



Native Ungulates

Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: This document represents an initial evaluation of vulnerability for native ungulates in northern California based on expert input and existing information. Specifically, the information presented below comprises vulnerability factors selected and scored by regional experts, relevant references from the scientific literature, and peer-review comments and revisions (see end of document for a glossary of terms and brief overview of study methods). The aim of this document is to expand understanding of habitat vulnerability to changing climate conditions, and to provide a foundation for developing appropriate adaptation responses.

Peer reviewers for this document included Joshua Bush (California Department of Fish and Wildlife), Greg Giusti (University of California, Agriculture and Natural Resources), Cassandra Hagemann (U.S. Forest Service), and Chad Roberts (Tuleyome). Vulnerability scores were provided by Joshua Bush and Eureka workshop participants. Upper Lake workshop participants provided additional comments on the climate change vulnerability of this species group.

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Species Group Description

Native ungulates considered in this assessment include Columbian black-tailed deer (*Odocoileus hemionus columbianus*), tule elk (*Cervus canadensis nannodes*), and Roosevelt elk (*C. c.*)

roosevelti). Roosevelt elk occur from the Oregon border south into northern Mendocino County, and inland to the Cascades (CDFW 2018). In northern California, tule elk occur in Mendocino, Lake, Colusa, Glenn, Solano, and Yolo Counties, as well as in other isolated populations further south and east (e.g., San Benito, Merced, Fresno, and Inyo counties; CDFW 2018). Comparatively, black-tailed deer have a much wider distribution, occurring along the coast from central California to central British Columbia (Mackie et al. 2003). In more inland areas in California, black-tailed deer co-occur with other mule deer sub-species, including California mule deer (*O. h. californicus*) and Rocky Mountain mule deer (*O. h. hemionus*; Higley 2002; Mackie et al. 2003; Sommer et al. 2007).¹

As herbivores, native ungulates utilize a variety of different forest, woodland, grassland, shrubland, riparian, and wetland habitat types in northern California (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013; CDFW 2018). While they are habitat generalists, native ungulates require certain habitat elements for persistence, including availability of forage, cover, and standing/free water (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013). Most native ungulates in the study region are non-migratory, although some populations exhibit seasonal shifts in local distribution in response to vegetation availability (e.g., between south and north slopes or lower and higher elevations; Loft et al. 1998; Vuln. Assessment Workshop, pers. comm., 2017). Migratory populations exist in more topographically complex areas (e.g., Marble Mountains, Trinity Alps; Loft et al. 1998).

Native ungulates have significant cultural value to northern California tribes (Anderson 2005; Norgaard et al. 2016; Karuk Tribe 2019). Both elk and deer were historically hunted for food, and remain important subsistence resources (Anderson 2005; Nelson et al. 2008; Norgaard et al. 2016; Long & Lake 2018). Native ungulates were also traditionally used for clothing (hides), regalia, and instruments (Anderson 2005; Lake 2007; Norgaard et al. 2016; Long & Lake 2018). For example, antlers and bones from native ungulates were used as tools (Anderson 2005). In addition to their value to tribal groups, native ungulates also have high societal and recreational value for non-tribal groups as a prized hunting species (Sommer et al. 2007; Nelson et al. 2008).

Executive Summary

The relative vulnerability of native ungulates in northern California was evaluated as moderate by regional experts due to moderate sensitivity to climate and non-climate stressors, moderate-high exposure to projected future climate changes, and moderate-high adaptive capacity.

Native Ungulates	Rank	Confidence
Sensitivity	Moderate	High
Future Exposure	Moderate-High	Low
Adaptive Capacity	Moderate-High	High
Vulnerability	Moderate	High

¹ This assessment focuses specifically on Columbian black-tailed deer, one of six distinct sub-species of mule deer in California (Higley 2002; Mackie et al. 2003). While much of the scientific literature discusses mule deer and black-tailed deer collectively, where possible, information specific to black-tailed deer is identified.

Sensitivity & Exposure Summary	<p><u>Climate and climate-driven factors:</u></p> <ul style="list-style-type: none"> • Precipitation amount and timing, soil moisture, drought <p><u>Disturbance regimes:</u></p> <ul style="list-style-type: none"> • Wildfire, disease <p><u>Non-climate stressors:</u></p> <ul style="list-style-type: none"> • Residential and commercial development, agriculture, invasive and problematic species, roads/highways/trails, livestock grazing, fire suppression, recreation, timber harvest, hunting <p><u>Other sensitivities:</u></p> <ul style="list-style-type: none"> • Mineral levels (selenium)
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Native ungulates are sensitive to climate stressors and disturbance regimes that alter forage availability and quality, water availability, and adequate thermal and escape cover. For example, drought and/or generally drier conditions decrease forage and surface water availability, reducing habitat carrying capacity. In general, resource scarcity negatively affects native ungulate survival, reproduction, and recruitment, ultimately influencing population stability. For example, scarce resource conditions can enhance native ungulate vulnerability to predation and/or lead to poor body condition by requiring prolonged and farther-ranging foraging activity. Mixed-severity fire regimes generally benefit native ungulates by creating early seral conditions with abundant forage and a diverse landscape mosaic that supports both foraging and cover habitat elements. However, uncharacteristically large, severe fires can create large areas of unusable habitat, and increasing fire frequency and severity may drive shifts in vegetation community composition that reduces long-term forage availability and quality. Wildfire and disease can also cause direct native ungulate mortality. Diseases can also indirectly increase native ungulate vulnerability to malnutrition or predation.

Native ungulates are also sensitive to a variety of non-climate stressors. Several stressors (e.g., vehicle collisions, hunting) contribute to direct mortality, while others (e.g., recreation, road presence) elevate physiological stress and alter behavioral patterns which can, in turn, reduce fitness. Many other non-climate stressors affect habitat availability, connectivity, and quality for native ungulates. For example, land-use conversion for urban development and agriculture can displace resident native ungulate herds, reduce habitat availability, and increase habitat fragmentation, particularly for elk. The presence of roads, highways, trails, and fences can further fragment habitat and isolate native ungulates, as well as increase exposure to additional stressors (e.g., vehicle mortality, poaching). Many non-climate stressors also alter forage availability and quality. For example, invasive plants may be less palatable than native vegetation. Feral pig activity and livestock grazing (depending on management) can also reduce available forage and riparian habitat for native ungulates. Additionally, historical fire suppression followed by more recent reductions and changes in timber harvesting methods have inhibited the development of early seral forest conditions that provide abundant forage. Landscape fragmentation and reduced access to abundant, high-quality forage may also contribute to native ungulate selenium deficiencies, which can decrease fecundity and juvenile survival, ultimately affecting population stability. However, some non-climate factors may

benefit native ungulates. For example, human-modified landscapes may increase available forage (e.g., lawns, landscaping) or water sources (e.g., stock ponds).

Adaptive Capacity Summary	<p><u>Factors that enhance adaptive capacity:</u></p> <ul style="list-style-type: none"> + Highly mobile + Generalist habitat and foraging preferences + High societal value (e.g., for hunting, photography) may increase support for management <p><u>Factors that undermine adaptive capacity:</u></p> <ul style="list-style-type: none"> – Restricted distribution relative to historic distribution (particularly elk) – Limited life history diversity – Limited genetic diversity in elk
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Native ungulate distributions have been reduced over time by habitat loss and alteration. Elk distribution has further been influenced by the introduction of livestock. While elk have experienced greater losses in distribution than deer, their populations are currently increasing due to targeted management and reintroduction efforts. Black-tailed deer populations have declined from historic highs and population stability varies by location. Native ungulates are highly mobile, but habitat fragmentation may limit migration in response to climate change. Land-use changes and modifications (e.g., fencing) could particularly affect elk. Native ungulates generally exhibit limited life history diversity, which can undermine their ability to accommodate change. Elk are also susceptible to genetic bottlenecks due to past population declines, which increases their risk of experiencing inbreeding depression. Overall, society values native ungulates for a variety of reasons (e.g., hunting, photography, wildlife watching), which may increase support for management in the future.

Sensitivity and Exposure

Native ungulates were evaluated by regional experts as having moderate overall sensitivity (high confidence in evaluation) and moderate-high overall future exposure (low confidence) to climate and climate-driven factors, changes in disturbance regimes, and non-climate stressors.

