Wet Meadows and Fens

Northern California Climate Change Vulnerability Assessment Synthesis

An Important Note About this Document: *This document represents an initial evaluation of vulnerability for wet meadows and fens 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 Nina Hemphill (U.S. Forest Service), Julie Nelson (U.S. Forest Service), and Karen Pope (U.S. Forest Service). Vulnerability scores were provided by Redding workshop participants. Upper Lake workshop participants provided additional comments on the climate change vulnerability of this habitat.

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Habitat Description

Wet meadows and fens comprise a diverse group of groundwater-dependent ecosystems, with wet meadows dominated primarily by herbaceous plant communities that grow on finely textured soils (Fites-Kaufmann et al. 2007; Weixelman et al. 2011; Viers et al. 2013) and fens characterized by a thick peat substrate that supports many distinctive plant species (Chimner & Cooper 2003; Sikes et al. 2012, 2013). In northern California, wet meadows and fens typically occur at 1,200–2,396 m [3,940–7,861 ft] in the southern Cascades (Weixelman et al. 2011; Sikes et al. 2013; Cole & North 2014), and from 1,400–1,950 m (4,600–6,400 ft) in the Klamath Mountains (Sikes et al. 2013). However, they can also occur in lower-elevation montane valleys and depressions in the Klamath Mountains and North Coast Range (Sikes et al. 2012, 2013; Wolf & Cooper 2015). The coastal Inglenook Fen is believed to be the only sea level fen in California (Erman et al. 1977).

Wet meadows are located in areas that confine or slow the release of groundwater over the landscape (Weixelman et al. 2011; Viers et al. 2013). Wet meadows can range from open, herbaceous-dominated areas to densely packed riparian shrubfields, with no two meadows identical in plant composition or hydrologic regime (Viers et al. 2013). Hydrologic factors that determine wet meadow plant composition and function include the presence or absence of surface water, whether water is flowing or standing, and the duration of inundation (Allen-Diaz 1991; Darrouzet-Nardi et al. 2006). A persistently high water table often limits the presence of tree and shrub species in wet meadows, instead favoring hydric herbaceous species (Viers et al. 2013). Wet meadow vegetation usually comprises a mixture of sedges (*Eleocharis* spp., *Carex* spp.), rushes (*Juncus* spp.), and hydric meadow grasses (e.g., tufted hairgrass [*Deschampsia cespitosa*]; Chambers & Miller 2011; Weixelman et al. 2011; Cole & North 2014). Riparian thickets may form alongside flowing water in meadows (Viers et al. 2013), often including willows (*Salix* spp.) and alders (*Alnus* spp.; Chambers & Miller 2011; Weixelman et al. 2011; Cole & North 2014). Dogwoods (*Cornus* spp.) are also important in shrubby areas as are heather (*Ericaceous* spp.) and *Spiraea* species (Vuln. Assessment Reviewer, pers. comm., 2018).

Fens, which may occur within wet meadows or adjacent to lakes or ponds, are usually small (less than 1 ha [2.5 ac] in size in California) and contain surface water for a significant portion of the year (Sikes et al. 2012, 2013). By definition, fens in California must contain a layer of peat at least 40 cm (15.7 in) thick, which can take up to 2,000 years to develop because plant matter decomposes very slowly in a standing water, oxygen-poor environment (Sikes et al. 2012, 2013). Fens support mosses, herbaceous vascular species, and woody vegetation, with plants deriving most or all of their water and nutrients from the peat layer rather than from underlying mineral soils (Chimner & Cooper 2003; Sikes et al. 2012, 2013). In the Klamath and North Coast Ranges, *Ptychostomum pacificum* is a characteristic moss of fens in coniferous forests (Spence & Shevock 2012). Within serpentine and other nutrient-poor landscapes, fens fed by cool water are dominated by the insectivorous California pitcher plant (*Darlingtonia californica*), which is restricted to northern California and southwestern Oregon (Sikes et al. 2013; CNPS 2019). The distinct vegetation of *Darlingtonia* fens is composed mainly of perennials, which may include California bog asphodel (*Narthecium californicum*), great burnet (*Sanguisorba officinalis*), California cornflower (*Rudbeckia californica*), and other herbaceous plants (CNPS 2019). Several trees and shrubs also grow in *Darlingtonia* fens, including Port-Orford-cedar (*Chamaecyparis lawsoniana*), western azalea (*Rhododendron occidentale*), and Labrador tea (*Rhododendron columbianum*; CNPS 2019).

Wet meadows and fens are highly valued by northern California tribes, providing food, fiber, medicines, and fresh water (Anderson 2005; Norgaard et al. 2016; Karuk Tribe 2019). These habitats harbor culturally-valued plant species such as trailing blackberry (*Rubus ursinus*), Mariposa lily (*Calochortus* spp.), panther lily (*Lilium pardalinum*), wild turnip (*Brassica rapa*), and multiple species of Indian potatoes (e.g., *Brodiaea coronaria*; Kimmerer & Lake 2001; Norgaard et al. 2016). Additionally, the rare leopard lily (*Lilium pardalinum wigginsii*), found in high-elevation serpentine wet meadows, is among the most valued bulbs in the Karuk diet (Schenck & Gifford 1952; Norgaard et al. 2016; Karuk Tribe 2019). Wet meadows and fens are also utilized by many wildlife species, including black bear (*Urus americanus*), Roosevelt elk (*Cervus canadensis roosevelti*), and black-tailed deer (*Odocoileus hemionus*; Norgaard et al. 2016; Karuk Tribe 2019).

Executive Summary

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

Wet Meadows and Fens	**Rank**	**Confidence**
Sensitivity	Moderate-High	High
Future Exposure	Moderate-High	Moderate
Adaptive Capacity	Moderate	High
Vulnerability	**Moderate-High**	**High**

Sensitivity & Exposure Summary	*Climate and climate-driven factors*: • Snowpack amount, timing of snowmelt and runoff, drought, precipitation amount and timing, soil moisture, streamflow, water temperature, storms *Disturbance regimes*: • Flooding, wildfire *Non-climate stressors*: • Agriculture, dams and water diversions, fire suppression, roads and trails, recreation, mining, livestock grazing, invasive and other problematic species

Wet meadows and fens are primarily sensitive to climate stressors that alter the hydrologic regime, including snowpack amount, timing of snowmelt and runoff, precipitation amount and timing, streamflow, and drought. Changes in these factors may result in a lowered groundwater table and reduced water retention, causing plant mortality and reduced habitat extent. This is particularly true for systems that have already been impacted by anthropogenic stressors that alter wetland structure and hydrology (e.g., dams, livestock grazing, roads/trails, and others). Degraded wet meadows and fens also exhibit reduced resistance to disturbance events such as flooding and wildfire, which may further exacerbate structural changes and drying. Fire suppression, coupled with disturbances that dewater meadows, can promote shade tree encroachment, lower water tables, and increase the risk of high-severity fire in wet meadow and fen habitat. Degraded and/or disturbed wet meadows and fens may experience draining and lowering of the groundwater table following structural changes (e.g., channelization and bank erosion), potentially initiating a transition to upland habitat types.

Adaptive Capacity Summary	*Factors that enhance adaptive capacity*: + Located in diverse habitat niches across a wide elevational range + High cultural and recreational interest in montane and subalpine wet meadows + Generally responsive to management efforts focused on restoring hydrology in degraded systems + Federal Clean Water Act supports protection and restoration initiatives *Factors that undermine adaptive capacity:* – Small areal extent of fens and wet meadows based on limiting physical, chemical, and biological characteristics – Further reductions in areal extent and distribution have occurred, largely due to anthropogenic stressors – Little or no capacity for system migration given specific hydrologic requirements

Wet meadows and fens are widely distributed in northwestern California, although they comprise a very small areal extent of the landscape and have been significantly reduced from their historical extent by anthropogenic stressors. These wetlands are rich in biodiversity, reflecting the variable topography, hydrology, and water and soil chemistry of the region. Component plant species in wet meadows and fens range from widely-distributed species to narrow endemics, and generally high dispersal ability in many of these adds to the capacity of this habitat to accommodate and recover from disturbances. However, migration opportunities in the face of climate change are limited because these habitats require specific hydrological and landscape conditions to exist (e.g., high groundwater tables, sediment deposition, peat formation). The scientific literature suggests multiple management actions that may build adaptive capacity in wet meadows and fens by maintaining or restoring structural and hydrologic integrity. These include prescribed fire to reduce woody species incursion, beaver (*Castor canadensis*) reintroduction to raise water table levels, improved grazing management, selective removal of invasive species, removal of constrictions to flow patterns, and restoration of eroded areas. Building adaptive capacity in wet meadows and fens is especially important as these habitats support groundwater storage and release, contribute to surface water availability, and support many culturally and ecologically important species.

Sensitivity and Exposure

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

Climate change is projected to reduce the extent of wet meadows and fens, primarily due to increased air and water temperatures, reduced snowpack, earlier timing of spring runoff, and increased drought (Crimmins et al. 2011; Sikes et al. 2013; Ryan et al. 2014; Karran et al. 2017). In general, wet meadows and fens are restricted by hydro-geomorphic characteristics rather than climatic conditions, so climate modeling efforts designed to demonstrate potential shifts in vegetation are largely unsuited to this habitat type (Vuln. Assessment Reviewer, pers. comm., 2019). For instance, climate conditions may result in altered climate suitability for wet meadow and fen vegetation (Thorne et al. 2016). However, the formation of new systems would also require appropriate hydrologic conditions (e.g., localized groundwater-confining topography), which may not occur in all areas that are otherwise suitable for vegetation (Crimmins et al. 2011).

