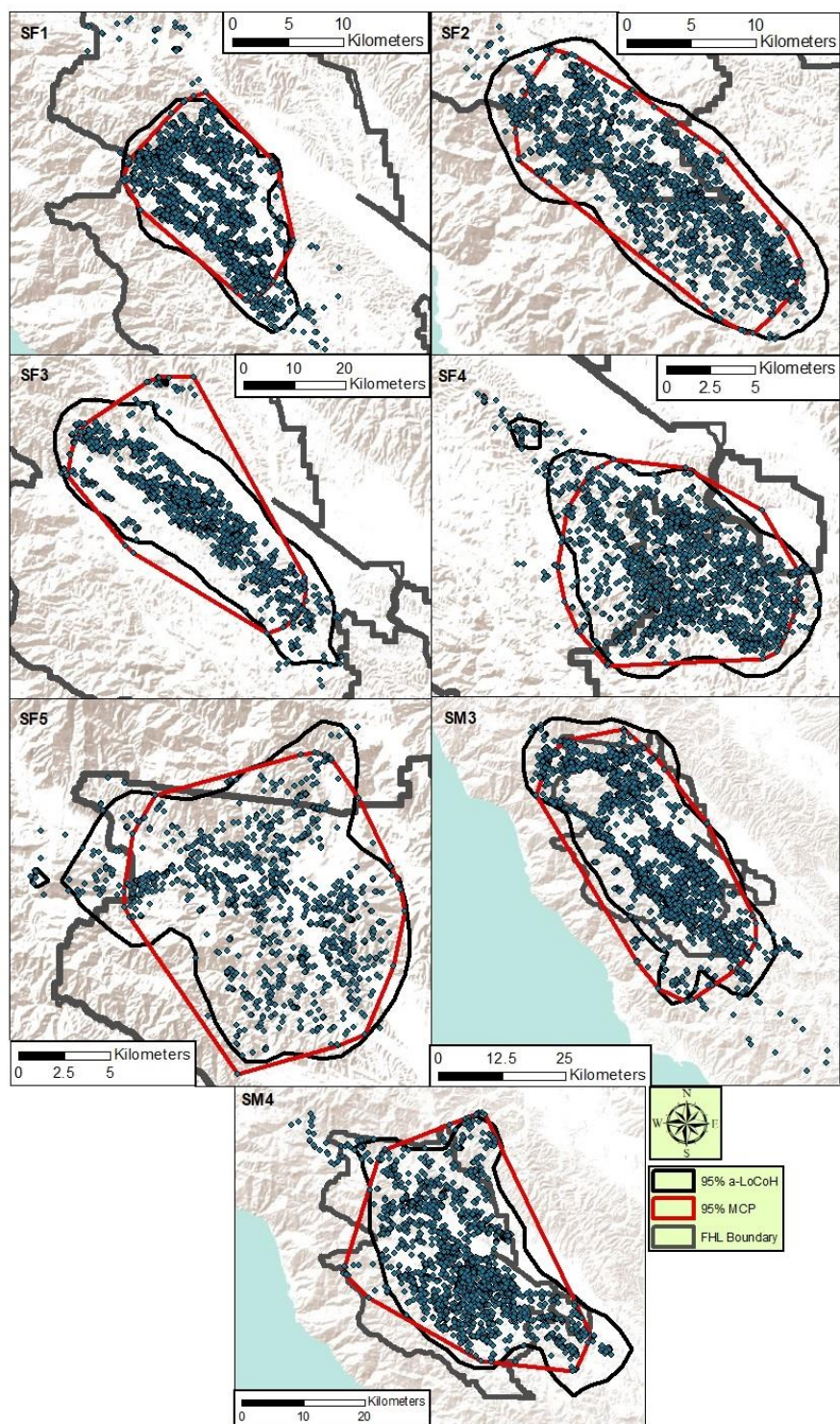


Figure 2.4 Comparison of 95% MCP (red) and 95% LOCOH (black) home range using GPS collar data from mountain lions collared in the California Central Coast from 2018-2020.