Although no regional modeling is available, native ungulates are likely to exhibit some distribution shifts in response to a changing climate, tracking climate-driven range shifts of their preferred habitat types (e.g., Farrell 2015). In general, regional climate refugia for native ungulates may include higher elevations, northern slopes, riparian areas, seeps/springs, and the urban interface (which can provide abundant forage and reduced predation risk; Vuln. Assessment Workshop, pers. comm., 2017).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated native ungulates as having moderate sensitivity to climate and climate-driven factors (high confidence in evaluation), with an overall moderate-high future exposure to these factors within the study region (low confidence). Key climatic factors that

affect native ungulates include precipitation amount and timing, soil moisture, and drought.² Of these, factors that may benefit the species group include increased precipitation, which could increase green growth available for consumption, and earlier fall rainfall, which could increase food availability in advance of winter (Vuln. Assessment Workshop, pers. comm., 2017).

Precipitation amount and timing and soil moisture

California's Mediterranean-type climate requires native ungulates to alter forage species selection and spread out across landscapes in response to changing conditions (Sommer et al. 2007; Nelson et al. 2008; Innes 2011, 2013). For example, seasonal dry periods occurring in late summer and early fall cause drying of herbaceous vegetation and declines in crude protein of browse plant species (Sommer et al. 2007). As a result, late summer and fall are typically nutritionally-demanding periods for native ungulates (Sommer et al. 2007; Cobb 2010; Innes 2011, 2013). During this period, black-tailed deer rely on oak (*Quercus* spp.) leaves and acorn crops, as well as germinating plants stimulated by fall rains, to gain energy reserves prior to their winter breeding season (Sommer et al. 2007; Innes 2013). However, they will take advantage of irrigated agricultural fields (e.g., alfalfa), pastures, and developed water sources on grazing lands, if available (Vuln. Assessment Reviewers, pers. comm., 2018). Elk have also been shown to alter their forage species selection in response to seasonal changes in vegetation availability, switching from herbaceous plants to shrubby vegetation as the summer progresses and their preferred forage dries (Cobb 2010). Forage available to native ungulates is also affected by the timing and length of seasonal herbaceous vegetation green-up (Parker et al. 2009; Cobb 2010), which is controlled by soil moisture fluctuations associated with fall, winter, and spring precipitation (Eviner 2016).

Changes in precipitation patterns and resulting changes in soil moisture are also likely to exert short-term, bottom-up controls on native ungulate population numbers and landscape distribution by affecting forage availability, distribution, and quality (Sommer et al. 2007; Nelson et al. 2008; Parker et al. 2009; Forrester & Wittmer 2013). For example, reduced spring precipitation that causes earlier summer vegetation desiccation and senescence (Reever Morgan et al. 2007; Eviner 2016) could lead to higher deer fatality (Innes 2013). Similarly, any changes in precipitation (e.g., weather event timing, seasonal and annual precipitation volume) that affect oak tree survival (Mahall et al. 2009; McLaughlin & Zavaleta 2012) and alter acorn production (Tappeiner et al. 1990; Koenig et al. 1994, 1996; McDonald & Tappeiner 2002) could exert significant influence on deer survival in August and September when oak mast is a primary food source (Innes 2013).

Reduced forage availability and quality affects native ungulate survival, reproduction, and recruitment, ultimately influencing population stability (Innes 2011; Forrester & Wittmer 2013). A review of mule deer and black-tailed deer studies across the western U.S. found nutritional deficiency to be the largest ultimate cause of adult and fawn mortality (Forrester & Wittmer 2013). Prior studies in the northern California study region have found similar patterns, even

² All climate and climate-driven factors presented were ranked as having a moderate or higher impact on this species group.

during periods when deer population numbers were much higher (Longhurst et al. 1976). Further, a nutrition enhancement study in Colorado found that young and adult female mule deer survival increased significantly in response to enhanced nutrition (Bishop et al. 2009). Nutritional condition also influences black-tailed deer fecundity and fawn birth weight (Parker et al. 2009). For example, does in poor body condition are less likely to carry twins, reducing fawn recruitment into the population (Vuln. Assessment Reviewers, pers. comm., 2018). Similarly, a variety of elk survival, reproduction, and recruitment metrics are affected by forage availability and nutritional status, including age of first breeding, seasonal breeding timing, conception success, fetal and calf growth rates and survival, and adult survival (Innes 2011).

Reductions in food availability can also influence native ungulate vulnerability to predation (Bishop et al. 2009; Wittmer et al. 2014). For example, a study in the Mendocino National Forest found that black-tailed deer with access to high quality forage within their home and summer range were less vulnerable to predation than individuals who were forced to forage more widely (Wittmer et al. 2014). The outcome was likely from increased likelihood of running into predators during foraging activities, as well as reduced ability to flee and/or withstand attack due to lower energy reserves and degraded body condition due to inadequate nutrition (Wittmer et al. 2014). In general, nutritionally-stressed native ungulates take more risks to forage, which can increase vulnerability to predation and/or other mortality agents (e.g., vehicle strikes; Innes 2011, 2013).

Regional Precipitation & Soil Moisture Trends ³	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • 2.6–9.4 cm (1.0–3.7 in) increase in mean annual precipitation between 1900 and 2009 for the Northwestern California, Southern Cascade, and Great Valley ecoregions (Rapacciuolo et al. 2014) • No trends available for soil moisture 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • 23% decrease to 38% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the North Coast, Northern Coast Range, Northern Interior Coast Range, Klamath Mountain, Southern Cascade, and Great Valley ecoregions (Flint et al. 2013; Flint & Flint 2014)⁴ • Seasonal changes are projected to be more significant as the wet season becomes wetter and shorter (i.e., later onset of fall rains and earlier onset of summer drought) and the dry

³ Trends in climate factors and natural disturbance regimes presented in this and subsequent summary tables are not species group-specific; rather, they represent broad trends and future projections for the study region. The precipitation, temperature, climatic water deficit, and snowpack projections for this project are derived from the Basin Characterization Model, which uses modified Jepson ecoregions (Flint et al. 2013; Flint & Flint 2014). Projections for all other factors are based on a review of relevant studies in the scientific literature. For this project, exposure was evaluated by calculating the magnitude and direction of projected change within the modified Jepson ecoregions that include habitat distribution within the study geography.

⁴ Projections for changes in annual and seasonal precipitation by ecoregion can be found in the full climate impacts table (<https://bit.ly/2LHgZaG>).

Regional Precipitation & Soil Moisture Trends ³	
	<p>season becomes drier and longer (Pierce et al. 2018; Swain et al. 2018)</p> <ul style="list-style-type: none"> • Overall, interannual variability is expected to increase (Pierce et al. 2018; Swain et al. 2018) • Decreased top-level soil moisture is likely even if precipitation increases due to temperature-related changes in evaporative demand (Pierce et al. 2018)
Summary of Potential Impacts on Species Group <i>(see text for citations)</i>	
<ul style="list-style-type: none"> • Altered forage availability, quality, and distribution, influencing native ungulate population numbers and landscape distribution • Reduced forage availability may undermine population stability by affecting survival, reproduction, and recruitment • Reduced forage availability may increase vulnerability to predation or other mortality agents 	

Drought

Native ungulates are adapted to the seasonal summer drought characteristic of California’s climate (Sommer et al. 2007). However, more frequent and/or prolonged drought events may significantly reduce forage availability and habitat carrying capacity for native ungulates, potentially contributing to population declines (Innes 2011, 2013; Pierce et al. 2012; CDFW 2018). For example, drought-induced shortages in food availability can severely limit the reproductive success and fawn recruitment of black-tailed deer (Sommer et al. 2007; Innes 2013). Drought impacts on forage availability and native ungulate survival can be particularly acute following wildfire, when forage may already be scarce (Innes 2011). Abrupt and elevated mortality levels experienced during extreme events, such as drought, can have long-lasting impacts on population numbers through cohort effects (Forrester & Wittmer 2013).

Climate-driven changes in drought frequency or length may also deplete or cause drying of drinking water sources such as springs and seasonal swales (Reiter et al. 2018). Reduced surface water availability may indirectly elevate mortality by requiring native ungulates to travel further to obtain water, which can increase predation risk and reduce body condition and recruitment (Vuln. Assessment Workshop, pers. comm., 2017). However, management activities such as spring improvement or providing seasonal artificial water impoundments could potentially mitigate these impacts (Vuln. Assessment Reviewers, pers. comm., 2018).