Sensitivity and future exposure to climate and climate-driven factors

Regional experts evaluated wet meadows and fens as having high 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 wet meadows and fens include snowpack amount, timing of snowmelt and runoff, drought, precipitation amount and timing, soil moisture, streamflow, water temperature, and storms.[1]

Snowpack amount and timing of snowmelt and runoff

In northern California, montane wet meadows receive their primary water supply from melting snowpack that provides surface water and recharges groundwater, elevating soil moisture and increasing stream baseflow during the growing season (Sikes et al. 2013; Viers et al. 2013). Gradual melting of the snowpack maintains a consistently high groundwater table (typically <1 m [3.3 ft] below the land surface; Loheide & Gorelick 2007), which is necessary to sustain wet meadow vegetation (Chambers & Miller 2011; Viers et al. 2013). Reduced snowpack decreases groundwater recharge and surface water inflow to wet meadows and fens (Chambers & Miller 2011; Viers et al. 2013; Hunt et al. 2018), potentially leading to habitat contraction or loss (Chambers & Miller 2011; Viers et al. 2013). Additionally, a greater proportion of winter precipitation falling as rain rather than snow (Knowles et al. 2006) would likely result in more immediate runoff and less water stored for gradual release during drier months (Viers et al. 2013; Long & Pope 2014). Large increases in precipitation, particularly winter rainfall, could also negatively impact wet meadows and fens due to stream scour and flooding (Viers et al. 2013; Long & Pope 2014).

Site-specific hydrologic factors may limit loss of these wetland types in response to changing snowpack regimes (Sikes et al. 2013; Drexler et al. 2013). A study found that fens in the southern Cascades were less affected by periods of high temperatures and low snowpack compared to those in the Sierra Nevada, which decreased in extent by 10–16% (Drexler et al. 2013). However, it is unknown whether the southern Cascade fens experienced less extreme changes in conditions or whether local hydrological conditions had a buffering effect (Drexler et al. 2013). In general, smaller wet meadows and fens and those that are ditched or highly eroded are likely to be more affected by decreased snowpack (Sikes et al. 2013; Drexler et al. 2013).

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[1] All climate and climate-driven factors presented were ranked as having a moderate or higher impact on this habitat type.

Regional Snowpack & Snowmelt Trends[2]	
Historical & current trends: • 15–39% decrease in April 1 snow water equivalent (SWE) between 1951 and 2010 for the Northwestern California and Southern Cascade ecoregions (Flint et al. 2013) • 15–40-day shift towards earlier date of 90% snowmelt across the western U.S. since 1915 (Hamlet et al. 2005) • 10–30-day shift towards earlier timing of snowmelt-driven runoff across the western U.S. since 1948 (Stewart et al. 2005)	*Projected future trends:* • Decreases in April 1 SWE by 2100 (compared to 1951–1980; Flint et al. 2013; Flint & Flint 2014): ○ 86–99% decrease on the North Coast ○ 82–99% decrease in the Northern Coast Range ○ 99–100% decrease in the Northern Interior Coast Range ○ 72–94% decrease in the Klamath Mountains ○ 61–89% decrease in the Southern Cascades • Likely 5–15-day shift towards earlier timing of snowmelt-driven runoff in northern California by 2100 (up to 60-day shift across the western U.S.; Stewart et al. 2004; Rauscher et al. 2008)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Reduced groundwater recharge and growing season soil moisture availability could hinder survival and growth of plant species dependent on a high water table • Possible reduction in wet meadow and fen extent due to reduced surface water and groundwater • Degradation of springs due to loss of groundwater recharge from declining snowpack	

Drought

In northern California, higher temperatures, reduced snowpack, and decreased summer precipitation are likely to lead to more severe summer drought conditions for wet meadows and fens (Cayan et al. 2008; Diffenbaugh et al. 2015; Thorne et al. 2015). Wet meadow plants stressed by drought may go dormant or die, causing a degradation of the network of fibrous root systems that prevent bank erosion and channel incision (Purdy & Moyle 2006; Patterson & Cooper 2007; Viers et al. 2013). In drought-stressed or otherwise degraded meadows, channel incision allows water to drain away from the meadow more quickly (Patterson & Cooper 2007; Viers et al. 2013). This process reduces water residence time within the meadow, decreasing groundwater recharge and, in turn, causing further drying (Null et al. 2010; Sikes et al. 2013; Viers et al. 2013) and potentially allowing the establishment of upland plants (Purdy & Moyle 2006; Patterson & Cooper 2007; Viers et al. 2013). In this way, drought can alter the morphology of wet meadows, decreasing water storage capacity and potentially leading to reduced habitat extent or conversion to drier habitat types following tree and shrub encroachment (Null et al. 2010; Sikes et al. 2013; Viers et al. 2013; Vuln. Assessment Workshop,

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[2] Trends in climate factors and natural disturbance regimes presented in this and subsequent summary tables are not habitat-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.

pers. comm., 2017). In fens, the drying and oxidizing of peat under extreme drought can similarly increase vulnerability to erosion (Fites-Kaufmann et al. 2007) or compromise ecosystem functioning (Ryan et al. 2014).

Some wet meadow and fen plants have characteristics that allow them to tolerate periods of drought, including roots that reach deep into the groundwater table and/or the capacity to reproduce by rhizomes rather than seeds (Fites-Kaufmann et al. 2007). In the Sierra Nevada, drought-tolerant characteristics allowed sedge and rush species in a subalpine wet meadow to maintain community dominance during an 8-year period of drought, as well as to quickly recover their pre-drought biomass with the return of precipitation (Rejmánková et al. 1999). However, very severe drought conditions can push wetland plants beyond their physiological limits, potentially initiating transformative hydrologic change in the system (Stillwater Sciences 2012; Sikes et al. 2013; Viers et al. 2013).

Regional Drought Trends	
Historical & current trends: • 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)	*Projected future trends:* • 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 Habitat *(see text for citations)*	
• Possible dormancy or death of wet meadow plants, causing degradation of the fibrous root system that supports streambanks, leading to channel incision and erosion • Tree and shrub encroachment during prolonged periods of drought • Compromised ecosystem functioning and/or possible loss of fens following drying and oxidizing of peat	

<u>Precipitation amount and timing and soil moisture</u>

Precipitation in the form of rain provides direct water inputs for wet meadows and fens and also feeds surface runoff that provides additional water supply (Weixelman et al. 2011; Viers et al. 2013). Decreased summer precipitation, in combination with reduced snowpack and increased evapotranspiration, are likely to cause declines in soil moisture and lower groundwater tables, leading to shorter hydroperiods and corresponding increases in water stress (Tague et al. 2009; Perry et al. 2012). Reduced water availability is likely to increase erosion as densely rooted wet meadow plants die or go dormant and are replaced by upland species whose root systems have less capacity to stabilize soils and streambanks during periods of high streamflow (Purdy & Moyle 2006; Patterson & Cooper 2007; Viers et al. 2013). Reduced precipitation may also lead to the desiccation of peat deposits in fens, leaving them vulnerable to erosion from overland and stream flows and potentially reducing their areal extent (Sikes et al. 2013).

Reduced water availability is likely to impact wet meadow and fen flora and fauna. For example, willows would likely decrease in wet meadow riparian areas (Perry et al. 2012) and conifer and grass encroachment may occur, depending on the amount of time that wet meadows and fens are impacted by acute water stress (Tague et al. 2009; Jules et al. 2011; Weixelman et al. 2011; Cooper et al. 2012; Perry et al. 2012). Reduced precipitation is also likely to contribute to decreases in the extent of standing water within montane and subalpine wet meadows, potentially degrading or eliminating critical habitat for amphibians (McMenamin et al. 2008; Ryan et al. 2014).

Increases in precipitation could benefit hydric plant species by increasing soil moisture and reducing water stress, potentially allowing the development of hydrological conditions that could support groundwater-dependent vegetation at lower elevations in the Klamath Mountains and southern Cascades (Crimmins et al. 2011). However, new habitat creation would be dependent on the limiting characteristics of individual plant species. For example, boreal fen species would likely not establish at lower levels under increased precipitation if air temperature also increased (Chadde et al. 1998; Sikes et al. 2013).

Regional Precipitation & Soil Moisture Trends	
Historical & current trends: • 7.2–9.4 cm (2.8–3.7 in) increase in mean annual precipitation between 1900 and 2009 for the Northwestern California and Southern Cascade ecoregions (Rapacciuolo et al. 2014) • No trends available for soil moisture	*Projected future trends:* • 20% decrease to 34% increase in mean annual precipitation by 2100 (compared to 1951–1980) for the North Coast, Northern Coast Range, Northern Interior Coast Range, Klamath Mountain, and Southern Cascade ecoregions (Flint et al. 2013; Flint & Flint 2014)[3] • 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 season becomes drier and longer (Pierce et al. 2018; Swain et al. 2018) • 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)

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[3] 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

Summary of Potential Impacts on Habitat *(see text for citations)*

- Decreased soil moisture and lowered groundwater tables where precipitation decreases, resulting in shorter hydroperiods and increased water stress
  - Possible encroachment of xeric upland plant species
  - Peat desiccation and subsequent increases in erosion
  - Reduction of amphibian breeding habitat
- Possible increase in habitat in lower elevations in the southern Klamath and Cascade Ranges where groundwater regimes become able to support wet meadow and fen vegetation

Streamflow

Snowpack amount and the timing of spring runoff are primary drivers of streamflow regimes, along with geology, elevation, and stream size (Chambers & Miller 2011; Grantham et al. 2018). Higher peak flows may increase wet meadow erosion (Stewart et al. 2005; Chambers & Miller 2011; Mayer & Naman 2011; Weixelman et al. 2011), while lower summer flows associated with reduced snowpack are likely to increase drying in wet meadows and fens (Chambers & Miller 2011). As these habitats dry, bank erosion may cause further loss of riparian vegetation and a widening of the stream channel (Chambers & Miller 2011; Viers et al. 2013). Over time, the channel area can become too eroded for flows to overtop the banks during periods of peak runoff, decreasing water inputs and contributing to further drying (Viers et al. 2013). Erosion during periods of high flows can also cut into the relatively soft meadow substrate, deepening the channel and eventually lowering the water table (Viers et al. 2013). This initiates shifts in species composition from herbaceous perennials to upland vegetation, at which point most of the critical water-retaining properties of the meadow are lost and the system may convert to a drier wetland type or experience tree encroachment (Weixelman et al. 2011; Viers et al. 2013).