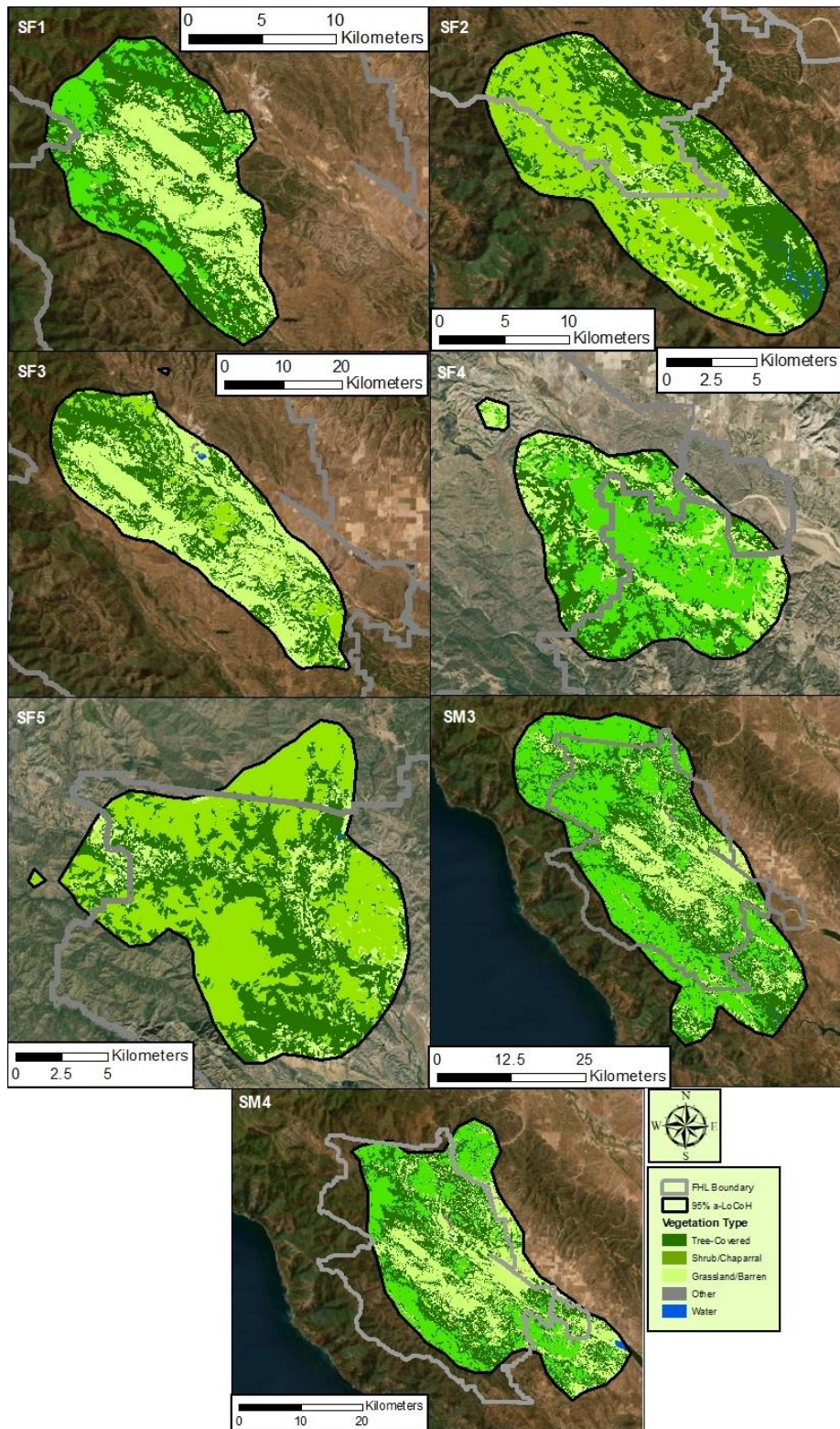


Figure 2.5 Vegetation composition of the 95% a-LoCoH home ranges of seven GPS collared mountain lions in the Central Coast of California from 2018-2020.

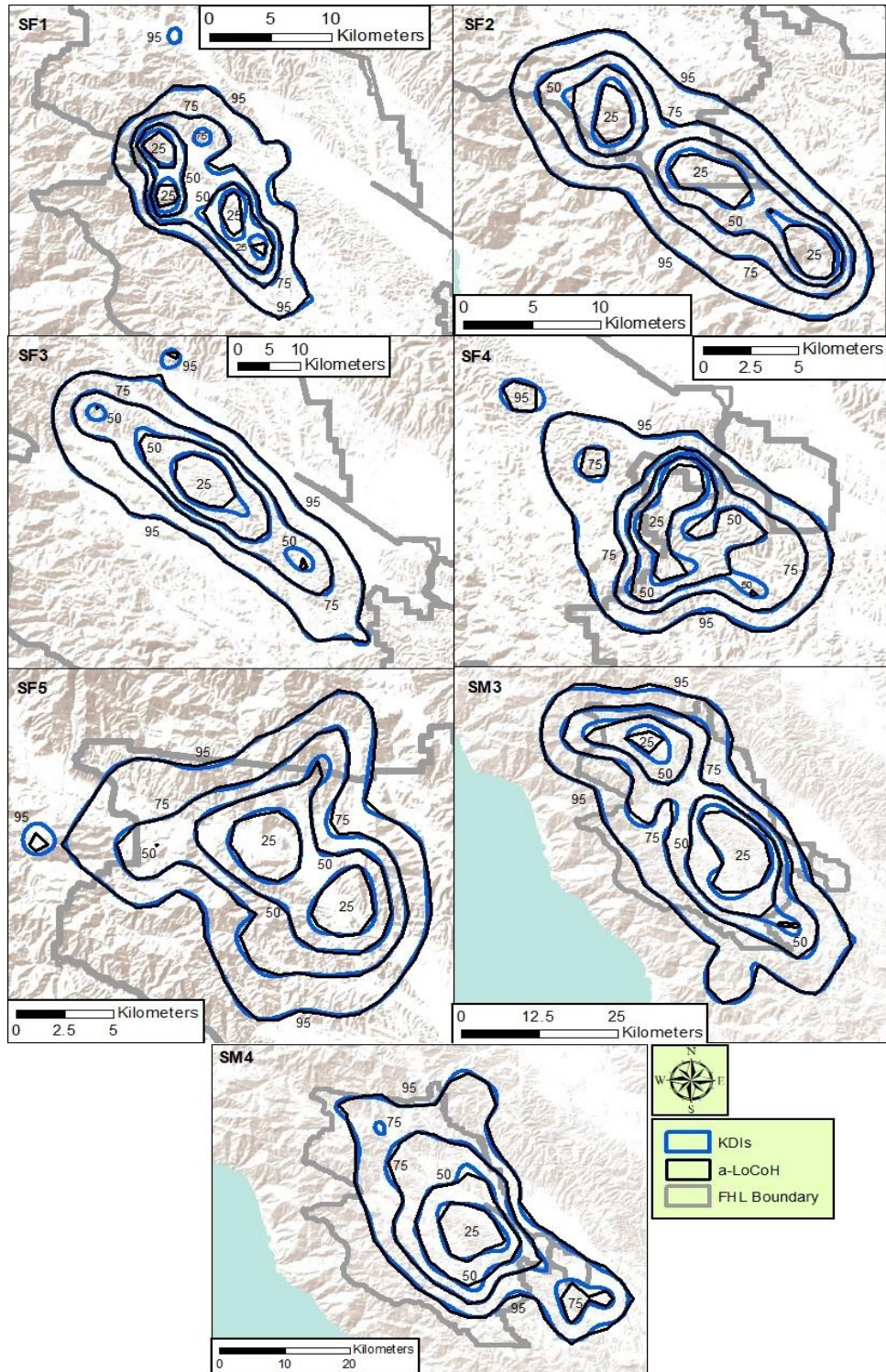


Figure 2.6. Comparison of kernel density isopleths (blue) and adaptive-local convex hulls (black) for identifying utilization distribution within home ranges using GPS collar data from seven mountain lions collared in the California Central Coast from 2018-2020.