Drought events are also likely to cause shifts in landscape distribution as native ungulates attempt to find areas with available forage (Innes 2011; CDFW 2018) and water (Innes 2011). While elk and deer do not typically compete for forage due to different preferences, seasonal or long-term resource scarcity may increase competition between species with overlapping ranges, further limiting nutritional intake, fitness, and survival (Sommer et al. 2007).

Regional Drought Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • Drought years have occurred twice as often over the last two decades compared to the previous century (Diffenbaugh et al. 2015) • 2012–2014 drought set records for lowest precipitation, highest temperatures, and most extreme drought indicators on record (Griffin & Anchukaitis 2014; Diffenbaugh et al. 2015) 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • Drought years are twice as likely to occur over the next several decades due to increased co-occurrence of dry years with very warm years (Cook et al. 2015) • 80% chance of multi-decadal drought by 2100 under a high-emissions scenario (Cook et al. 2015) • Severe droughts that now occur once every 20 years will occur once every 10 years by 2100 and once-in-a-century drought will occur once every 20 years (Pierce et al. 2018)
Summary of Potential Impacts on Species Group <i>(see text for citations)</i>	
<ul style="list-style-type: none"> • Reductions in forage availability and habitat carrying capacity, reducing native ungulate population numbers • Reduced availability and/or drying of water sources used for drinking • Potential enhancement of elk-deer competition as a result of forage and water shortages 	

Sensitivity and future exposure to changes in natural disturbance regimes

Regional experts evaluated native ungulates as having moderate-high sensitivity to changes in natural disturbance regimes (high confidence in evaluation), with an overall moderate future exposure to these stressors within the study region (low confidence). Key natural disturbance regimes that affect native ungulates include wildfire and disease.⁵

Wildfire

Wildfire regimes across northern California were historically influenced by both lightning ignitions and human burning practices by tribes and early settlers (Anderson 2005; Van de Water & Safford 2011; Safford & Van de Water 2014; Norgaard et al. 2016; Karuk Tribe 2019). Northern California tribes traditionally used frequent cultural burning in a variety of habitats to enhance the availability of early-successional and nutritious forage for native ungulates (Anderson 2005; Long et al. 2016; Norgaard et al. 2016; Karuk Tribe 2019), as well as to facilitate hunting (e.g., by clearing underbrush; Anderson 2005). For example, coastal prairies were burned in the late summer and fall to reduce shrub encroachment and maintain forage availability for elk (Anderson 2005; Norgaard et al. 2016). Early settlers followed similar practices until the early- to mid-1900s, when political and social pressures reduced the application of fire (Taylor & Skinner 1998; Skinner et al. 2006; Stuart & Stephens 2006; Steel et al. 2015). Fire exclusion has altered regional habitat quality for native ungulates (along with other factors that have reduced the abundance of early seral landscape conditions; Longhurst

⁵ All disturbance regimes presented were ranked as having a moderate or higher impact on this species group; native ungulates were evaluated as having moderate-high future exposure to wildfire while having only low-moderate future exposure to disease.

et al. 1976; Long et al. 2016; Norgaard et al. 2016), causing reductions in forage availability, diversity, and quality, with implications for recruitment and population size (Sommer et al. 2007).⁶

Wildfires (particularly mixed-severity fires) generally benefit native ungulate populations by creating diverse landscape conditions, including increased edge habitat and open foraging areas with regenerating shrubs and herbaceous plants, interspersed with forested and shrub areas used for cover (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013). For example, in forested habitats wildfire opens tree canopies and stimulates the understory shrub growth, which are preferred native ungulate browse (Sommer et al. 2007; Nelson et al. 2008; Innes 2011, 2013). In shrubland habitats, wildfires promote new shrub and herbaceous growth, both of which have higher nutritional quality than decadent, mature shrubs (Sommer et al. 2007; Nelson et al. 2008; Innes 2011, 2013). In grassland habitats, wildfire reduces thatch (Ford & Hayes 2007; Stromberg et al. 2007), improving forage availability and access (Innes 2011). Fire may improve native ungulate forage access by lowering browse height and/or removing tree and shrub cover that previously limited access to higher quality forage (Innes 2011), although downed fire debris may also affect mobility and forage access (Norgaard et al. 2016). Diverse stand ages created by mixed severity and discontinuous fire also creates staggered vegetation maturation, helping maintain landscape-scale native ungulate foraging opportunities (Innes 2011, 2013). However, exact impacts and the longevity of elevated forage quality and availability vary by habitat type (Sommer et al. 2007; Innes 2011). For example, deer density in California shrublands and redwood forests has been shown to increase for up to 10 years following fire, while in oak woodlands, enhanced deer usage of burned areas continued for only up to 4 years following fire (Innes 2013 and citations therein).

Additionally, the impact of fire is dependent on fire size, frequency, and severity. For example, large fires can eliminate suitable thermal and escape cover and forage availability over large land areas, displacing native ungulates, reducing short-term habitat suitability, and potentially enhancing starvation and/or vulnerability to predation (Sommer et al. 2007; Innes 2011, 2013). Too-frequent fires can reduce short- and long-term forage availability by eliminating key species and/or altering community composition or structure (Innes 2011, 2013). For example, in chaparral habitats, too-frequent fire can create simplified shrub communities that may feature less palatable species (Risser & Fry 1988; Haidinger & Keeley 1993; Keeley & Davis 2007; Cornwell et al. 2012; Halsey & Keeley 2016) and/or facilitate type conversion to less valuable forage, such as exotic annual grassland (Haidinger & Keeley 1993; Lenihan et al. 2008; Keeley & Brennan 2012; Halsey & Keeley 2016). Similarly, in oak woodlands, too-frequent, high-severity fires can reduce the abundance of mature, acorn-bearing oak trees (Swiecki & Bernhardt 1998; George & Alonso 2008; Long et al. 2016; Norgaard et al. 2016; Nemens et al. 2018), affecting long-term availability of oak mast for native ungulate consumption (Sommer et al. 2007).

⁶ Refer to the section on non-climate stressors for a more complete discussion of the impacts of fire suppression on native ungulates.

Native ungulates are vulnerable to direct fire mortality, although fire mortality rates are generally low (Innes 2011, 2013). Mortality rates are typically highest during conflagrations with fast rates of spread, active crown burning, and heavy ground smoke (Innes 2011, 2013). For example, in Yellowstone National Park, elk mortality occurred primarily in areas where fire spread ranged from 2.5–4.2 miles/hour (4.1–6.9 km/hour) and where fire fronts were greater than 1 mi (2 km) wide (Singer et al. 1989). Fawns and calves may be particularly vulnerable to fire mortality during their initial week of life, when they are relatively immobile (Innes 2011, 2013).

Regional Wildfire Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • 85% of U.S. Forest Service lands in northern California are burning less frequently compared to pre-1850 fire return intervals, largely due to fire suppression (Safford & Van de Water 2014) • Fire size and total area burned increased on U.S. Forest Service lands in northwestern California between 1910-2008, with the highest values occurring after 2000 (Miller et al. 2012) • Changes in large fires (over 400 ha) in the inland northern California/Sierra Nevada region since the 1970s (Westerling 2016): <ul style="list-style-type: none"> ○ 184–274% increase in frequency ○ 270–492% increase in total area burned ○ 215% increase in length of the fire season • Changes in fire size, area burned, and fire frequency over the past several decades remain well below historical tribally-influenced frequency and extent of burning in California (Stephens et al. 2007) • No significant trends in the average areal proportion of high-severity fire were documented in northwestern CA from 1984–2008 (Miller et al. 2012; Parks et al. 2015; Law & Waring 2015; Keyser & Westerling 2017) <ul style="list-style-type: none"> ○ The relatively short period of record for fire severity data may obscure long-term trends ○ To date, there are no peer-reviewed studies on trends in northern California fire severity that include data from the last ten years 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • State-wide, up to 77% increase in mean annual area burned and 50% increase in the frequency of extremely large fires (>10,000 ha) by 2100 (Westerling 2018) <ul style="list-style-type: none"> ○ Greatest increases in burned area (up to 400%) occur in montane forested areas in northern California (Westerling et al. 2011; Westerling 2018) ○ Less significant increases or possible decrease along the North Coast (Westerling et al. 2011) • Little projected change in fire severity in northwestern California by 2050 in models based solely on historical fire-climate relationships (Parks et al. 2016) <ul style="list-style-type: none"> ○ However, human activity and fuel buildup from decades of fire suppression have altered historical fire-climate relationships (Taylor et al. 2016; Syphard et al. 2017; Wahl et al. 2019), and projections that incorporate these factors suggest that more significant increases in fire severity and size may occur (Mann et al. 2016; Wahl et al. 2019) • The majority of impacts to natural and human ecosystems come from extreme fire events (i.e., fires that have a low probability of occurring in any given place and time), which are likely to increase over the coming century (Westerling 2018) <ul style="list-style-type: none"> ○ Generally, these patterns are not well-represented in studies that evaluate indices of mean fire size, intensity/severity, etc.