Regional Streamflow Trends

Historical & current trends:	*Projected future trends:*
• Shift towards earlier spring peak flows in snowmelt-dominated basins (Stewart et al. 2005; Pierce et al. 2018) • In rain-dominated coastal rivers in northern California, minimum annual flows have decreased and late summer recession rates have increased over the past 40-80 years (Sawaske & Freyberg 2014; Asarian & Walker 2016; Klein et al. 2017) • September streamflow declined at 73% of undammed sites in northern California and southwest Oregon (Asarian & Walker 2016)	• Generally, wet season flows are projected to increase and dry season flows are projected to decrease (Leng et al. 2016; Grantham et al. 2018) • Overall increase in flow variability and earlier timing of spring peak flows (by up to 30 days; Stewart et al. 2005) • As a result of more extreme dry conditions, the lowest streamflow per decade is projected to be 30–40% lower by 2100 (Pierce et al. 2018)

Summary of Potential Impacts on Habitat *(see text for citations)*

- Increased drying where summer flows decline, particularly if surface water inputs are not compensated by groundwater sources

Regional Streamflow Trends
• Increased erosion due to higher spring peak flows, potentially leading to a lowered groundwater table and/or conversion to upland vegetation

Water temperature

Earlier timing of runoff and longer duration of low flows may result in more days with high water temperatures, which can be detrimental to aquatic fauna associated with wet meadows and fens (Kupferberg et al. 2009; Viers et al. 2013). For example, warmer water temperatures could impact macroinvertebrate emergence (Harper & Peckarsky 2006; Viers et al. 2013) and fish and macroinvertebrate community composition (Marchetti & Moyle 2001; Viers et al. 2013). Warmer water temperatures may also affect amphibian survival, recruitment and growth; for instance, warmer water temperatures may lengthen the growth period in tadpoles even as adults experience more days when they are exposed to thermal stress (Kupferberg 1996; Viers et al. 2013). Increased thermal stress, in combination with decreased water availability, may contribute to further population declines in the endemic foothill yellow-legged frog (*Rana boylii*; Kupferberg 1996; Viers et al. 2013).

<table>
<tr><th colspan="2">Regional Water Temperature Trends</th></tr>
<tr><td><i>Historical & current trends:</i>
• ~0.1°C (0.2°F) per decade increase in mean August stream temperatures in northwestern California from 1976–2015 (Isaak et al. 2017)
  ○ Corresponds to a 0.4°C (0.7°F) increase in air temperature and 5.3% decrease in discharge per decade</td><td><i>Projected future trends:</i>
• 0.4–0.8°C (0.7–1.4°F) per decade increase in mean August stream temperatures in northwestern California by the 2080s (Isaak et al. 2017)
  ○ Corresponds to a 3.6°C (6.5°F) increase in air temperature and 1.2% decrease in stream discharge</td></tr>
<tr><th colspan="2">Summary of Potential Impacts on Habitat <i>(see text for citations)</i></th></tr>
<tr><td colspan="2">• Changes in the distribution, abundance, and health of aquatic organisms
• Mixed impacts on amphibians (e.g., may lengthen growth period of tadpoles but may increase thermal stress on adults and reduce suitable habitat)</td></tr>
</table>

Storms

More frequent and/or intense storm events are likely to lead to increased erosion and flooding in montane wet meadows, fens, and associated riparian areas, particularly when combined with higher and flashier winter and spring runoff events (Butz et al. 2015). Runoff associated with extreme precipitation can degrade wet meadows and fens through channel incision, downcutting, and erosion of moist peat and topsoil, leading to drying if enough water drains out of the habitat and is not replaced (Micheli & Kirchner 2002; Weixelman et al. 2011; Viers et al. 2013; Long & Pope 2014). Increased erosion and downcutting of streambanks could incite a transition to drier upland vegetation (Viers et al. 2013; Long & Pope 2014). However, more frequent storms could also have some benefits for wet meadows and fens by increasing water availability (Vuln. Assessment Workshop, pers. comm., 2017).

<table>
<tr><th colspan="2">Regional Storm Trends</th></tr>
<tr><td>Historical & current trends:
• Decline in the frequency of extreme two-day precipitation events between 1950 and 2009, with a slight decrease in the amount of precipitation received during extreme two-day events (Mass et al. 2010)</td><td>Projected future trends:
• Increase in storm intensity and duration, resulting in greater maximum precipitation rates and volume (Dettinger 2011; Shields & Kiehl 2016; Prein et al. 2017)
• Slight to moderate increase in storm frequency (up to 30% increase in atmospheric river days, or ~2.5 days per year; Dettinger 2011)
• Projected statewide increases in daily extreme precipitation values of 5–20% by 2100 (Pierce et al. 2018)</td></tr>
<tr><th colspan="2">Summary of Potential Impacts on Habitat (see text for citations)</th></tr>
<tr><td colspan="2">• Increased flooding, particularly when combined with higher and flashier winter/spring runoff
• Increased erosion and possible channel incision, resulting in habitat degradation or loss</td></tr>
</table>

Sensitivity and future exposure to changes in disturbance regimes

Regional experts evaluated wet meadows and fens as having moderate-high sensitivity to changes in disturbance regimes (high confidence in evaluation), with an overall moderate-high future exposure to these stressors within the study region (moderate confidence). Key disturbance regimes that affect wet meadows and fens include flooding and wildfire.[4]

Flooding

Periodic flooding in wet meadows and fens is necessary to maintain healthy hydrologic system functioning (Hammersmark et al. 2008, 2010; Sikes et al. 2013). The movement of water through wet meadows is supported by a well-developed network of fibrous plant roots that build bank stability (Loheide & Gorelick 2007; Viers et al. 2013). Stable stream banks allow water to overtop banks during moderate flooding events while providing a barrier to the rapid outflow of water, supporting groundwater recharge and surface water retention that maintains meadow vegetation (Loheide & Gorelick 2007; Viers et al. 2013). Stable streambanks limit more extreme flooding that damage vegetation and erode soil or peat (Micheli & Kirchner 2002; Purdy & Moyle 2006; Patterson & Cooper 2007; Viers et al. 2013).

Changes outside of the historical range of variability for snowmelt timing and precipitation/runoff volume may increase flooding impacts in wet meadows and fens beyond the level at which these systems can maintain sound hydrologic regimes (Fites-Kaufmann et al. 2007; Weixelman et al. 2011; Viers et al. 2013). Rain-on-snow precipitation can result in extreme floods that wash out or degrade meadows and fens and exacerbate stream incision and downcutting (Micheli & Kirchner 2002; Weixelman et al. 2011; Viers et al. 2013; Karran et al. 2017). Flash floods carrying heavy sediment loads and debris further erode degraded stream

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[4] All disturbance regimes presented were ranked as having a moderate or higher impact on this habitat type.

channels within wet meadows, drastically increasing channel incision during single events; this can result in a positive feedback loop that decreases the ecological integrity of the meadow and may decrease habitat extent (Herbst & Cooper 2010; Weixelman et al. 2011; Sikes et al. 2013). Where increased flooding occurs, willows may expand from riparian areas on stream edges farther into wet meadows, potentially limiting light availability to herbaceous vegetation and increasing shrub dominance (Weixelman et al. 2011).

Regional Flooding Trends	
Historical & current trends: • No trends available for flooding	*Projected future trends:* • More frequent/severe winter flooding due to an increase in extreme precipitation events (Dettinger 2011; AghaKouchak et al. 2018; Swain et al. 2018; Grantham et al. 2018) • State-wide, 200-year floods are expected to increase in frequency by 300–400%, becoming 50-year floods (Swain et al. 2018)
Summary of Potential Impacts on Habitat *(see text for citations)*	
• Increased erosion, stream incision, and downcutting, draining water away from habitat • Habitat degradation following increased sediment deposition from flooding and debris flows • Possible increases in willow recruitment, increasing shrub cover in wet meadows • Partial or total loss of habitat depending on extent of alteration of hydrologic regime and habitat integrity	

Wildfire

Wetlands and riparian areas are generally characterized by cooler air temperatures and higher relative humidity compared to adjacent uplands (Dwire & Kauffman 2003). These conditions generally limit fire intensity and severity due to higher fuel and soil moisture content (Dwire & Kauffman 2003). Additionally, wind speeds in montane riparian areas are often lower than in surrounding uplands and along ridgelines, which results in less extreme fire behavior, including decreased intensity and rate of spread (Dwire & Kauffman 2003). Wet meadows, fens, and associated riparian areas may serve as fire breaks in upland forests until late in the fire season (Dwire & Kauffman 2003). Overall, subalpine wet meadows and fens have a shorter fire season than lower-elevation habitats due to the presence of snowpack late into the season (Skinner 2003; Skinner et al. 2006; Fites-Kaufmann et al. 2007; Gergel et al. 2017).