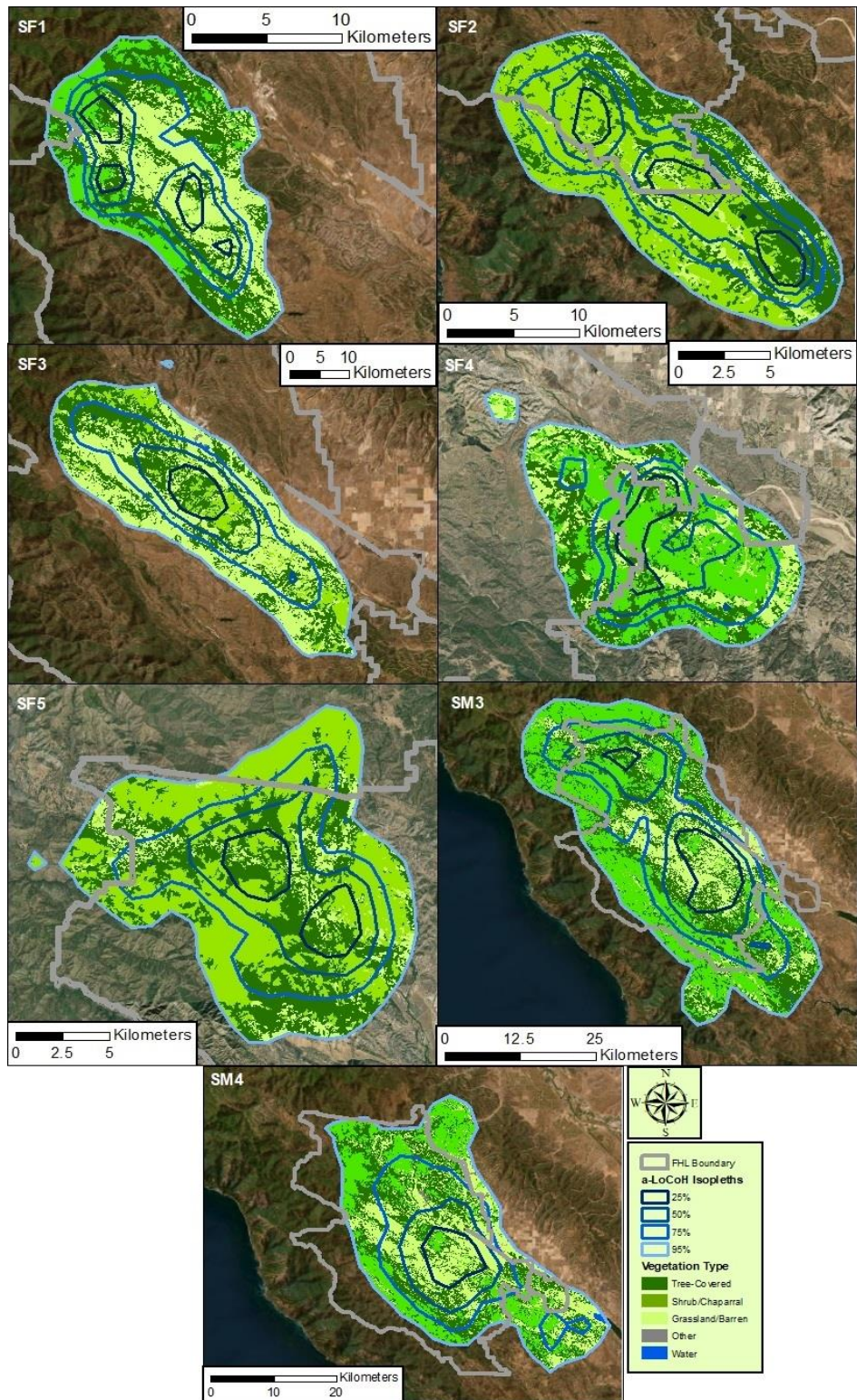


Figure 2.7 Vegetation composition maps overlaid with 25, 50, 75, 95% a-LoCoH isopleths of seven GPS collared mountain lions in the Central Coast of California from 2018-2020.

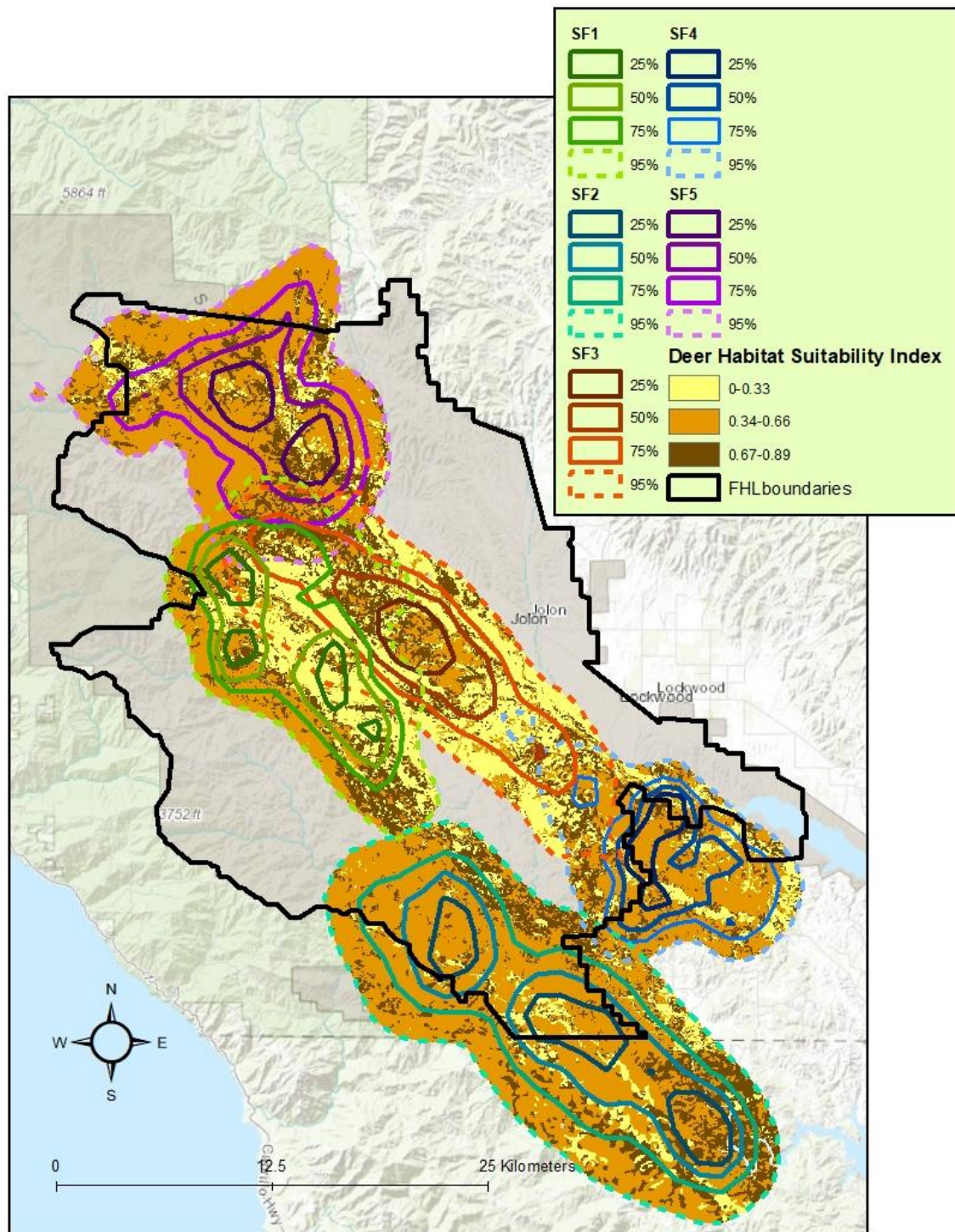


Figure 2.8 Characteristics of five female GPS collared mountain lions' home ranges, calculated using a-LoCoH, relative to deer habitat suitability.