Regional Wildfire Trends

Summary of Potential Impacts on Species Group (see text for citations)

- *Immediate:*
 - Potential mortality, especially for fawns and calves; mortality most likely with large, fast-spreading fires with crown burning and heavy ground smoke
 - Temporary reduction of forage availability and elimination of thermal and escape cover, displacing native ungulates; may enhance starvation and/or predation, especially if fire occurs over large land area
- *Short-term (~2-year):*
 - Creates early successional habitat with higher forage availability, diversity, and quality (e.g., regenerating herbaceous understory and shrubs)
 - Often improves forage access (e.g., lowers browse height, improves within-stand access by removing dense shrub and tree cover), although downed debris can limit mobility
 - Habitat recovery and utilization by native ungulates may be slower in the wake of larger-scale, high-severity burns
- *Long-term:*
 - Mixed severity fire regimes promote landscape-scale diversity in forage availability and quality and mosaic habitat conditions (e.g., edge habitat, open areas, thermal and escape cover) that benefit native ungulates
 - Increasing fire frequencies may reduce forage availability and/or quality by eliminating key forage species and/or driving shifts in vegetation community composition

Disease

Disease has not historically played a large role in driving region-wide native ungulate population declines, but can act as a localized mortality agent (Forrester & Wittmer 2013; CDFW 2018). For example, an epizootic adenovirus outbreak in northern California in 1993 caused the death of over 1000 mule deer across 17 counties (Woods et al. 1996), and Johne's disease (caused by the bacterium *Mycobacterium paratuberculosis*) has been linked with elk deaths in Point Reyes National Seashore (CDFW 2018). Disease outbreaks tend to occur when populations are artificially dense (i.e., in and around suburban development with abundant forage resources and lower rates of predation) or limited to confined landscapes as at Point Reyes National Seashore (Farnsworth et al. 2005; Polfus & Krausman 2012; Sorensen et al. 2014).

Even if diseases do not cause direct mortality, they may indirectly increase native ungulate vulnerability to malnutrition or predation (Forrester & Wittmer 2013). For example, chronic wasting disease has been shown to increase mule deer vulnerability to mountain lion (*Puma concolor*) predation in Colorado, either by reducing deer vigilance and flight capacity and/or by degrading condition causing them to be targeted for attack (Krumm et al. 2009). Both black-tailed deer and elk are susceptible to chronic wasting disease (Sorensen et al. 2014; CDFW 2018), and while it is not currently present in California, it could have significant implications for native ungulate population stability if it appears (Vuln. Assessment Workshop, pers. comm., 2017). Additional diseases and parasites that may reduce general fitness or cause direct mortality include calf scours and liver flukes, among others (Vuln. Assessment Reviewers, pers. comm., 2018).

Regional Disease Trends	
<p><i>Historical & current trends:</i></p> <ul style="list-style-type: none"> • Disease primarily acts as a local mortality agent, and has not been linked with region-wide population declines (Forrester & Wittmer 2013; CDFW 2018) • Disease outbreaks are most common with high population density (Farnsworth et al. 2005; Polfus & Krausman 2012; Sorensen et al. 2014) 	<p><i>Projected future trends:</i></p> <ul style="list-style-type: none"> • No native ungulate disease projections are available
Summary of Potential Impacts on Species Group <i>(see text for citations)</i>	
<ul style="list-style-type: none"> • Increased localized mortality • May enhance native ungulate vulnerability to malnutrition or predation 	

Dependency on habitat and/or other species

Regional experts evaluated native ungulates as having low dependency on sensitive habitats (high confidence in evaluation) and moderate dependency on forage species (high confidence).

Native ungulates are habitat generalists, utilizing a variety of different habitat types across a wide elevational range within the study region, including forests, oak woodlands, shrublands, grasslands, riparian areas, and wetlands (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013; Wittmer et al. 2014). While native ungulates are not dependent on one specific habitat type, they are reliant on the availability of three general habitat conditions: forage, cover, and free/standing water (Sommer et al. 2007; Innes 2011). Fawns and calves are highly vulnerable to predation during the first several weeks of life (Forrester & Wittmer 2013), and during this period, they require adequate cover for hiding (Forrester & Wittmer 2013). Some habitats are particularly important: for example, riparian, spring, and wetland areas act as critical water sources and thermal refugia for deer, providing fresh water, fawning and escape cover, and shade (Sommer et al. 2007). Additionally, many native ungulates in the region rely on stock ponds, springs, artificial wildlife drinkers, or seasonal swales for water (Vuln. Assessment Workshop, pers. comm., 2017).

Both deer and elk are herbivores, but forage preferences vary. Black-tailed deer are primarily browsers, consuming woody shrub leaves, twigs, and buds (Nelson et al. 2008). However, deer also consume seasonal forbs, grasses, fruits, lichens, oak mast, and acorns (Nelson et al. 2008). Oak leaves and acorns represent an important food source in late summer and through early winter, before annual plants stimulated by fall rains become available (Sommer et al. 2007). In chaparral habitats, chamise (*Adenostoma fasciculatum*) is particularly important, acting as a “fall-back” browse resource that deer can rely on for adequate nutrition and moisture even when surrounding herbaceous vegetation is dry (Sommer et al. 2007). Due to a smaller rumen in relation to body size, deer have to be more selective in their foraging choices than elk, which can eat large quantities of low quality forage (e.g., grass; Nelson et al. 2008). Roosevelt and tule

elk are opportunistic feeders. They typically consume grasses and forbs in the spring, forbs and browse in late summer, dry grasses and browse in fall, and a combination of grasses, forbs, and shrubs in winter (CDFW 2018).

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated native ungulates as having moderate-high sensitivity to non-climate stressors (high confidence in evaluation), with an overall moderate-high current exposure to these stressors within the study region (high confidence). Key non-climate stressors that affect native ungulates include residential and commercial development, agriculture, invasive and problematic species, agriculture, roads/highways/trails, livestock grazing, fire suppression, timber harvest, recreation, and hunting.⁷

Residential and commercial development and agriculture

Land-use conversion has contributed to native ungulate habitat loss and fragmentation across the study region, impacting abundance, distribution, and population connectivity (Sommer et al. 2007; Bolger et al. 2008; Innes 2011, 2013; CDFW 2018). Impacts have been particularly acute for elk species; comparatively, black-tailed deer appear to thrive in many human-modified environments (e.g., golf courses, suburban areas; Vuln. Assessment Reviewers, pers. comm., 2018). Impacts also vary depending on extent of conversion and associated management choices. For example, elimination of riparian vegetation and utilization of deer-proof fencing in many commercial agricultural areas completely displaces resident native ungulate herds and affects their ability to move through agricultural zones to more suitable habitat areas, including climatic refugia (Sommer et al. 2007; Schloss et al. 2012). These landscape modifications can also force native ungulates onto roadways, elevating mortality risk (Vuln. Assessment Reviewers, pers. comm., 2018). Increasing urban development pressure in the wildland urban interface will likely increase enhance ungulate exposure to domestic animals and livestock, potentially increasing disease transmission and mortality (Sommer et al. 2007; Polfus & Krausman 2012). Historical and on-going loss of oak woodland areas to development and agriculture (Gaman & Firman 2006) is particularly problematic for deer due to their reliance on oak leaves and mast during resource-scarce late summer and early fall months (Sommer et al. 2007).

However, urban and agricultural development can also have some beneficial impacts on native ungulates. For example, development can enhance the availability of forage (e.g., lawns, gardens, landscaping, crops) and artificial water sources (e.g., ponds), although native ungulate use of these features can be viewed as problematic by some landowners (Sommer et al. 2007; Polfus & Krausman 2012; CDFW 2018). Native ungulate-friendly management choices can further promote co-habitation; for example, the use of hedgerows and cover crops can provide forage while relieving grazing pressure on primary crops, and maintenance and restoration of riparian corridors can facilitate ungulate movement through agricultural landscapes to maintain habitat connectivity (Sommer et al. 2007).

⁷ All non-climate stressors presented were ranked as having a moderate or higher impact on this species group.