Fires in northern California wet meadows and fens have historically been a result of both human and natural ignitions (Dwire & Kauffman 2003; Turner et al. 2011; Lake & Long 2014; Norgaard et al. 2016; Karuk Tribe 2019). Relatively frequent low- to moderate-intensity fire kills encroaching conifer seedlings and small trees that would otherwise lower water tables and lead to overall drying (Lake 2007; Norgaard et al. 2016). Fire at the meadow/forest interface can sometimes result in wet meadow expansion into burned areas formerly occupied by forest (Ratliff 1985). Cultural and prescribed burning can also maintain populations of tribally-valued plants such as leopard lily and Indian potatoes, and can improve forage for black-tailed deer and other native ungulates (Lake 2007; Norgaard et al. 2016). Wet meadows and fens are more sensitive to the negative impacts of high-intensity fires (Norgaard et al. 2016; Long & Davis 2016), particularly if they occur in adjacent riparian areas and/or burn a large proportion of the catchment area (Minshall 2003). The most significant concern is generally changes in soil structure and the loss of upland and riparian vegetation within the watershed that contribute to erosion, debris flows, and associated channel incision (Shakesby & Doerr 2006; Cannon & DeGraff 2009; Long & Davis 2016). High-intensity fires that spread into wet meadows and fens from surrounding forests can cause high rates of plant mortality (Norgaard et al. 2016), though this can also include the removal of larger encroaching conifers (Vuln. Assessment Reviewer, pers. comm., 2019).

Wet meadows and fens are generally able to recover relatively rapidly from fire (Carothers & Frost 2006; Jules et al. 2011). For example, in several fens within the Six Rivers National Forest, there was no significant difference in the percent cover of herbaceous and graminoid species three years after a major fire that affected several fens (Carothers & Frost 2006), and only minor, temporary impacts on plant community composition (Jules et al. 2011). Moss layers in burned fens have also demonstrated recovery within a decade (Norgaard et al. 2016). *Darlingtonia* fens in the Klamath Range have fire-protective characteristics that can diminish the direct impact of burning on vegetation, including high soil moisture and the well-developed underground rhizome systems of many of its component species (Carothers & Frost 2006; Jules et al. 2011; Kettridge et al. 2015).

Regional Wildfire Trends

Historical & current trends:
- 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):
  - 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)

Projected future trends:
- 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)
  - 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)
  - 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

Regional Wildfire Trends

- 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)
    - 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

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)
    - Generally, these patterns are not well-represented in studies that evaluate indices of mean fire size, intensity/severity, etc.

Summary of Potential Impacts on Habitat *(see text for citations)*

- *Immediate*:
    - Removal of encroaching trees and shrubs and reduced fuel loading
    - High rates of plant mortality during severe fires that spread from surrounding forests
    - Erosion and debris flows following high-severity fire due to loss of vegetation and changes in soil structure
- *Short-term (~2-year)*:
    - Increased abundance and productivity of culturally-valued plants and improved forage for native ungulates following low- to moderate-intensity fire
    - Potential expansion at meadow edges into areas previously occupied by forest
- *Long-term*:
    - Maintenance of open meadows with a high water table
    - Continued presence and high levels of productivity in culturally-valued species

Sensitivity and current exposure to non-climate stressors

Regional experts evaluated wet meadows and fens as having 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 wet meadows and fens include agriculture, dams and water diversions, fire suppression, roads and trails, recreation, mining, livestock grazing, and invasive and other problematic species.[5]

Agriculture and dams/water diversions

Agricultural activities, water diversions, and dams (designed for irrigation and/or hydroelectric power generation) negatively affect wet meadows, fens, and associated riparian areas by decreasing water quantity and quality (Sikes et al. 2013; Viers et al. 2013). Flow alterations as a result of dams and water diversions is a fundamental cause of channel and vegetation change throughout montane riparian areas (Cooper et al. 1998; Patterson & Cooper 2007; Long & Pope

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[5] Non-climate stressors presented are those ranked as having a moderate or higher impact on this habitat type; additional non-climate stressors that may influence the habitat to a lesser degree include residential/commercial development.

2014; Asarian & Walker 2016), particularly where infrastructure is degraded (Vuln. Assessment Workshop, pers. comm., 2017). For instance, water diversions introduce channels that allow water to drain away from wet meadows and fens and raise stream temperature by lowering flow volume (Chambers et al. 2004; Patterson & Cooper 2007; Bury 2008). Ditches and water diversions can also pose a major threat to fen integrity by incising the peat body, triggering erosion and organic matter decomposition that may lower the water table below the 20 cm (7.9 in) generally required to maintain peat accumulation within the region (Chimner & Cooper 2003; Cooper et al. 2012; Sikes et al. 2013). Where large water withdrawals occur, fens may be unable to persist (Patterson & Cooper 2007). Overall, dewatering as a result of dams and water diversions can transition systems toward more xeric species and increase vulnerability to fire (Chambers et al. 2004; Patterson & Cooper 2007). Flow alterations and associated increases in water temperature also negatively affect amphibians (Kupferberg et al. 2012; Ryan et al. 2014) and other aquatic organisms.

In northwestern California, cannabis (*Cannabis sativa, C. indica*) production has a variety of negative impacts on aquatic habitats throughout forested watersheds, particularly in Humboldt, Trinity, and Mendocino Counties (Bauer et al. 2015; Carah et al. 2015; Butsic & Brenner 2016; Butsic et al. 2017). In some smaller headwater tributaries, flow diversions for cannabis cultivation may completely dewater streams and degrade springs (Bauer et al. 2015; Vuln. Assessment Reviewer, pers. comm., 2018). In larger fish-bearing streams, flow diversions result in lower and/or intermittent summer flows and resulting increases in water temperatures (Bauer et al. 2015). These water diversions in rivers and streams may increase water temperature or direct water away from wet meadows and fens (Chambers et al. 2004; Patterson & Cooper 2007; Bury 2008). Other impacts of cannabis production that may affect wet meadows include grading and burying of streams; runoff of sediment, nutrients, petroleum products, and pesticides; and illegal land clearing (Mills 2012; Bauer et al. 2015; Carah et al. 2015; Butsic & Brenner 2016).

Fire suppression

In the absence of fire, trees and woody shrubs encroach into wet meadows, lowering water tables and further accelerating shifts toward upland vegetation (see Table 1; Ratliff 1985; Norgaard et al. 2016). Populations of leopard lily, a culturally important, perennial meadow species that produces edible bulbs, have been observed to collapse as a result of overly-shaded meadow conditions in meadows affected by fire suppression (Anderson 2005).

Increased tree density and lack of periodic wildfire have increased fuel loads, increasing the risk of high-intensity fires that may spread into wet meadows and fens or contribute to erosion and debris flows in systems downstream of burned areas (Dwire & Kauffman 2003; Norgaard et al. 2016). Wet meadows and fens can also be adversely affected by fire suppression activities, including disturbance from bulldozer lines, impacts from fire retardants, and the introduction of invasive species via vehicles, equipment, and gear (Norgaard et al. 2016; Vuln. Assessment Reviewer, pers. comm., 2018). Additionally, post-fire repairs designed to prevent erosion may prevent recovery of culturally-valued plants such as the leopard lily (Norgaard et al. 2016). However, Resource Advisors are generally brought on to guide fire suppression activities away from and/or minimize damage in sensitive areas (Vuln. Assessment Reviewer, pers. comm., 2019).

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

Prior to Fire	**During Fire**	**After Fire**
• Conifer encroachment and subsequent drying in wet meadows and fens • Loss of plant diversity and certain meadow species such as the culturally-valued leopard lily • Increased risk of uncharacteristically severe fire as a result of high fuel loads, which may spread into wet meadows and fens or impact downstream meadows through erosion and debris flows	• Possible damage to wet meadow and fen vegetation during fire suppression and firefighting activity (e.g., fireline construction)	• Reduced recovery of meadow species impacted by post-fire repairs to prevent erosion
Source(s): Dwire & Kauffman 2003; Norgaard et al. 2016		

<u>Roads, trails, and recreation</u>

Roads, trails, and recreation are common stressors in wet meadows and fens, where they alter water flow and erosion/sedimentation processes, fragment habitats, and introduce pollutants and/or invasive species (Panno et al. 1999; Patterson & Cooper 2007; Cooper et al. 2012). For example, where roads permit direct vehicle access onto wet meadows and fens, tire tracks may create depressions that lead to gully formation, resulting in water draining away from the wetland (Cooper et al. 2012; Sikes et al. 2013). Beaver trapping, facilitated by road and trail access, can diminish the areal extent of riparian and wet meadow (Naiman et al. 1988; Cooper et al. 2012; Pollock et al. 2015). Road construction may result in stream flow changes that can restrict water inputs to wet meadows and fens (Burns 1972; Patterson & Cooper 2007), and poor culvert placement can cause channel erosion and ditching (Cooper et al. 2012). Soil and other organic materials become less stable on hillsides that have been excavated for road construction, resulting in large amounts of sediment and debris entering wet meadows and fens during heavy rain events or other disturbances (Snyder 2000); this sediment can disrupt peat accumulation in fens (Jones et al. 2000; Cooper et al. 2012).

Recreational hikers using wetland trails or participating in off-trail hiking can increase localized erosion and soil compaction in wet meadows and fens (Ratliff 1985). Recreational activities (e.g., hiking, off-road vehicles) can also introduce non-native species and disease (Anderson et al. 2015). For example, the movement of soil containing spores of the introduced pathogen

Phytophthora lateralis can spread the disease over long distances (Jules et al. 2002), which has resulted in high rates of Port-Orford-cedar mortality in riparian areas (Hansen et al. 2000). Roads also introduce pollutants from vehicles and the road surface via stormwater runoff (Coffin 2007). For instance, de-icing agents (e.g., magnesium chloride) applied to roads can decrease plant diversity in fens by encouraging the growth of more salt-tolerant species (Panno et al. 1999).