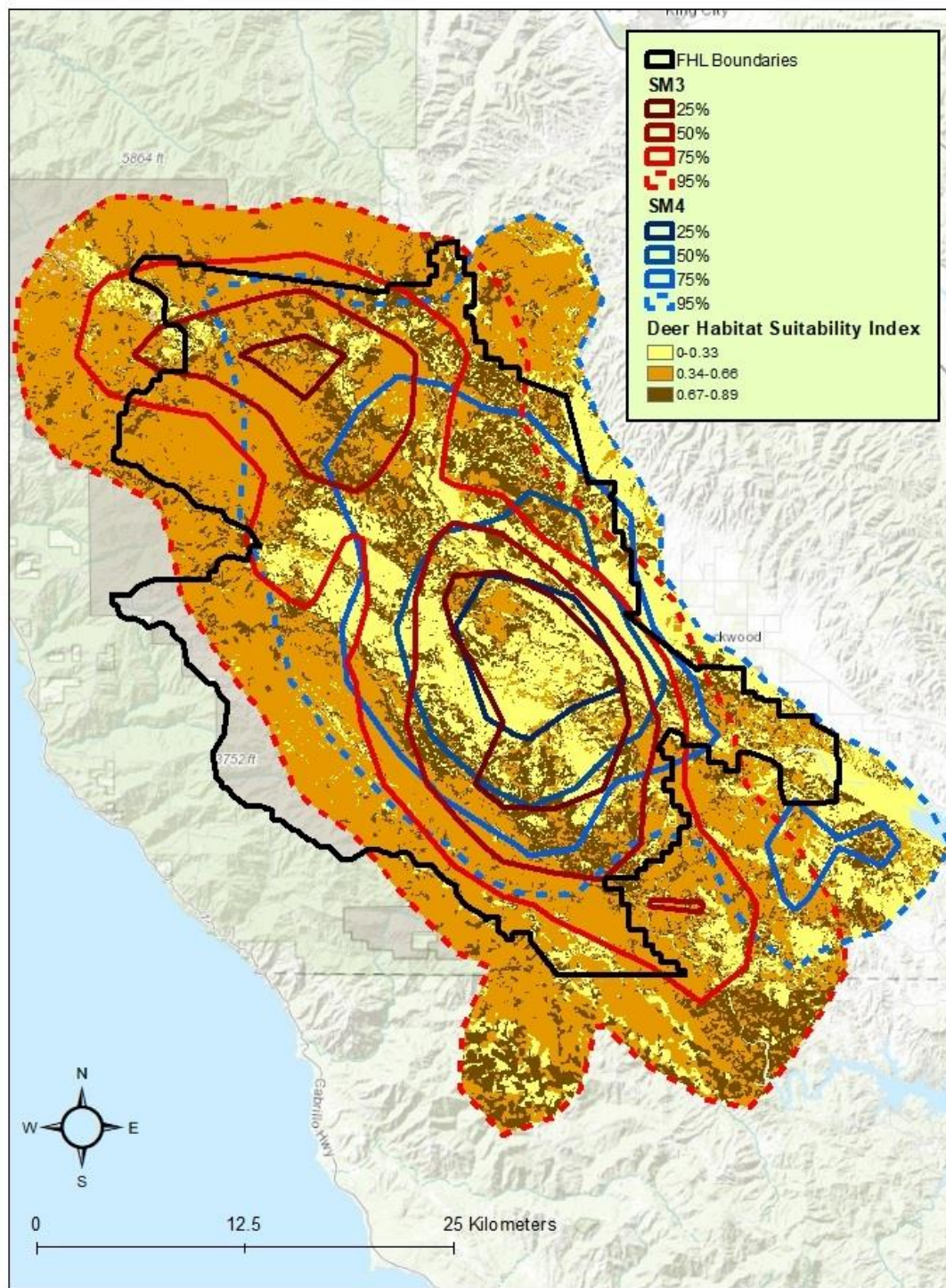


Figure 2.9 Characteristics of two male GPS collared mountain lions' home ranges, calculated using a-LoCoH, relative to deer habitat suitability.