Invasive and problematic species

There are few direct studies on the impacts of invasive vegetation on native ungulates. However, invasive plants are prevalent in many habitat types utilized by native ungulates in northwestern California, including grasslands, chaparral, oak woodlands, and forests (Jimerson & Carothers 2002; Keeley et al. 2011; Eviner 2016; Halsey & Keeley 2016). It is widely believed that invasive vegetation reduces forage availability, quality, and diversity for native ungulates by altering native plant communities, particularly under scenarios of high disturbance (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008). For example, medusahead (*Taeniatherum caput-medusae*), barbed goatgrass (*Aegilops triuncialis*), and yellow star-thistle (*Centaurea solstitialis*) commonly outcompete and displace native perennial and naturalized non-native annual grassland species (Eviner 2016), and are less palatable and nutritionally valuable for native ungulate species (Sommer et al. 2007). However, there are some non-native vegetation species that increase native ungulate forage, including filaree (*Erodium* spp.) and clover (*Trifolium* spp.; Sommer et al. 2007).

Exotic vegetation can also affect native ungulates indirectly. For example, exotic establishment and spread can increase competition among grazing animals by concentrating forage on remnant native-dominated resource patches (Hayden et al. 2008). Exotic vegetation can also hinder native ungulate movement and access to water and/or areas with higher forage availability (Nelson et al. 2008). For example, Himalayan blackberry (*Rubus armeniacus*) often invades riparian areas and meadows, and can limit movement of large animals such as native ungulates (Nelson et al. 2008). Additionally, exotic vegetation can exacerbate changing climate conditions that affect native ungulates, such as declining surface water availability and changing fire regimes (Hayden et al. 2008; Nelson et al. 2008). For example, exotic annual grasses facilitate more frequent fires (Dukes & Shaw 2007; Livingston & Varner 2016). While deer and elk prefer early successional habitat created by fires, too-frequent fires can eliminate some native vegetation species utilized for forage (Innes 2011, 2013).

Native ungulates are also sensitive to negative interactions with wild pigs (*Sus scrofa*; Sommer et al. 2007). Wild pigs directly compete with native ungulates for acorns, and can further reduce forage availability through their rooting behavior, which has been found to decrease above-ground biomass in oak woodlands and reduce oak seedling recruitment (Sommer et al. 2007). Wild pigs may also prey on native ungulate young, including black-tailed deer fawns (Vuln. Assessment Workshop, pers. comm., 2017).

Roads, highways, and trails

Native ungulates exhibit altered behavior in response to transportation corridors (Sommer et al. 2007; Bolger et al. 2008; Cobb 2010; Prokopenko et al. 2017). Elk, in particular, avoid the busiest areas (e.g., larger roads and highways, busy trails; Sommer et al. 2007; Cobb 2010; Prokopenko et al. 2017). Road and trail traffic can also heighten physiological stress and cause elk to lose foraging time due to enhanced vigilance (i.e., switching to alert behavioral mode as cars or trail users pass; Cobb 2010; Ciuti et al. 2012).

Roads, highways, and trails can also limit local and regional movement patterns, potentially shifting or restricting home ranges (Sommer et al. 2007; Bolger et al. 2008; Cobb 2010; Prokopenko et al. 2017). Roads and highways typically fragment native ungulate habitat in the absence of wildlife-friendly designs (e.g., underpasses or overpasses), limiting herd movement and access to crucial habitat areas and/or climate refugia (Sommer et al. 2007). By restricting landscape movements, roads and highways can reduce genetic exchange and render herds more vulnerable to local extreme events (e.g., disease outbreaks), ultimately affecting population health and size (Sommer et al. 2007). Roads and highways can also contribute to direct mortality via vehicle collisions (Trombulak & Frissell 2000).

Roads, highways, and trails can also alter habitat conditions necessary for native ungulate survival. For example, road construction often contributes to a loss of native vegetation utilized for cover and/or food; even if roadside vegetation is palatable, roadside grazing increases deer and elk mortality risk from vehicle strikes (Sommer et al. 2007). Additionally, increased road, highway, and trail prevalence can increase poaching, off-highway vehicle use, wildfire risk from vehicle ignitions, and/or exotic species spread, exacerbating other stressors known to affect native ungulate populations (Trombulak & Frissell 2000; Sommer et al. 2007).

Livestock grazing

Competition between livestock and native ungulates has likely increased as natural landscapes available to native ungulates have declined (Loft et al. 1998). Many interactions between livestock and native ungulate are negative (Loft et al. 1998; Schieltz & Rubenstein 2016), although impacts vary depending on habitat, livestock species, stocking rate, and grazing timing, among other factors (Loft et al. 1998; Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013; Schieltz & Rubenstein 2016). Livestock grazing can reduce the availability and diversity of forage for native ungulates, including herbaceous vegetation, browse, and oak mast (Loft et al. 1998; Sommer et al. 2007). Competition impacts are particularly acute during resource-scarce periods (e.g., summer and late fall, drought periods; Loft et al. 1998; Sommer et al. 2007). Livestock can also compete with native ungulates for available surface water (Sommer et al. 2007). Livestock presence may also alter native ungulate behavior and energy expenditures (Innes 2011), and they increase the opportunity for disease transmission (Cobb 2010).

Livestock grazing can affect native ungulates indirectly by altering habitat conditions. Grazing and overutilization of shrub and herbaceous cover in riparian areas from spring to fall can degrade riparian habitat conditions, especially hiding and escape cover needed for fawn and adult survival (Loft et al. 1998; Sommer et al. 2007). Similarly, grazing can have significant impacts on oak recruitment (Swiecki & Bernhardt 1998; Tyler et al. 2006; Arévalo et al. 2009; Davis et al. 2011); oak leaves and acorn crops are important for fawn survival and adult nutrition, so declines in oak woodland abundance and diversity as a result of grazing can be detrimental (Sommer et al. 2007).

There are some beneficial impacts of livestock grazing on native ungulates (Sommer et al. 2007; Schieltz & Rubenstein 2016). For example, sheep husbandry on the north coast improved

habitat conditions for native deer in the 1950s and 1960s via range improvements, water impoundments, nutritional supplements, and predator control, leading to deer population irruptions (Vuln. Assessment Reviewers, pers. comm., 2018). Regionally, well-managed livestock grazing may reduce annual grass density, releasing perennial forbs preferred by native ungulates (Loft et al. 1998; Sommer et al. 2007; Innes 2013). A worldwide review of livestock grazing impacts on wildlife species found that grazing can facilitate the development and availability of preferred forage, but noted that research is needed to fully understand potential impacts at the local scale, particularly across longer timeframes and multiple seasons, with different livestock species, and at variable stocking rates (Schieltz & Rubenstein 2016).

Fire suppression

Fire suppression activities beginning in the early- to mid-1900s have excluded fire from large portions of the landscape in northern California (Skinner et al. 2006, 2009; Stuart & Stephens 2006; Cornwell et al. 2012; Steel et al. 2015; Safford & Stevens 2017), including habitats utilized by native ungulates. Altered habitat characteristics in the absence of fire, particularly the lack of early seral conditions and maintenance of non-forested landscapes (e.g., coastal prairies), can reduce habitat availability, quality, and utilization by native ungulates, limiting population numbers (see Table 1; Loft et al. 1998; Hayden et al. 2008; Norgaard et al. 2016; Long & Lake 2018; Karuk Tribe 2019). For example, in shrubland habitat types, the development of mature stand conditions in the absence of fire has reduced browse availability, diversity, and quality for black-tailed deer (Sommer et al. 2007). Specifically, the development of mature chaparral canopies in the absence of fire reduces the abundance and diversity of understory herbaceous vegetation (England 1988), and old, mature shrubs are less nutritious and palatable relative to new growth (Sommer et al. 2007). Additionally, the development of dense, late-seral shrub stands can reduce thermal and escape cover, and prevent movement and access to higher quality forage areas (Sommer et al. 2007). In coastal prairie habitats, the absence of fire allows encroachment of woody species (Ford & Hayes 2007; Stromberg et al. 2007; BLM 2008, 2013), which can reduce habitat extent and forage availability for elk (Underwood et al. 2003; Lake 2007; Norgaard et al. 2016). Fire suppression in forest and woodland habitats has contributed to significant changes in forest structure, composition, and landscape distribution (Taylor & Skinner 2003; Gilligan & Muir 2011; Cocking et al. 2012; Steel et al. 2015; Safford & Stevens 2017), reducing the abundance and productivity of oaks and other species important to native ungulates (Lake 2007; Cocking et al. 2012; Lake & Long 2014; Long et al. 2016; Norgaard et al. 2016).