Mining

Fens in the Klamath Mountains are still impacted by the legacy effects of gold mining, which occurred at both large and small scales (Alpers et al. 2005). Tailing piles that remain in previously mined areas can physically cover peat, disrupt surface and groundwater flow, and alter chemical and mineral sediment influx to fens (Cooper et al. 2012). Fens become severely degraded when the peat itself has been mined for horticultural uses or to remove bog iron, leaving little or no vegetation (Cooper & MacDonald 2000; Chimner et al. 2010). Many mined fens have remained bare for decades and require active restoration (Cooper & MacDonald 2000; Chimner et al. 2010). Hard rock mining is also common in the Klamath Ranges, with indirect impacts on wet meadows and fens through contaminated runoff and road access (Vuln Assessment Reviewer, pers. comm., 2018).

Suction dredge mining, a common method of small-scale gold mining that occurred in northern California, uses a gasoline-powered vacuum to suck up river and stream bottoms, extract gold, and then release dredged materials back into the channel (Harvey & Lisle 1998; Horizon Water and Environment 2009). The practice was outlawed in January 1, 2016 (California Department of Fish and Wildlife Code §§ 5653, 5653.1, 12000, subd. [a]). However, suction dredge mining has left mercury and other toxic metals in streambed sediments that can be mobilized into aquatic systems when the streambeds are disturbed (Harvey & Lisle 1998; Horizon Water and Environment 2009), harming or killing aquatic larvae, macroinvertebrates, and fish (Griffith & Andrews 1981; Harvey & Lisle 1998).

Livestock grazing

Livestock grazing can affect wet meadow and fen function through direct physical damage, changes to watershed inputs, and impacts on vegetation growth (Sikes et al. 2013). Livestock physically disturb wet meadows and fens by walking across water-saturated ground, which creates depressions in the substrate, increases the occurrence of bare patches, and hinders peat accumulation by exposing it to oxygen (Sikes et al. 2013; Wolf & Cooper 2015). Livestock grazing can also increase erosion in wet meadows and fens by trampling peat and vegetation or creating trails that function as ditches (Wolf & Cooper 2015). Channel incision, gullying, or other hydrological modification from livestock can alter groundwater flow patterns and increase the rate of water transport away from wet meadows (Patterson & Cooper 2007; Loheide et al. 2009). Water availability in these habitats may be further reduced if groundwater is pumped to provide water for livestock (Sikes et al. 2013).

Livestock grazing can have a variety of negative impacts on wet meadow and fen vegetation. For example, grazing removes plant biomass, increases plant trampling, and can hinder plant growth through the deposition of urine and feces (Sikes et al. 2013). Grazing also significantly impacts plant species composition due to selective foraging and varying responses of plant species to herbivory and competitive interactions (Ratliff 1985; Augustine & McNaughton 1998; Sikes et al. 2013; Merriam et al. 2018). For instance, more palatable herbs and grasses may be lost (Sikes et al. 2013), and tall plants are often replaced by lower-stature species (Ratliff 1985). Grazing also tends to reduce nonvascular plants (e.g., mosses) and willow, and increase sagebrush/wormwood (*Artemisia* spp.), *Rosaceae* species, and Baltic rush (*Juncus balticus*; Fites-Kaufmann et al. 2007; Cole & North 2014). However, grazing exclusion does not necessarily restore former species composition, likely due in part to changes in hydrology and nutrient availability that favor the dominance of disturbance-adapted upland species after the removal of grazing pressure (Merriam et al. 2018).

Finally, livestock grazing increases sedimentation and can, over time, eliminate pools important for amphibian breeding and larval survival in wet meadows and fens (Vuln. Assessment Workshop, pers. comm., 2017). In combination with increasing temperatures and drought, livestock grazing is likely to reduce the amount of future habitat available to amphibians in the Klamath Mountains (Cole & North 2014; Ryan et al. 2014).

Invasive and other problematic species

Invasive plant species can alter hydrology, productivity, and nutrient availability in northern California wet meadows (Stillwater Sciences 2012). Exotic plant invasions tend to occur following erosion events, and their spread is promoted by livestock as well as vehicle and foot traffic along road and trail networks (Millar & Rundel 2016). Many invasive species have shallow root systems and are poor at stabilizing soils and preventing further erosion (Millar & Rundel 2016). Regional wet meadow and fen exposure to exotic vegetation is variable, but abundance tends to decline with increasing elevation in this region (Alexander et al. 2011; Rundel & Keeley 2016). Invasive species appear to be mostly absent from *Darlingtonia* fens in California (Carothers & Frost 2006; Tolman 2007). Canada thistle (*Cirsium arvense*) is widely present as a wetland invader at low- to mid- elevations (Vuln. Assessment Reviewer, pers. comm., 2018).

Other problematic species that impact northern California montane and subalpine wetlands include non-native predatory trout (e.g., *Oncorhynchus* spp., *Salmo* spp., *Salvelinus* spp.), which have been introduced over the last century (Knapp 2005; Welsh et al. 2006; Cole & North 2014; Ryan et al. 2014). Surveys of wet meadows, ponds, and lakes in the Klamath Range showed amphibians were generally absent in locations where predatory non-native trout are found (Knapp 2005; Welsh et al. 2006). These now occupy 95% of large mountain lakes, and have limited the habitat distributions of native frogs, salamanders, and wetland invertebrates to smaller, more ephemeral pools where trout cannot survive (Ryan et al. 2014). However, these shallower pools are at greater risk for biodiversity loss due to drying and increasing temperatures (Ryan et al. 2014). In addition, the range of the American bullfrog (*Rana catesbeiana*), an introduced species that is common in California (Elliott et al. 2009), can increase competition or predation risk for amphibians in wet meadow and other wetland habitats (Fuller 2008). American bullfrogs can also be a vector that spreads the fungal disease chytridiomycosis (caused by *Batrachochytrium dendrobatidis*) to salamanders and frogs (Huss et al. 2013; Sette et al. 2015).

Adaptive Capacity

Wet meadows and fens were evaluated by regional experts as having moderate overall adaptive capacity (high confidence in evaluation).

Habitat extent, integrity, continuity, and permeability

Regional experts evaluated wet meadows and fens as having a low-moderate geographic extent (high confidence in evaluation), moderate structural and functional integrity (high confidence), and low continuity (high confidence). Landscape permeability for wet meadows and fens was evaluated as low (high confidence). Land-use conversion, roads, dams and water diversions, geologic features, and invasive species were identified as the primary barriers to habitat continuity and dispersal across the study region.[6]

Wet meadows and fens occur in multiple locations across the state and are not restricted to a narrow elevation range (Cooper et al. 2012; Sikes et al. 2013). In the Klamath Mountains, Southern Cascades, and North Coast Range, most wet meadows and fens occur in subalpine and montane riparian terraces or seeps where the groundwater table is close to or at the ground surface (Sikes et al. 2012; Thorne et al. 2016). Wet meadows and fens are also present at lower elevations in the Klamath Mountains and North Coast Range because the greater precipitation in these ranges allows for groundwater flow sufficient to support these habitats at lower levels in the watersheds (Sikes et al. 2012, 2013; Wolf & Cooper 2015). *Darlingtonia* fen habitat is associated with high-elevation serpentine areas in the Klamath and Trinity Mountains as well as more coastal areas with serpentine and nutrient-poor substrates (Sikes et al. 2012; Vuln. Assessment Reviewer, pers. comm., 2018).

Wet meadows and fens provide valuable species refugia and habitat connectivity functions in mountain ecosystems (Moritz et al. 2013; Morelli et al. 2016; Norgaard et al. 2016). By slowing and storing water, wet meadows and fens act as an important connector between high-elevation areas with seasonal snowpack and lower elevations, replenishing lower elevation aquatic habitats and alleviating drought conditions throughout watersheds (Wolf & Cooper 2015; Morelli et al. 2016; Norgaard et al. 2016). Wet meadows can also contain permanent and temporary ponds that enhance landscape connectivity and provide refugia for amphibians (Semlitsch 2000).

Many wet meadow and fen areas have been degraded or lost to development, roads, grazing, hydrologic alteration, and/or conifer and fir encroachment associated with fire suppression (Naiman et al. 1988; Patterson & Cooper 2007; Loheide et al. 2009; Sikes et al. 2013; Norgaard et al. 2016). The elimination of beaver across the western United States during the past two centuries likely reduced wet meadow area and decreased saturation in associated riparian

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[6] All barriers presented were ranked as having a moderate or higher impact on this habitat type.

zones, altering plant cover and species composition (Naiman et al. 1988). Degraded conditions make wet meadows and fens more vulnerable to future climatic changes; for example, drought disrupts wet hydrological functioning and reduces groundwater storage, particularly in systems that are already degraded (Patterson & Cooper 2007; Halpern et al. 2010; Sikes et al. 2013; Viers et al. 2013; Ryan et al. 2014).

Given their dependence on groundwater-confining topography, there is limited ability for wet meadows and fens to shift their distribution in response to climate change (Cooper et al. 2012; Sikes et al. 2013). If precipitation increases significantly in the region, more areas at lower elevations may support wet meadows or fens as groundwater regimes change (Crimmins et al. 2011; Dettinger 2011; Sikes et al. 2013; Wolf & Cooper 2015). Wet meadow and fen plants generally have good dispersal abilities, but component species would need to disperse to locations with the right combination of topographic, geological, and hydrological conditions in order to colonize new areas (Sikes et al. 2013; Wolf & Cooper 2015).

Habitat diversity

Regional experts evaluated wet meadows and fens as having moderate-high physical and topographical diversity (moderate confidence in evaluation), high component species diversity (high confidence), and high functional diversity (low confidence).