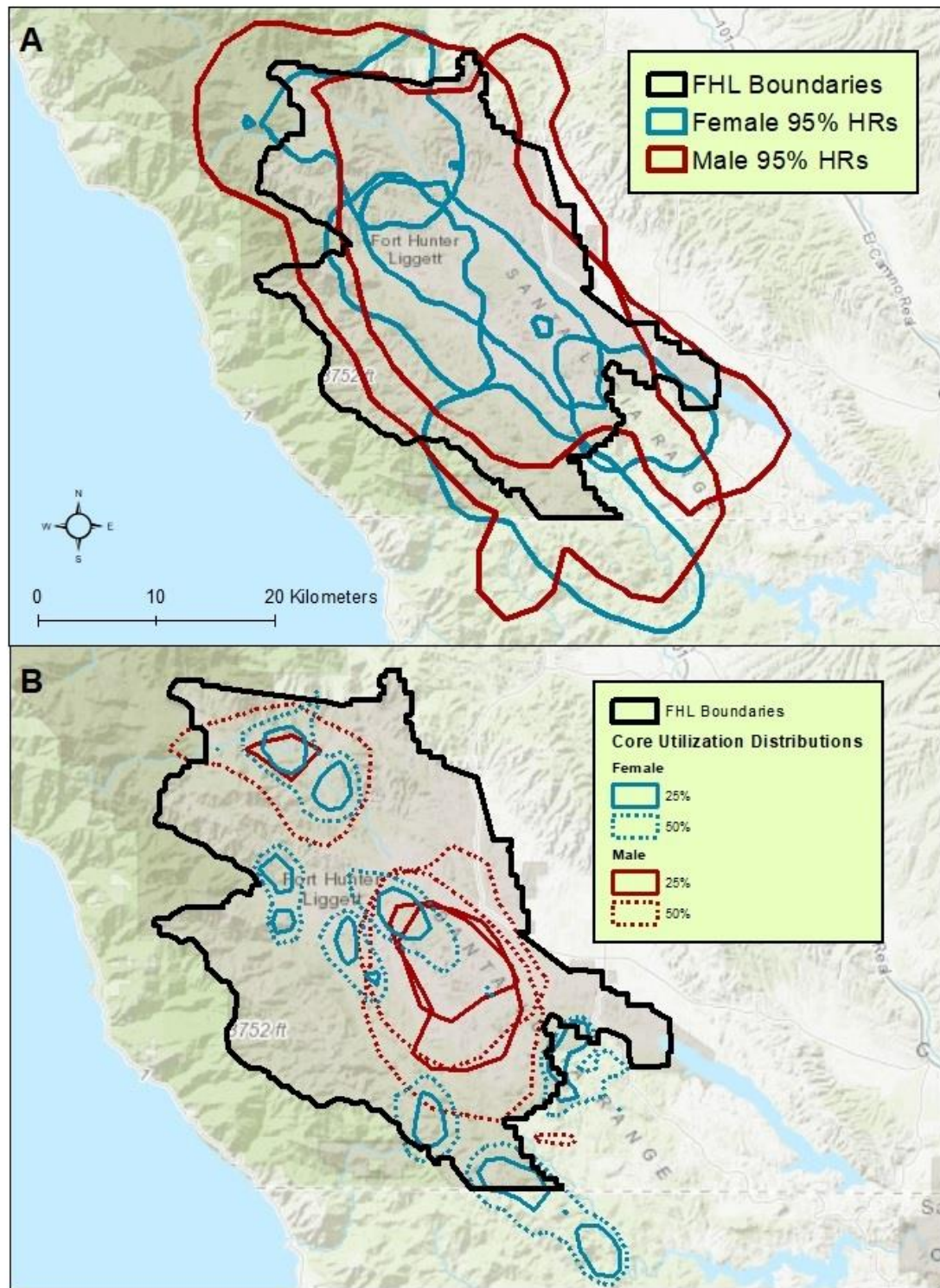


Figure 2.10 Spatial overlap of **A** home range outer boundary and **B** 25 and 50% a-LoCoH core area utilization distributions for seven GPS collared mountain lions in the Central Coast of California from 2018-2020.

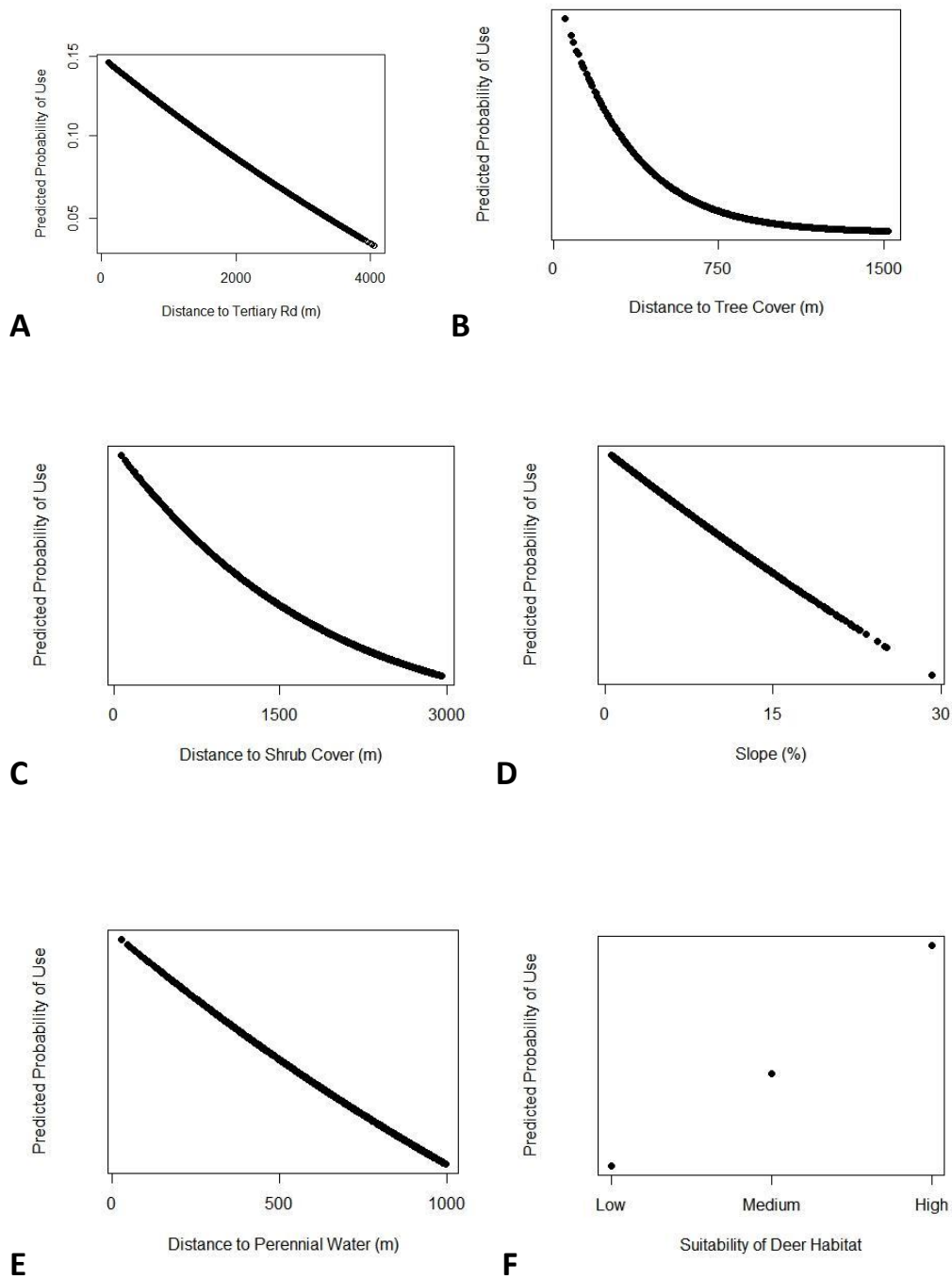


Figure 2.11. Relative probability of habitat use by seven GPS collared mountain lions on the Central Coast of California based on individual variables on the landscape from 2018-2020.