Table 1. Impacts of fire suppression on the resilience of native ungulates to climate stressors and climate-driven changes in fire regimes (table adapted from Norgaard et al. 2016).

Prior to Fire	During Fire	After Fire
<ul style="list-style-type: none"> • Declines in forage quantity and quality, impacting ungulate populations • Increased tree encroachment into meadows, reducing elk habitat 	<ul style="list-style-type: none"> • Disturbance and/or displacement of native ungulates from desired habitat during fire suppression activities • Mortality can occur during backburning if individuals cannot escape 	<ul style="list-style-type: none"> • Increased or decreased travel and mobility with the aid of fire lines • Impacts on feeding and mobility due to post-fire erosion control treatments and other post-fire repairs
Source(s): Karuk DNR 2009; Norgaard 2014; Norgaard et al. 2016		

Timber harvest

Some timber harvest activities (e.g., silviculture) can help create early successional habitats that provide abundant and high quality forage for native ungulates, enhancing growth and recruitment (Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013). The impacts of timber harvest on native ungulates vary depending on scale, method (e.g., clear-cut vs. thinning), pre-harvest forest condition, and post-harvest activities, among other factors (Nelson et al. 2008; Innes 2011). In Oregon, thinned areas (i.e., uneven-aged management) were less utilized by elk than clear-cut (i.e., even-aged areas) or untreated areas 5 years following treatment, likely because they had lower forage availability than clear-cut areas and less cover than untreated areas (Severson et al. 1983 cited in Innes 2011). However, although clear-cuts can stimulate ungulate population increases in the short- to medium-term (30–40 years), subsequent succession to dense, even-aged coniferous forests causes eventual declines (Hayden et al. 2008; Nelson et al. 2008). Overall, decreased timber harvesting on public lands combined with successional progression of past clear-cuts to dense, homogenous forests has likely reduced foraging opportunities for native ungulates across the region by decreasing landscape-level availability of early-successional habitat (Nelson et al. 2008; CDFW 2018). Current and future timber management activities for native ungulate benefits must be balanced with other management objectives, such as the legal requirement under the Northwest Forest Plan to manage federal forest lands in northwestern California for late successional-dependent wildlife such as the northern spotted owl (Spies et al. 2018).

Some timber management practices can be detrimental for native ungulates. For example, chemical treatments of understory plants in timber plantations can reduce available herbaceous and shrub forage (Nelson et al. 2008; CDFW 2018). Similarly, removal of certain tree species relied on by native ungulates for food, such as black oak (*Quercus kelloggii*) and Oregon white oak (*Q. garryana*), can reduce native ungulate population numbers (Loomis et al. 1995). Specifically, Loomis et al. (1995) found that in mixed conifer forests, reducing oak basal area to 5 and 10 ft²/acre during timber harvest reduced deer carrying capacity by 26% and 16%, respectively, while retaining 30 ft²/acre of oaks during timber harvest increased deer carrying capacity by 33%.

Recreation

Recreational activities can alter native ungulate behavior in several ways, including short-term relocation in response to human disturbance (i.e., fleeing) and/or longer-term avoidance of high-use areas such as those adjacent to forest roads or trails (Sommer et al. 2007; Ciuti et al. 2012; Coppes et al. 2017). These behavioral modifications, which are most significant in elk, can reduce native ungulate fitness by elevating energetic costs and/or decreasing time spent foraging (Sommer et al. 2007; Coppes et al. 2017), though the severity of impact depends on recreational activity (Ciuti et al. 2012). Off-highway vehicle use typically has the most impact on behavior, causing reductions in foraging time as elk spend extended time in an alert, observational mode (Ciuti et al. 2012). However, a study in Alberta, Canada also found that non-motorized recreational activities can impact native ungulate behavior; hiking caused elk relocation, while equestrian and biking had no effect on elk behavior, likely because equestrian and cyclists typically stay on established trails, making their presence more predictable (Ciuti et al. 2012). Studies in other regions have found shifts to nocturnal foraging behavior in response to recreation intensity (Coppes et al. 2017). Peak periods of regional recreation use (i.e., summer) often coincide with fawning and lactation in native ungulates, and recreation pressure in California is only likely to increase with continued population growth (Sommer et al. 2007).

Hunting

All native ungulates can legally be hunted in California, subject to state regulation (CDFW 2018). Hunting pressure affects the local movements and population numbers of native ungulates, but is managed by state agencies with the goal of maintaining herds at landscape-sustainable levels (Wittmer et al. 2014; CDFW 2018). For example, data on black-tailed deer populations in northwestern California are currently being collected, and refined population numbers will inform future hunting tag allotments (Vuln. Assessment Workshop, pers. comm., 2017). Revenue generated from the sale of big game hunting tags provides funding for native ungulate habitat management, with additional benefits to other wildlife species (Wittmer et al. 2014). Poaching occurs in some areas, with localized impacts on population numbers (CDFW 2018).

Sensitivities to other factors

In addition to the factors discussed above, regional experts evaluated native ungulates as having moderate-high sensitivity to selenium levels (high confidence in evaluation).

Selenium is a dietary trace mineral obtained from forage and mineral licks, and it plays a critical role in ungulate immune function, bone and iodine metabolism, reproductive success, and recruitment (Flueck et al. 2012). Selenium deficiencies decrease fecundity and juvenile survival, which can undermine population stability and/or impede population recovery (Flueck 1994; Cobb 2010; Flueck et al. 2012). For example, selenium deficiencies often express in 3–6 week old fawns or calves deficient in selenium experience muscle stiffness and respiratory distress (known as white muscle disease), which can cause direct mortality and/or make afflicted young more vulnerable to predation due to lower mobility and impaired function (Flueck et al. 2012). In a northern California study, selenium enhancement increased black-tailed deer fawn survival

by 51% (Flueck 1994). However, selenium can negatively impact pregnancy success and fetus development at higher dosages (Ecometrix Inc. 2008). Human land-use changes can make selenium deficiencies in native ungulate populations more likely by restricting access to areas naturally highest in bioavailable selenium, such as valley bottoms, riparian areas, floodplains, and exposed sedimentary bedrock (Flueck et al. 2012).

Adaptive Capacity

Native ungulates were evaluated by regional experts as having moderate-high overall adaptive capacity (high confidence in evaluation).

Species group extent, integrity, connectivity, and dispersal ability

Regional experts evaluated black-tailed deer as having a high geographic extent and Roosevelt elk as having a moderate-high extent; by contrast, tule elk were evaluated as having a low-moderate geographic extent (high confidence in evaluation). Moderate overall health and functional integrity were evaluated as moderate (high confidence), with a high degree of connectivity between populations (high confidence).

Regional experts evaluated native ungulates as having a moderate-high dispersal ability (high confidence in evaluation), and barriers to dispersal were evaluated as having a low impact on the species group (high confidence). Land-use conversion, agriculture (including agricultural deer fencing), and roads/highways/trails were identified as the primary barriers to dispersal.⁸

Native ungulates were historically widespread in California and across the western U.S., but current distributions are more limited as a result of land-use changes and historical population declines (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013; CDFW 2018). However, native ungulate population numbers still fluctuate significantly in response to top-down regulation from predators (e.g., mountain lions, black bears, coyotes) and bottom-up regulation of habitat quality and carrying capacity (Forrester & Wittmer 2013; Wittmer et al. 2014). In general, land-use conversion and intensification of human land uses can alter nutritional carrying capacity (e.g., through loss of foraging areas to development) and predation risk (e.g., by altering native ungulate movement patterns); elevated mortality, particularly of adult females, can destabilize populations (Innes 2011; Forrester & Wittmer 2013).

Data are currently being gathered to better understand black-tailed deer population numbers and dynamics region-wide, but overall, deer populations are believed to be declining (Vuln. Assessment Workshop, pers. comm., 2017). Declining numbers and shifts in rutting timing to earlier in the season have been reported by tribal members in the Elk Valley area (Vuln. Assessment Workshop, pers. comm., 2017). Recent population analyses in the Mendocino National Forest indicate that predation and the interaction of food availability with predation risk are likely contributing to declines related to low survival rates of fawns, yearlings, and

⁸ No barriers were ranked as having a significant impact on this species group; barriers presented were all ranked as having only a low impact on this species group.

adults, despite continuing high reproductive rates and high deer density (Wittmer et al. 2014). However, regional deer populations near the Sacramento River appear to still be doing well (Vuln. Assessment Workshop, pers. comm., 2017).