In northern California, wet meadows and fens exist within a highly variable physical template that can be characterized by location on the landscape, hydrology (i.e., source of water input and how water enters and exits the system), and vegetation (Weixelman et al. 2011; Sikes et al. 2012, 2013; Wolf & Cooper 2015). Although wet meadows comprise a relatively small areal component of the landscape, they support a disproportionate diversity of plants (Cheng 2004; Jones 2011). For instance, a single subalpine meadow or riparian meadow complex may include shrubby areas with an herbaceous understory, saturated areas dominated by sedges and a diverse mix of forbs (e.g., little elephant's head [*Pedicularis attolens*], Mendocino gentian [*Gentiana setigera*], white marsh marigold [*Caltha leptosepala*]), and drier areas dominated by more xeric grasses and forbs (Cheng 2004).

The main types of wet meadows include depressional seasonal, lacustrine fringe, discharge slope, riparian, and subsurface (Weixelman et al. 2011). For *depressional seasonal meadows*, which form in topographic depressions within the landscape, the primary water source is precipitation, but may include overland runoff and groundwater in seasonal perched water tables. Vegetation is highly variable, but generally includes sedges, grasses, rushes, popcornflowers (*Plagiobothrys* spp.), and tarweeds (*Madia* spp.). Because they experience pronounced late-season dryness, they are often characterized by more xeric species compared to other wet meadow types (Weixelman et al. 2011). *Lacustrine fringe meadows* occur along a lake or reservoir where the primary water input is inflow from the adjacent water body. Within these meadows, fens may develop where organic matter accumulates in areas sufficiently protected from shoreline wave erosion (Weixelman et al. 2011). For *discharge slope meadows*, the primary water source is seeps, springs, or other areas where groundwater discharges at the ground surface; these may contain small areas of fens (Weixelman et al. 2011). *Riparian meadows* are associated with stream or rivers and have a discernable bed and bank morphology; water inputs include overbank flow, subsurface water from the stream channel, and groundwater from hillslopes. Riparian meadows on steeper gradients may support tall herbaceous dicots such as California corn lily (*Veratrum californicum*) and lupine (*Lupinus* spp.; Weixelman et al. 2011). Finally, *subsurface meadows* often do not contain a stream or river channel, and surface water and groundwater are the dominant water sources. They often contain tall herbaceous dicots and can also support scattered conifers (Weixelman et al. 2011).

Fen vegetation is particularly diverse in northern California, and includes many endemic and/or rare bryophytes and vascular plants as well as some more common species (Sikes et al. 2012, 2013; Spence & Shevock 2012). Widely-distributed species often reach the southernmost edge of their range in California (Sikes et al. 2012, 2013). The most biodiverse fens tend to occur at lower elevations, possibly due to a longer snow-free growing season and warmer temperatures (Wolf & Cooper 2015). Northern California has not been extensively surveyed for fens, but unique fen types likely exist in the Klamath, Mendocino, Six Rivers, and Shasta-Trinity National Forests (Sikes et al. 2012, 2013). For instance, some may have developed peat accumulations through biota such as lilies, orchids, and carnivorous plants (Sikes et al. 2012, 2013).

Watershed geology strongly influences the chemical content of source water for sloping, groundwater-dominated fens, which comprise the majority of montane and subalpine fens in northern California (Chimner et al. 2010; Lemly & Cooper 2011). Fens can range from iron-poor to extremely iron-rich based on underlying geology (Chimner et al. 2010; Cooper et al. 2012). Rich and extremely rich fens are common in watersheds with calcareous rocks (i.e., limestone or dolomite), while poor fens and acidic, geothermal fens occur where outcrops of iron pyrite or sulfur from volcanic vents oxidize to form sulfuric acid, creating naturally acidic groundwater (Lemly 2007; Chimner et al. 2010; Cooper et al. 2012).

Characteristic fauna associated with wet meadows and fens includes tailed frog (*Ascaphus truei*), Roosevelt elk (*Cervus canadensis roosevelti*), snowshoe hare (*Lepus americanus*), merlin (*Falco columbarius*), and willow flycatcher (*Empidonax traillii*; Sanders & Flett 1989; Sikes et al. 2013; Viers et al. 2013; Ryan et al. 2014). Wet meadows and fens also provide habitat for small mammals (e.g., voles, mice, shrews; Cooper et al. 2012) and insects (Holmquist et al. 2011). Northern California supports the highest diversity of subalpine, lentic-breeding amphibians in the western United States (Yarnell et al. 2010), and many high-elevation amphibians use both lakes and wet meadows for foraging and breeding (Sikes et al. 2013; Ryan et al. 2014).

California wet meadows and fens only persist in areas that have maintained appropriate hydrological conditions, and these are highly localized (Sikes et al. 2013). Many aquifers underlying wet meadows and fens in the Klamath and Sierra Nevada Mountains have existed for thousands of years (Mohr et al. 2000; Daniels et al. 2005). Glaciation approximately 14,000 years before present probably eliminated many California wet meadows and fens at elevations higher than 1800 m (6000 ft; Sikes et al. 2013). Mosses, sedges, grasses, and other plants typical of wet meadows and fens are excellent dispersers, which likely helped in recolonizing after glacial retreat (Chadde et al. 1998; Middleton et al. 2006).

Resistance and recovery

Regional experts evaluated wet meadows and fens as having low-moderate resistance to climate stressors (high confidence in evaluation). Recovery potential was evaluated as moderate (high confidence).

Wet meadows and fens vary in their ability to resist and recover from climate change impacts based on their different hydrologic regimes and disturbance tolerances (Seavy et al. 2009; Weixelman et al. 2011; Sikes et al. 2013). Persistence is largely dependent on the height of the groundwater table, which must be accessible to plants in order for them to survive (Weixelman et al. 2011). Riparian species associated with wet meadows are well-adapted to physical disturbances such as changes in streamflow and runoff volume (Seavy et al. 2009). By contrast, fens develop in low-energy, low-oxygen environments, and fen plants may be less resistant to the impacts of climate stressors and climate-driven disturbance events (e.g., extreme flooding or drought; Weixelman et al. 2011; Sikes et al. 2013). Fens and wet meadows are likely to persist with changing temperature and precipitation regimes unless a tolerance threshold is crossed beyond which the system does not retain enough water for continued wetland plant growth, either through hydrologic input change or tree and shrub encroachment (Sikes et al. 2013; Viers et al. 2013; Ryan et al. 2014). For example, hydrologically degraded wet meadows and fens, such as those impacted by water diversions and ditches, have decreased resistance to wildfire impacts (Sikes et al. 2013; Viers et al. 2013).

The physical size and location of wet meadows and fens also play a key role in their ability to accommodate and rebound from stressors (Wolf & Cooper 2015). Fens in smaller watersheds have thinner peat accumulations, most likely due to the lower volume of local aquifers that is more likely to be depleted during periodic droughts (Wolf & Cooper 2015). Fens with thinner peat layers are likely to be more vulnerable to stressors (e.g., roads, culverts, livestock grazing) that alter hydrology and increase sedimentation and/or erosion (Sikes et al. 2013), reducing the ability of fens to adapt to increasing temperatures and changing hydrologic regimes (Sikes et al. 2013; Ryan et al. 2014).

Management potential

Public and societal value

Regional experts evaluated wet meadows and fens as having high public and societal value (high confidence in evaluation).

There is a high public and societal interest in wet meadows and fens as these habitats have many cultural, spiritual, wilderness, and recreational values for regional tribes and communities (Kimmerer & Lake 2001; Weixelman et al. 2011; Cook et al. 2014; Norgaard et al. 2016; Karuk Tribe 2019). There is also increasing public awareness that wet meadows provide other important ecosystem services (Weixelman et al. 2011; Cook et al. 2014; Norgaard et al. 2016).

For example, meadows reduce peak flows, recharge groundwater, protect streambanks and shorelines, filter sediments, provide habitat for a wide variety of wildlife, and sequester carbon (Weixelman et al. 2011; Norgaard et al. 2016).

There is some regulatory support for wet meadow and fen management. For example, activities that impact development and water quality in this habitat are regulated by the Federal Clean Water Act of 1972 (33 U.S.C. §§1251-1387). However, the ability of the State of California to regulate and enforce the Clean Water Act is regionally variable (Vuln. Assessment Workshop, pers. comm., 2017). Most mid- to high-elevation wet meadows in northern California are managed by the U.S. Forest Service or National Park Service, both of which adhere to Clean Water Act standards (Vuln. Assessment Reviewer, pers. comm., 2018). The 2016 California State Water Plan supports meadow restoration as green infrastructure, and tracking modifications to ecological restoration in updates to the plan is important for protecting wet meadow and fen habitat (Vuln. Assessment Reviewer, pers. comm. 2018).

Management capacity and ability to alleviate impacts[7]

Regional experts evaluated the potential for reducing climate impacts on wet meadows and fens through management as moderate-high (moderate confidence in evaluation).

Restoration of degraded wet meadows and fens can provide a suite of benefits, including the promotion of ecosystem services and socioecological resilience (Long & Pope 2014; Norgaard et al. 2016; Karuk Tribe 2019). The scientific literature documents multiple opportunities to adjust land use practices and reduce non-climate impacts on wet meadows and fens. For instance, management activities that can restore hydrologic functioning in wet meadows and fens include redesigning ditches, culverts, and road crossings (Patterson & Cooper 2007); coordinating with stakeholders on targeted dam removals at lower-elevation sites (Cook et al. 2014; Long & Pope 2014); and the use of plug and pond techniques or the reintroduction of beaver to increase surface and groundwater storage and restore incised stream channels (Collen & Gibson 2000; Pollock et al. 2014, 2015; Karran et al. 2017; Hunt et al. 2018). In 2012, the plug and pond method (i.e., filling sections of an incised channel with earth) was used to successfully restore a wet meadow in the Sierra Nevada, resulting in increases in baseflow by at least five times and raised groundwater levels over the next several years, despite record-setting drought conditions in 2012–2015 (Hunt et al. 2018). The use of datasets that model potential fine-scale habitat changes could be used to prioritize wet meadow and fen hydrologic regime restoration in light of changing climate conditions (Ryan et al. 2014). The reintroduction of low- and moderate-intensity fire, such as occurs during prescribed and cultural burns, may also help control conifer encroachment, maintain high water tables, and potentially expand habitat area (see Table 2; Lake 2007; Cook et al. 2014; Norgaard et al. 2016).