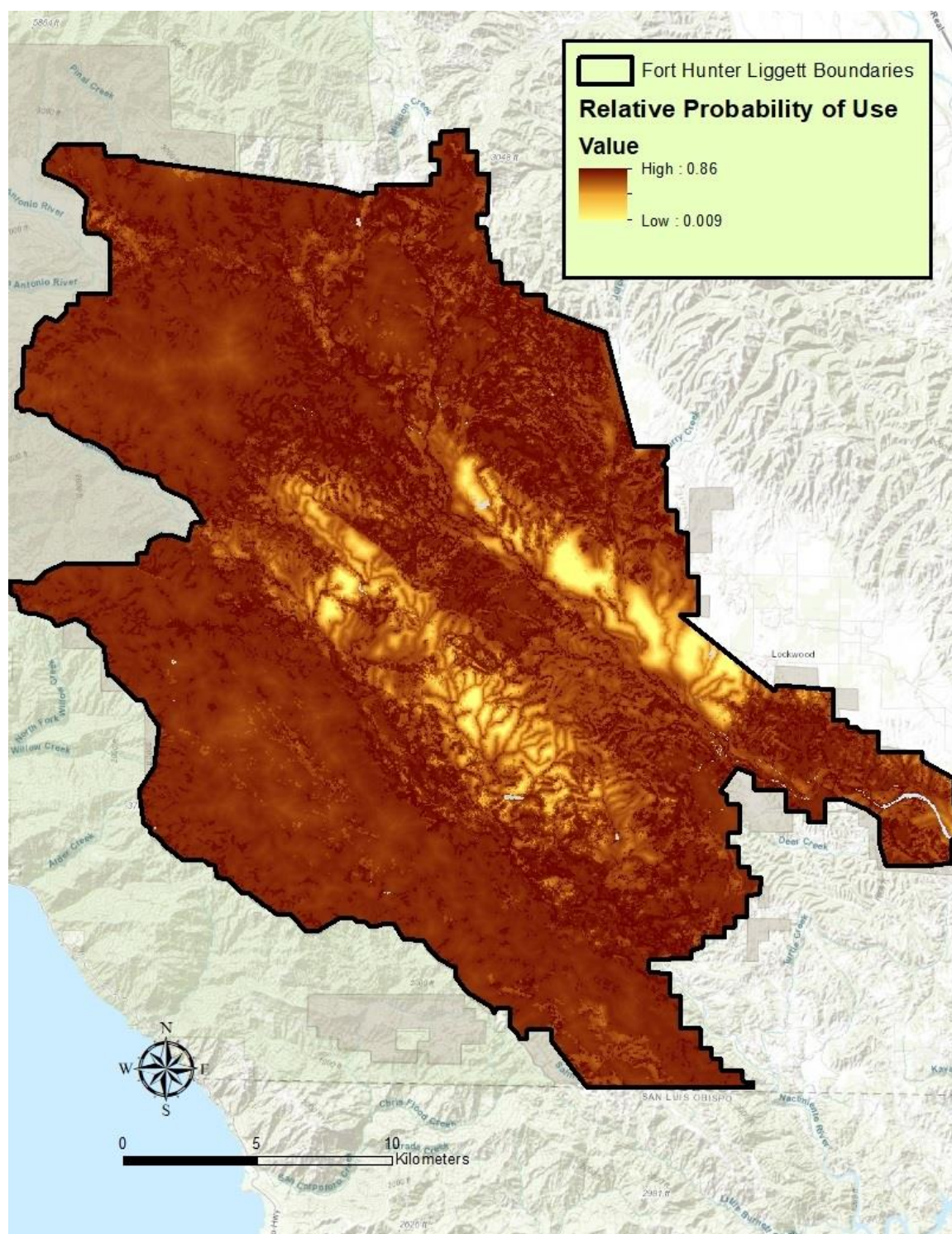


Figure 2.12 Predicted relative probability of use by mountain lions on Fort Hunter Liggett Army Base.

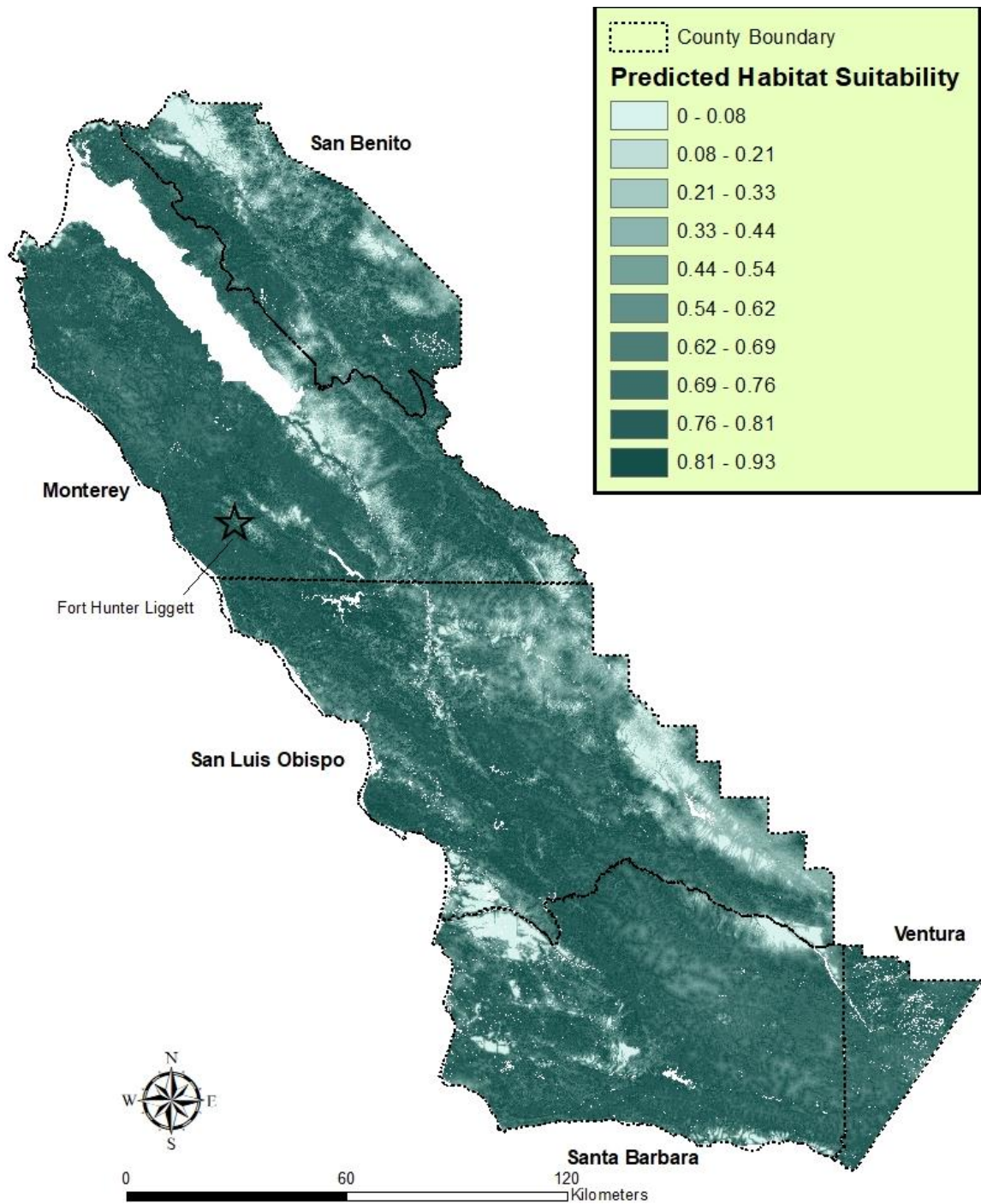


Figure 2.13 Predicted mountain lion habitat suitability in the Central Coast of California. Salinas Valley and Channel Islands were truncated from analyses.