Significant elk population declines were recorded in the late 1860s to early 1900s, likely due to habitat loss and alteration (e.g., transition from perennial grasslands to exotic annual grasslands), introduction of and competition with livestock, and elevated market hunting pressure associated with European settlement (Sommer et al. 2007). Elk are now increasing in range and number due to management interventions that have aided population recovery, including habitat conservation, herd protection, and relocation efforts (CDFW 2018). However, many areas of historic range are still not currently occupied (CDFW 2018). State-wide, population numbers are thought to be around 5,700 for tule elk and 5,000–6,000 for Roosevelt elk (CDFW 2018). The California Department of Fish and Wildlife is monitoring elk dispersal and population numbers to better understand current distribution and dispersal patterns (CDFW 2016, 2018).

While native ungulates are highly mobile, population connectivity and dispersal capacity can be affected by land-use changes and habitat fragmentation due to agriculture, development, and road and highway construction (Sommer et al. 2007; Schloss et al. 2012). Given that most herds are non-migratory, land-use changes primarily affect local movements and dispersing individuals (Vuln. Assessment Reviewers, pers. comm., 2018). In general, elk populations are less connected than those of black-tailed deer, as elk experience more dispersal limitations in response to land-use changes (Vuln. Assessment Workshop, pers. comm., 2017). For example, observations suggest that tule elk are more reluctant than deer to cross landscape barriers such as roads and fences (Vuln. Assessment Workshop, pers. comm., 2017).

Intraspecific/life history diversity

Regional experts evaluated native ungulates as having low-moderate life history diversity (high confidence in evaluation), low-moderate genetic diversity⁹ (high confidence), high behavioral plasticity (high confidence), and low-moderate phenotypic plasticity (high confidence).

Native ungulates exhibit very similar life histories and behavior across with species, although with slight differences between populations and species suggest some plasticity (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013). For example, breeding takes place from late August to mid-October for both tule and Roosevelt elk (Innes 2011), and from late September to early December for black-tailed deer, although populations in different eco-regions (e.g., coastal vs. interior) may exhibit more variable breeding timing (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008). Calves and fawns are born in the spring, with black-tailed deer fawning occurring from April to May (Sommer et al. 2007), and elk calves being born from May to June (Innes 2011). For the first few weeks following birth, both fawns and calves require adequate cover to hide from predators (Innes 2011, 2013).

⁹ Deer were evaluated as having moderate genetic diversity; elk were evaluated as having low genetic diversity (Vuln. Assessment Workshop, pers. comm., 2017).

Native ungulates are well known for exhibiting high levels of behavioral plasticity, particularly as adults. For example, in southeast Alaska, Sitka black-tailed deer (*Odocoileus hemionus sitkensis*) have been shown to alter habitat use depending on availability of preferred conditions (e.g., early seral stages) and other environmental variables (e.g., snow depth; Gilbert et al. 2017). Similarly, a study in Alberta, Canada found that while elk avoid high-use roads, they exhibited less behavioral alteration in response to low-use roads, indicating some accommodation to altered landscape conditions (Prokopenko et al. 2017). Anecdotal evidence suggests native ungulates in northern California are similarly flexible; for example, they utilize irrigated agricultural areas and livestock pastures and ponds to meet foraging and drinking water needs (Vuln. Assessment Reviewers, pers. comm., 2018). California populations are also known to display seasonal shifts in resource utilization and landscape occupancy, as well as to vacate burning areas and then return to browse on post-fire growth (Vuln. Assessment Reviewers, pers. comm., 2018).

Regional elk populations, particularly tule elk, are vulnerable to continued effects of previous genetic bottlenecks due to past population declines and the current isolation of remnant herds (CDFW 2018). Existing herds were established from a small, genetically-limited source population, and genetic exchange among current herds is low due to geographic isolation and increasing prevalence of migration barriers as a result of intensifying human land use (CDFW 2018). Limited genetic exchange can threaten the viability of remnant elk populations, leading to inbreeding depression (CDFW 2018).

Resistance and recovery

Regional experts evaluated native ungulates as having moderate-high resistance to climate stressors and natural disturbance regimes (high confidence in evaluation). Recovery potential was evaluated as moderate (moderate confidence).

Native ungulate resistance to climate change impacts is supported by their generalist habitat use and foraging behavior (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013; Wittmer et al. 2014) and high mobility (Schloss et al. 2012). These attributes increase the likelihood that native ungulates will be able to move across the landscape and utilize various resources as conditions change, as long as landscape conditions allow dispersal (Schloss et al. 2012). However, landscape-scale carrying capacity and habitat connectivity can be significantly affected by both climate change (e.g., prolonged drought, uncharacteristically large, severe wildfires) and various land-use and management activities, including timber harvesting, livestock grazing, the presence and use of roads and highways, and predator management (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013; Schloss et al. 2012; Wittmer et al. 2014).

Native ungulate population stability and growth is largely dependent on adult female survival, as they generally bear few young per year (1–2 fawns or calves; Innes 2011, 2013). Black-tailed deer experience higher and more variable fawn mortality rates than other native ungulates, but

this is balanced by high fecundity and adult female survival rates (Forrester & Wittmer 2013). Similarly, elk population recovery depends on age of first breeding (tied to physical condition) and cow survival, with pregnancy rates peaking between 3.5 and 7.5 years (Innes 2011).

Management potential

Public and societal value

Regional experts evaluated native ungulates as having high public and societal value (high confidence in evaluation).

As charismatic megafauna, native ungulates are valued for photographic and wildlife-watching opportunities (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013). They represent a critical subsistence resource for regional Native American tribes, (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013; Karuk Tribe 2019), and are also valued by non-tribal groups for hunting opportunities (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013). Hunting tag revenue supports habitat management activities that sustain native ungulate populations (Wittmer et al. 2014), and hunting organizations provide additional monetary and political support (Vuln. Assessment Workshop, pers. comm., 2017). For example, the Rocky Mountain Elk Foundation and Mule Deer Foundation provide funding for management support of these species on federal lands (Vuln. Assessment Workshop, pers. comm., 2017).

While many societal groups value native ungulates, there are some sectors that are more likely to have conflicting interests. For example, local homeowners, gardeners, and commercial agricultural entities face issues with herbivory (CDFW 2018). Commercial agriculture producers must also balance food safety regulations with wildlife conservation activities. For instance, deer intrusion along the central California coast has prevented the sale of crops due to concerns about *E. coli*, and the threat of future revenue losses has resulted in the adoption of measures designed to discourage wildlife intrusion (e.g., fences, riparian vegetation removal; Beretti & Stuart 2008; Karp et al. 2015). Increasing land-use conversion may continue to escalate such wildlife-human conflicts, which could further undermine societal support for native ungulate management (CDFW 2018).

Management capacity and ability to alleviate impacts¹⁰

Regional experts evaluated the potential for reducing climate impacts on native ungulates through management as moderate-high (high confidence in evaluation).

The scientific literature identifies many opportunities for creating early seral habitat preferred by native ungulate species, including increased use of fire (e.g., wildland and prescribed fire) and timber harvest management (Sommer et al. 2007; Hayden et al. 2008; Nelson et al. 2008; Innes 2011, 2013; Norgaard et al. 2016; Karuk Tribe 2019). The use of low- to moderate-intensity fire, in particular, has been utilized for centuries by northern California tribes to

¹⁰ Further information on climate adaptation strategies and actions for northern California can be found on the project page (<https://bit.ly/31AUGs5>).

improve habitat conditions for native ungulates and other valued resources (see Table 2; Long et al. 2016; Norgaard et al. 2016; Karuk Tribe 2019). However, all management actions must be evaluated at the landscape scale due to potential conflicts between habitat conditions preferred by other sensitive wildlife species. Some management actions favoring deer and elk may also have other unacceptable ecological consequences (e.g., elevated erosion; Sommer et al. 2007).

Table 2. Effects of prescribed burning on native ungulates across time (table adapted from Norgaard et al. 2016). Cultural burning practices, in particular, have played a role in maintaining these species on the landscape over very long time scales.