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

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

Immediate	2-Year	Long Term
• Removal of encroaching vegetation and decreased fuel loading	• Supports plant and wildlife species that benefit from frequent low-intensity fire, including culturally-valued species	• Maintains open meadows and helps retain water tables at a level that can support native wetland vegetation
Source(s): Norgaard et al. 2016; Karuk Tribe 2019		

Other management strategies that may increase the adaptive capacity of wet meadows and fens include:

- Preventing the establishment of invasive species through early detection initiatives and weed prevention measures for motorized vehicles used for recreation and land management (Vuln. Assessment Reviewer, pers. comm., 2018). These efforts may be particularly important if and when identification of new meadows and fens brings more recreational visitors (Millar & Rundel 2016).
- Increasing oversight of water use for existing cannabis cultivation operations as well as increasing enforcement by state and local agencies to minimize illegal stream grading and land-use conversion (Bauer et al. 2015).
- Continuing to prioritize the completion of wet meadow and fen inventories in northern California in order to evaluate current conditions, track changes over time, and design management programs to address specific management issues (Sikes et al. 2013). This may include installing additional streamflow gages and devices that measure water quality and quantity to help fill data gaps in remote watersheds (Bauer et al. 2015).
- Increase multi-scale collaborations to reduce the impacts of non-climate stressors within the watershed that may affect wet meadows and fens, including livestock grazing, suction dredge mining, road construction, recreation, and energy development (Cook et al. 2014).

Ecosystem services

Wet meadows and fens provide a variety of ecosystem services, including:

- Regulation of air quality, climate/microenvironments (e.g., shade), flood/erosion control, water purification, and pollination (Loheide et al. 2009; Lowry et al. 2011);
- Support of primary production, carbon sequestration (Drexler et al. 2015), oxygen production, soil formation/retention, nutrient cycling, and water cycling (Woltemade 2000; Hammersmark et al. 2008; Vuln. Assessment Workshop, pers. comm., 2017); and
- Spiritual and religious, knowledge systems, educational values, aesthetic values, social relations, sense of place, cultural heritage, inspiration, and recreation (Norgaard et al. 2016; Vuln. Assessment Workshop, pers. comm., 2017).

Wet meadows and fens also play larger-scale roles on the landscape, including serving as records of past ecological conditions (i.e., long-term climate change and plant composition; Sikes et al. 2013), providing resource areas for species such as bear and elk that move between upper and lower elevations, mediating fire dynamics at lower elevations, and providing water storage and regulation services that minimize flooding in lower-elevation habitats and for human communities (Norgaard et al. 2016; Vuln. Assessment Workshop, pers. comm., 2017).

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

Literature Cited

AghaKouchak A, Ragno E, Love C, Moftakhari H. 2018. Projected changes in California's precipitation intensity-frequency curves. California's Fourth Climate Change Assessment. Publication Number: CCCA4-CEC-2018-005. California Energy Commission, Sacramento, CA.

Alexander JM, Kueffer C, Daehler CC, Edwards PJ, Pauchard A, Seipel T, Arévalo J, Cavieres L, Dietz H, Jakobs G. 2011. Assembly of nonnative floras along elevational gradients explained by directional ecological filtering. Proceedings of the National Academy of Sciences **108**:656–661.

Allen-Diaz BH. 1991. Water table and plant species relationships in Sierra Nevada meadows. American Midland Naturalist **126**:30–43.

Alpers CN, Hunerlach MP, May JT, Hothem RL. 2005. Mercury contamination from historical gold mining in California. USGS Fact Sheet 2005-3014. U.S. Geological Survey, Reston, VA.

Anderson LG, Rocliffe S, Haddaway NR, Dunn AM. 2015. The role of tourism and recreation in the spread of non-native species: a systematic review and meta-analysis. PLoS ONE **10**:e0140833.

Anderson MK. 2005. Tending the wild: Native American knowledge and the management of California's natural resources. University of California Press, Berkeley, CA.

Asarian JE, Walker JD. 2016. Long-term trends in streamflow and precipitation in northwest California and southwest Oregon, 1953-2012. Journal of the American Water Resources Association **52**:241–261.

Augustine DJ, McNaughton SJ. 1998. Ungulate effects on the functional species composition of plant communities: herbivore selectivity and plant tolerance. The Journal of Wildlife Management **62**:1165–1183.

Bauer S, Olson J, Cockrill A, van Hattem M, Miller L, Tauzer M, Leppig G. 2015. Impacts of surface water diversions for marijuana cultivation on aquatic habitat in four northwestern California watersheds. PLoS ONE **10**:e0120016.

Burns JW. 1972. Some effects of logging and associated road construction on northern California streams. Transactions of the American Fisheries Society **101**:1–17.

Bury RB. 2008. Low thermal tolerances of stream amphibians in the Pacific Northwest: implications for riparian and forest management. Applied Herpetology **5**:63–74.

Butsic V, Brenner JC. 2016. Cannabis (*Cannabis sativa* or *C. indica*) agriculture and the environment: a systematic, spatially-explicit survey and potential impacts. Environmental Research Letters **11**:044023.

Butsic V, Schwab B, Baumann M, Brenner JC. 2017. Inside the Emerald Triangle: modeling the placement and size of cannabis production in Humboldt County, CA USA. Ecological Economics **142**:70–80.

Butz RJ, Sawyer S, Safford H. 2015. A summary of current trends and probable future trends in climate and climate-driven processes for the Klamath National Forest and surrounding lands. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Cannon SH, DeGraff J. 2009. The increasing wildfire and post-fire debris-flow threat in western USA, and implications for consequences of climate change. Pages 177–190 in K. Sassa and P. Canuti, editors. Landslides - Disaster Risk Reduction. Springer-Verlag Berlin Heidelberg.

Carah JK et al. 2015. High time for conservation: adding the environment to the debate on marijuana liberalization. BioScience **65**:822–829.

Carothers S, Frost EJ. 2006. Post-fire monitoring of *Gentiana setigera* and *Viola lanceolata (primulifolia)* var. *occidentalis* in serpentine fens of the Six Rivers National Forest. U.S. Forest Service, Six Rivers National Forest and California Native Plant Society, North Coast Chapter, Eureka and Arcata, CA.

Cayan DR, Maurer EP, Dettinger MD, Tyree M, Hayhoe K. 2008. Climate change scenarios for the California region. Climatic Change **87**:21–42.

Chadde SW, Shelly JS, Bursik RJ, Moseley RK, Evenden AG, Mantas M, Rabe F, Heidel B. 1998. Peatlands on national forests of the northern Rocky Mountains: ecology and conservation. Gen. Tech. Rep. RMRS-GTR-11. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT.

Chambers JC, Miller DJ. 2011. Geomorphology, hydrology, and ecology of Great Basin meadow complexes - implications for management and restoration. Gen. Tech. Rep. RMRS-GTR-258. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Chambers JC, Tausch RJ, Korfmacher JL, Germanoski D, Miller JR, Jewett D. 2004. Effects of geomorphic processes and hydrologic regimes on riparian vegetation. Pages 196–231 in J. C. Chambers and J. R. Miller, editors. Great Basin riparian ecosystems: ecology, management and restoration. Island Press, Washington, D.C.

Cheng S. 2004. Forest Service research natural areas in California. Gen. Tech. Rep. PSW-GTR-188. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Chimner RA, Cooper DJ. 2003. Influence of water table levels on CO_2 emissions in a Colorado subalpine fen: an *in situ* microcosm study. Soil Biology & Biochemistry **35**:345–351.

Chimner RA, Lemly JM, Cooper DJ. 2010. Mountain fen distribution, types and restoration priorities, San Juan Mountains, Colorado, USA. Wetlands **30**:763–771.

CNPS. 2019. A manual of California vegetation, online edition. California Native Plant Society Press. Available from http://www.cnps.org/cnps/vegetation/ (accessed January 2, 2019).

Coffin AW. 2007. From roadkill to road ecology: a review of the ecological effects of roads. Journal of Transport Geography **15**:396–406.

Cole EM, North MP. 2014. Environmental influences on amphibian assemblages across subalpine wet meadows in the Klamath Mountains, California. Herpetologica **70**:135–148.

Collen P, Gibson RJ. 2000. The general ecology of beavers (*Castor* spp.), as related to their influence on stream ecosystems and riparian habitats, and the subsequent effects on fish–a review. Reviews in Fish Biology and Fisheries **10**:439–461.

Cook A, Thaler T, Griffith G, Crossett T, Perry JA. 2014. Renew Siskiyou: a road map to resilience. Model Forest Policy Program in association with the Mount Shasta Bioregional Ecology Center and the Cumberland River Compact, Sagle, ID.

Cook BI, Ault TR, Smerdon JE. 2015. Unprecedented 21st century drought risk in the American Southwest and Central Plains. Science Advances **1**:e1400082.

Cooper DJ, Chimner RA, Merritt DM. 2012. Western mountain wetlands. University of California Press, Berkeley, CA.

Cooper DJ, MacDonald LH. 2000. Restoring the vegetation of mined peatlands in the southern Rocky Mountains of Colorado, USA. Restoration Ecology **8**:103–111.

Cooper DJ, MacDonald LH, Wenger SK, Woods SW. 1998. Hydrologic restoration of a fen in Rocky Mountain National Park, Colorado, USA. Wetlands **18**:335–345.