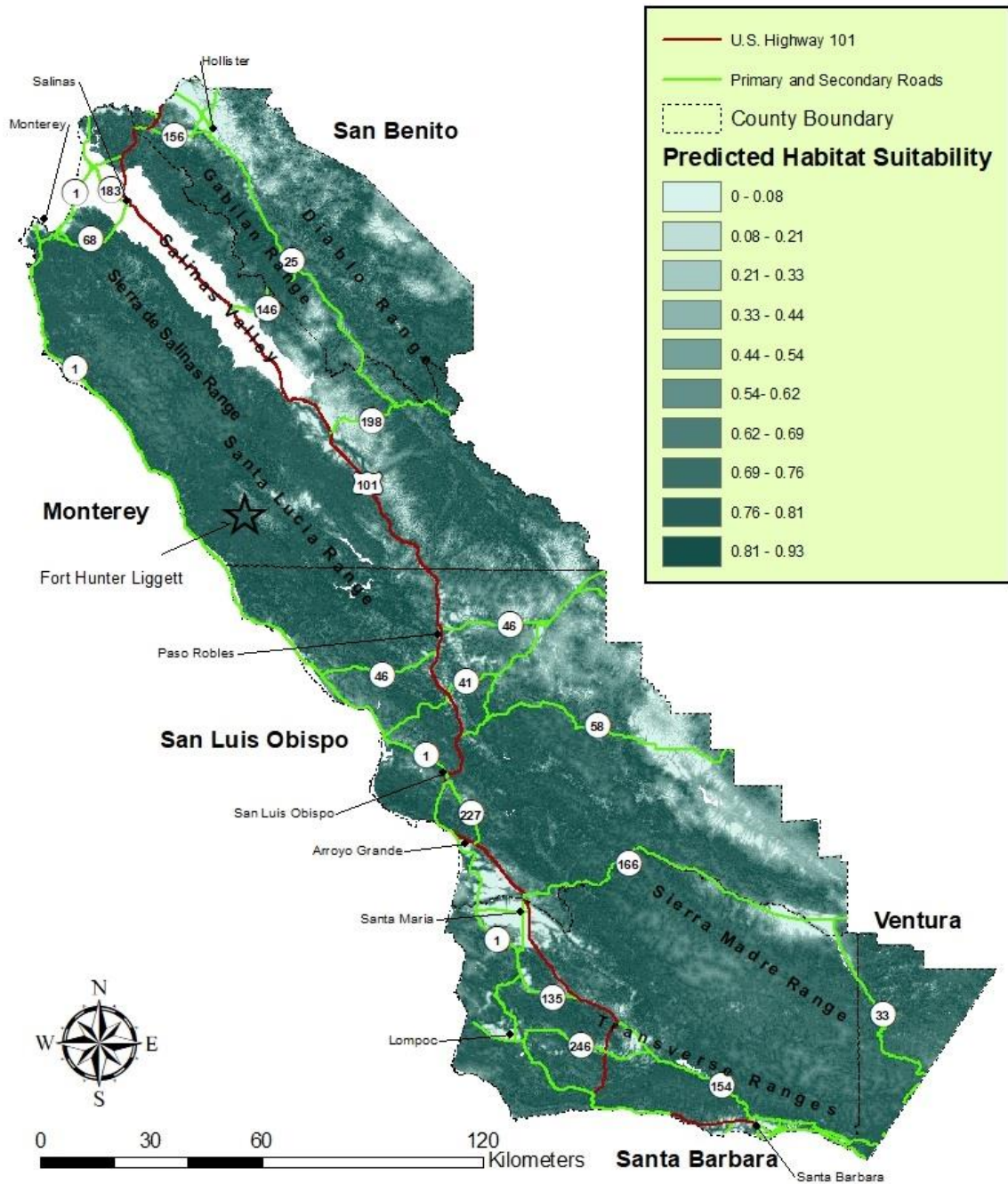


Figure 2.14. Predicted mountain lion habitat suitability in the Central Coast of California (excluding the Channel Islands and Salinas Valley) overlaid with causes of habitat fragmentation, including varying levels of human-developed land and primary and secondary roads (e.g. highways, freeways, etc.).