Immediate	2-Year	Long Term
<ul style="list-style-type: none"> • Promotion of nut crops, fruits, greens, and shoots that act as ungulate food sources due to the release of soil nutrients • Provision of ash and charcoal that provides opportunities for elk to reduce parasites 	<ul style="list-style-type: none"> • Protection of critical foraging habitat and cover through reduced conifer encroachment into meadows and forest openings • Increased quality of acorn crops (due to pest reduction) and browse crops 	<ul style="list-style-type: none"> • Promotion of increased ecological diversity and productivity across the landscape, benefitting native ungulates at individual, herd, and population scales
Source(s): Dasmann & Dasmann 1963; Kremsater & Bunnell 1992; Lake 2007, 2013; Norgaard et al. 2016; Karuk Tribe 2019		

Management practices can be adjusted based on emerging knowledge. For example, timber harvest practices could maintain higher remnant densities of oak trees to sustain acorn availability, a critical food resource for deer (Loomis et al. 1995). Similarly, tag allotments for black-tailed deer can be adjusted based on population data currently being collected by the California Department of Fish and Wildlife (Vuln. Assessment Workshop, pers. comm., 2017). Management activities can also be designed to address known or emerging climate-related stressors. For instance, spring restoration or improving seasonal artificial water sources may help maintain surface water availability for native ungulates during dry periods (Vuln. Assessment Reviewers, pers. comm., 2018).

Ecosystem services

Native ungulates provide a variety of ecosystem services, including:

- Provisioning of food and ornamental resources; and
- Cultural/tribal uses for spiritual/religious purposes, knowledge systems, educational values, aesthetic values, social relations, sense of place, cultural heritage, inspiration, and recreation (Anderson 2005; Long et al. 2016; Norgaard et al. 2016; Vuln. Assessment Workshop, pers. comm., 2017; Karuk Tribe 2019).

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Further information on the Northern California Climate Adaptation Project is available on the project website (<https://tinyurl.com/NorCalAdaptation>).

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Northern California Climate Adaptation Project: Vulnerability Assessment Methods and Application

Defining Terms

Exposure: A measure of how much of a change in climate or climate-driven factors a resource is likely to experience (Glick et al. 2011).

Sensitivity: A measure of whether and how a resource is likely to be affected by a given change in climate or factors driven by climate (Glick et al. 2011).

Adaptive Capacity: The ability of a resource to accommodate or cope with climate change impacts with minimal disruption (Glick et al. 2011).

Vulnerability: A function of the sensitivity of a particular resource to climate changes, its exposure to those changes, and its capacity to adapt to those changes (IPCC 2007).

Vulnerability Assessment Model

The vulnerability assessment model applied in this process was developed by EcoAdapt (EcoAdapt 2014a; EcoAdapt 2014b; Kershner 2014; Hutto et al. 2015; Gregg 2018),¹¹ and includes evaluations of relative vulnerability by local and regional stakeholders who have detailed knowledge about and/or expertise in the ecology, management, and threats to focal habitats, species groups, individual species, and the ecosystem services that these resources provide. Stakeholders evaluated vulnerability for each resource by discussing and answering a series of questions for sensitivity and adaptive capacity. Exposure was evaluated by EcoAdapt using projected future climate changes from the scientific literature. Each vulnerability component (i.e., sensitivity, adaptive capacity, and exposure) was divided into specific elements. For example, habitats included three elements for assessing sensitivity and six elements for adaptive capacity. Elements for each vulnerability component are described in more detail below.

In-person workshops were held in Eureka, Redding, and Upper Lake between May and October 2017. Participants self-selected habitat and species group/species breakout groups and evaluated the vulnerability of each resource. Participants were first asked to describe the habitat and/or to list the species to be considered in the evaluation of an overarching species group. Due to limitations in workshop time and participant expertise, multiple resources were not assessed during these engagements. Evaluations for remaining habitats, species groups, and species were completed by contacting resource experts.¹²

¹¹ Sensitivity and adaptive capacity elements were informed by Lawler 2010, Glick et al. 2011, and Manomet Center for Conservation Sciences 2012.

¹² Resources evaluated by experts included: coastal bluff/scrub habitats, coastal conifer hardwood forest, true fir forest, lakes/ponds, freshwater marshes, vernal pools, seeps/springs, native insect pollinators, native ungulates, salamanders, frogs, native mussels, marbled murrelet, and northwestern pond turtle.

Stakeholders assigned one of five rankings (High, Moderate-High, Moderate, Low-Moderate, or Low) for sensitivity and adaptive capacity. EcoAdapt assigned rankings for projected future climate exposure. Rankings for each component were then converted into scores (High-5, Moderate-High-4, Moderate-3, Low-Moderate-2, or Low-1), and the scores were averaged (mean) to generate an overall score. For example, scores for each element of habitat sensitivity were averaged to generate an overall habitat sensitivity score. Scores for exposure were weighted less than scores for sensitivity and adaptive capacity because the uncertainty about the magnitude and rate of future change is greater. Sensitivity, adaptive capacity, and exposure scores were combined into an overall vulnerability score calculated as:

$$\text{Vulnerability} = [(\text{Climate Exposure} * 0.5) \times \text{Sensitivity}] - \text{Adaptive Capacity}$$

Elements for each component of vulnerability were also assigned one of three confidence rankings (High, Moderate, or Low). Confidence rankings were converted into scores (High-3, Moderate-2, or Low-1) and the scores averaged (mean) to generate an overall confidence score. These approximate confidence levels were based on the Manomet Center for Conservation Sciences (2012) 3-category scale, which collapsed the 5-category scale developed by Moss and Schneider (2000) for the IPCC Third Assessment Report. The vulnerability assessment model applied here assesses the confidence associated with individual element rankings and, from these rankings, estimates the overall level of confidence for each component of vulnerability and then for overall vulnerability.

Stakeholders and decision-makers can consider the rankings and scores presented as measures of relative vulnerability and confidence to compare the level of vulnerability among the focal resources evaluated in this project. Elements that received lower confidence rankings indicate knowledge gaps that applied scientific research could help address.

Vulnerability Assessment Model Elements

Sensitivity & Exposure (Applies to Habitats, Species Groups, Species)

- **Climate and Climate-Driven Factors:** e.g., air temperature, precipitation, freshwater temperature, soil moisture, snowpack, extreme events: drought, altered streamflows, etc.
- **Disturbance Regimes:** e.g., wildfire, flooding, drought, insect and disease outbreaks, wind
- **Future Climate Exposure:** e.g., consideration of projected future climate changes (e.g., temperature and precipitation) as well as climate-driven changes (e.g., altered fire regimes, altered water flow regimes, shifts in vegetation types)
- **Stressors Not Related to Climate:** e.g., tectonic and volcanic events; residential or commercial development; agriculture and/or aquaculture; roads, highways, trails; dams and water diversions; invasive and other problematic species; livestock grazing; fire suppression; timber harvest; mining; etc.

Sensitivity & Exposure (Applies to Species Groups and Species)

- **Dependencies:** e.g., dependencies on sensitive habitats, specific prey or forage species, and the timing of the appearance of these prey and forage species (concern for mismatch)

Sensitivity & Exposure (Applies to Species ONLY)

- **Life History:** e.g., species reproductive strategy, average length of time to reproductive maturity

Adaptive Capacity (Applies to Habitats, Species Groups, Species)

- **Extent, Integrity, and Continuity/Connectivity:** e.g., resources that are widespread vs. limited, structural and functional integrity (e.g., degraded or pristine) of a habitat or health and functional integrity of species (e.g., endangered), isolated vs. continuous distribution
- **Landscape Permeability:** e.g., barriers to dispersal and/or continuity (e.g., land-use conversion, energy production, roads, timber harvest, etc.)
- **Resistance and Recovery:** e.g., *resistance* refers to the stasis of a resource in the face of change, *recovery* refers to the ability to “bounce back” more quickly from the impact of stressors once they occur
- **Management Potential:** e.g., ability to alter the adaptive capacity and resilience of a resource to climatic and non-climate stressors (societal value, ability to alleviate impacts, capacity to cope with impacts)
- **Ecosystem Services:** e.g., provisioning, regulating, supporting, and/or cultural services that a resource produces for human well-being

Adaptive Capacity (Applies to Habitats ONLY)

- **Habitat Diversity:** e.g., diversity of physical/topographical characteristics, component native species and functional groups

Adaptive Capacity (Applies to Species Groups, Species)

- **Dispersal Ability:** i.e., ability of a species to shift its distribution across the landscape as the climate changes
- **Intraspecific/Life History Diversity:** e.g., life history diversity, genetic diversity, phenotypic and behavioral plasticity

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