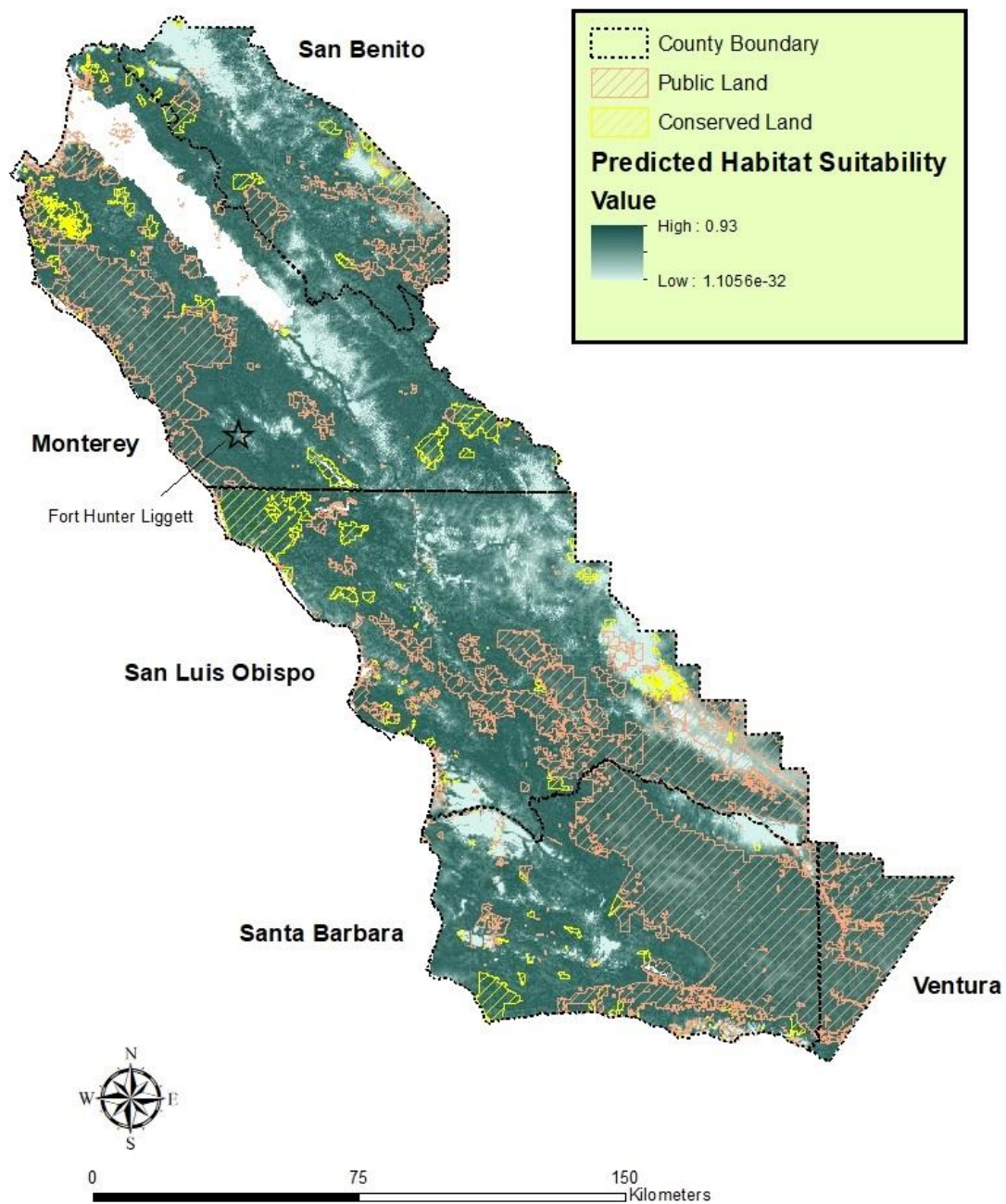


Figure 2.15 Conserved lands (i.e., protected by public land status or conservation easements) within the California Central Coast overlaid onto predicted mountain lion habitat suitability.

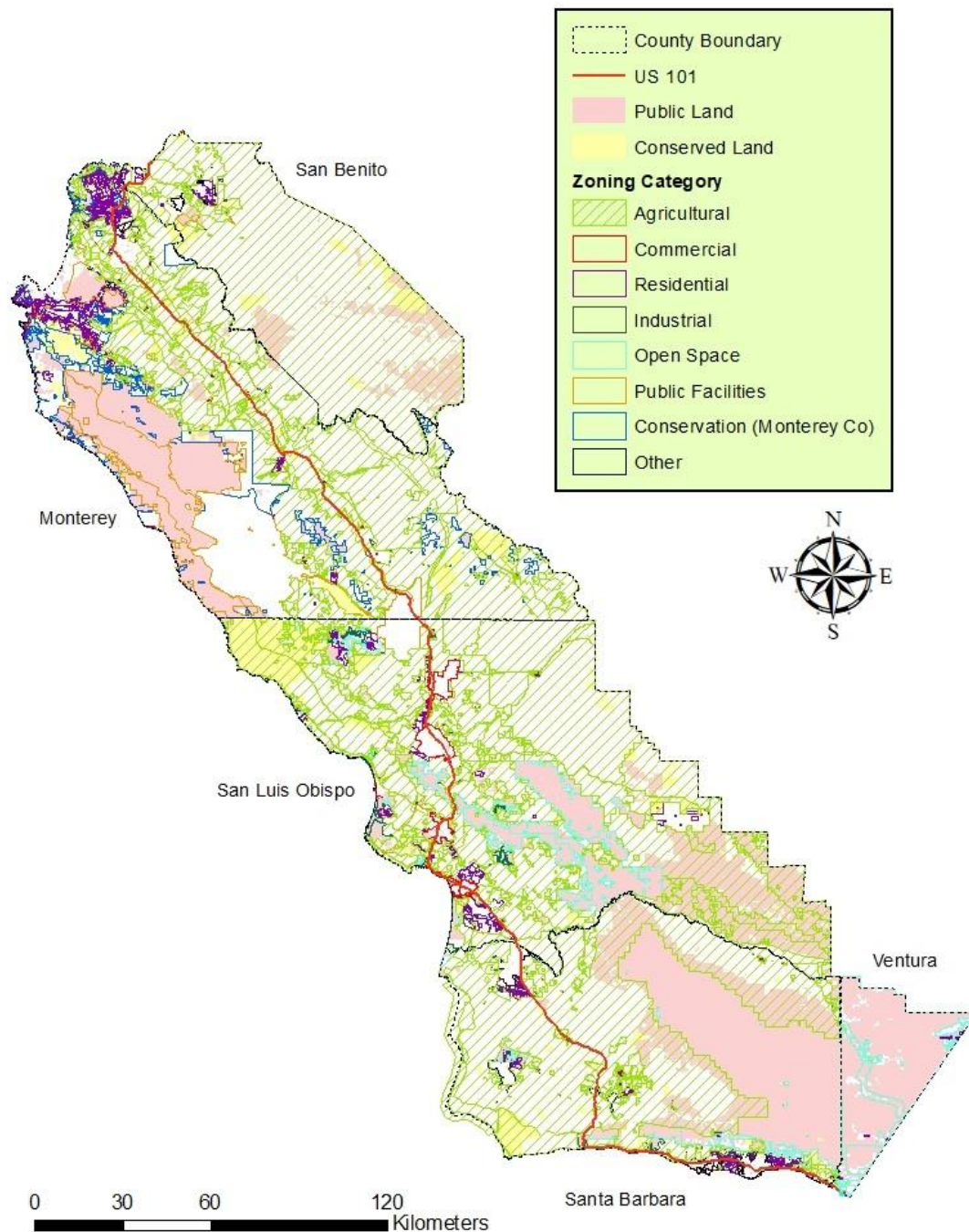


Figure 2.16 Protected lands and zoning status within the California Central Coast (County of San Benito 2023, County of San Luis Obispo 2020, County of Santa Barbara 2023, County of Ventura 2023, Monterey County Geographic Information Services Department 2023).

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