Crimmins SM, Dobrowski SZ, Greenberg JA, Abatzoglou JT, Mynsberge AR. 2011. Changes in climatic water balance drive downhill shifts in plant species' optimum elevations. Science **331**:324–327.

Daniels ML, Anderson S, Whitlock C. 2005. Vegetation and fire history since the Late Pleistocene from the Trinity Mountains, northwestern California, USA. The Holocene **15**:1062–1071.

Darrouzet-Nardi A, D'Antonio CM, Dawson TE. 2006. Depth of water acquisition by invading shrubs and resident herbs in a Sierra Nevada meadow. Plant and Soil **285**:31–43.

Dettinger M. 2011. Climate change, atmospheric rivers, and floods in California – a multimodel analysis of storm frequency and magnitude changes. Journal of the American Water Resources Association **47**:514–523.

Diffenbaugh NS, Swain DL, Touma D. 2015. Anthropogenic warming has increased drought risk in California. Proceedings of the National Academy of Sciences **112**:3931–3936.

Drexler JZ, Fuller CC, Orlando J, Moore PE. 2015. Recent rates of carbon accumulation in montane fens of Yosemite National Park, California, USA. Arctic, Antarctic, and Alpine Research **47**:657–669.

Drexler JZ, Knifong D, Tuil J, Flint LE, Flint AL. 2013. Fens as whole-ecosystem gauges of groundwater recharge under climate change. Journal of Hydrology **481**:22–34.

Dwire KA, Kauffman JB. 2003. Fire and riparian ecosystems in landscapes of the western USA. Forest Ecology and Management **178**:61–74.

Elliott L, Gerhardt C, Davidson C. 2009. The frogs and toads of North America: a comprehensive guide to their identification, behavior, and calls. Houghton Mifflin Harcourt, New York, NY.

Erman DC, Roby KB, Eames M. 1977. Hydrological features of a California coastal fen. The Great Basin Naturalist **37**:57–66.

Fites-Kaufmann JA, Rundel P, Stephenson N. 2007. Montane and subalpine vegetation of the Sierra Nevada and Cascade ranges. Pages 456–501 in M. Barbour, T. Keeler-Wolf, and A. A. Schoenherr, editors. Terrestrial vegetation of California, 3rd edition. University of California Press, Berkeley, CA.

Flint LE, Flint AL. 2014. California Basin Characterization Model: a dataset of historical and future hydrologic response to climate change (Ver. 1.1, May 2017). U.S. Geological Survey Data Release. Available from https://doi.org/10.5066/F76T0JPB.

Flint LE, Flint AL, Thorne JH, Boynton R. 2013. Fine-scale hydrologic modeling for regional landscape applications: the California Basin Characterization Model development and performance. Ecological Processes **2**:25.

Fuller TE. 2008. The spatial ecology of the exotic bullfrog (*Rana catesbeiana*) and its relationship to the distribution of the native herpetofauna in a managed river system. Master of Science thesis. Humboldt State University, Arcata, CA.

Gergel DR, Nijssen B, Abatzoglou JT, Lettenmaier DP, Stumbaugh MR. 2017. Effects of climate change on snowpack and fire potential in the western USA. Climatic Change **141**:287–299.

Grantham TEW, Carlisle DM, McCabe GJ, Howard JK. 2018. Sensitivity of streamflow to climate change in California. Climatic Change **149**:427–441.

Griffin D, Anchukaitis KJ. 2014. How unusual is the 2012–2014 California drought? Geophysical Research Letters **41**:9017–9023.

Griffith JS, Andrews DA. 1981. Effects of a small suction dredge on fishes and aquatic invertebrates in Idaho streams. North American Journal of Fisheries Management **1**:21–28.

Halpern CB, Antos JA, Rice JM, Haugo RD, Lang NL. 2010. Tree invasion of a montane meadow complex: temporal trends, spatial patterns, and biotic interactions. Journal of Vegetation Science **21**:717–732.

Hamlet AF, Mote PW, Clark MP, Lettenmaier DP. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate **18**:4545–4561.

Hammersmark CT, Dobrowski SZ, Rains MC, Mount JF. 2010. Simulated effects of stream restoration on the distribution of wet-meadow vegetation. Restoration Ecology **18**:882–893.

Hammersmark CT, Rains MC, Mount JF. 2008. Quantifying the hydrological effects of stream restoration in a montane meadow, northern California, USA. River Research and Applications **24**:735–753.

Hansen EM, Goheen DJ, Jules ES, Ullian B. 2000. Managing Port-Orford-cedar and the introduced pathogen *Phytophthora lateralis*. Plant Disease **84**:4–14.

Harper MP, Peckarsky BL. 2006. Emergence cues of a mayfly in a high-altitude stream ecosystem: potential response to climate change. Ecological Applications **16**:612–621.

Harvey BC, Lisle TE. 1998. Effects of suction dredging on streams: a review and an evaluation strategy. Fisheries **23**:8–17.

Herbst DB, Cooper SD. 2010. Before and after the deluge: rain-on-snow flooding effects on aquatic invertebrate communities of small streams in the Sierra Nevada, California. Journal of the North American Benthological Society **29**:1354–1366.

Holmquist JG, Jones JR, Schmidt-Gengenbach J, Pierotti LF, Love JP. 2011. Terrestrial and aquatic macroinvertebrate assemblages as a function of wetland type across a mountain landscape. Arctic, Antarctic, and Alpine Research **43**:568–584.

Horizon Water and Environment. 2009. Literature review on the impacts of suction dredge mining in California. California Department of Fish and Game, Redding, CA.

Hunt LJH, Fair J, Odland M. 2018. Meadow restoration increases baseflow and groundwater storage in the Sierra Nevada Mountains of California. JAWRA Journal of the American Water Resources Association **54**:1127–1136.

Huss M, Huntley L, Vredenburg V, Johns J, Green S. 2013. Prevalence of *Batrachochytrium dendrobatidis* in 120 archived specimens of *Lithobates catesbeianus* (American bullfrog) collected in California, 1924–2007. EcoHealth **10**:339–343.

Isaak DJ et al. 2017. The NorWeST summer stream temperature model and scenarios for the western U.S.: a crowd-sourced database and new geospatial tools foster a user community and predict broad climate warming of rivers and streams. Water Resources Research **53**:9181–9205.

Jones JA, Swanson FJ, Wemple BC, Snyder KU. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology **14**:76–85.

Jones JR. 2011. Patterns of floristic diversity in wet meadows and fens of the southern Sierra Nevada, California, USA. Master of Science thesis. Colorado State University, Fort Collins, CO.

Jules ES, Ellison AM, Gotelli NJ, Lillie S, Meindl GA, Sanders NJ, Young AN. 2011. Influence of fire on a rare serpentine plant assemblage: a 5-year study of Darlingtonia fens. American Journal of Botany **98**:801–811.

Jules ES, Kauffman MJ, Ritts WD, Carroll AL. 2002. Spread of an invasive pathogen over a variable landscape: a nonnative root rot on Port Orford cedar. Ecology **83**:3167–3181.

Karran DJ, Westbrook CJ, Bedard-Haughn A. 2017. Beaver-mediated water table dynamics in a Rocky Mountain fen. Ecohydrology **11**:e1923.

Karuk Tribe. 2019. Karuk Climate Adaptation Plan. Karuk Tribe, Orleans, CA. Available from https://karuktribeclimatechangeprojects.com/climate-adaptation-plan/ (accessed August 22, 2019).

Kettridge N, Turetsky MR, Sherwood JH, Thompson DK, Miller CA, Benscoter BW, Flannigan MD, Wotton BM, Waddington JM. 2015. Moderate drop in water table increases peatland vulnerability to post-fire regime shift. Scientific Reports **5**:8063.

Keyser A, Westerling AL. 2017. Climate drives inter-annual variability in probability of high severity fire occurrence in the western United States. Environmental Research Letters **12**:065003.

Kimmerer RW, Lake FK. 2001. The role of indigenous burning in land management. Journal of Forestry **99**:36–41.

Klein RD, McKee T, Nystrom K. 2017. Shrinking streamflows in the Redwood Region. Pages 187–197 in R. B. Standiford and Y. S. Valachovic, editors. Coast Redwood Science Symposium—2016: past successes and future direction. Gen. Tech. Rep. PSW-GTR-258. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Knapp RA. 2005. Effects of nonnative fish and habitat characteristics on lentic herpetofauna in Yosemite National Park, USA. Biological Conservation **121**:265–279.

Knowles N, Dettinger MD, Cayan DR. 2006. Trends in snowfall versus rainfall in the western United States. Journal of Climate **19**:4545–4559.

Kupferberg SJ. 1996. Hydrologic and geomorphic factors affecting conservation of a river-breeding frog (*Rana boylii*). Ecological Applications **6**:1332–1344.

Kupferberg SJ, Catenazzi A, Lunde K, Lind AJ, Palen WJ. 2009. Parasitic copepod (*Lernaea cyprinacea*) outbreaks in foothill yellow-legged frogs (*Rana boylii*) linked to unusually warm summers and amphibian malformations in northern California. Copeia **2009**:529–537.

Kupferberg SJ, Palen WJ, Lind AJ, Bobzien S, Catenazzi A, Drennan J, Power ME. 2012. Effects of flow regimes altered by dams on survival, population declines, and range-wide losses of California river-breeding frogs. Conservation Biology **26**:513–524.

Lake FK. 2007. Traditional ecological knowledge to develop and maintain fire regimes in northwestern California, Klamath-Siskiyou bioregion: management and restoration of culturally significant habitats. Ph.D dissertation. Oregon State University, Corvallis, OR.

Lake FK, Long JW. 2014. Fire and tribal cultural resources. Pages 173–186 in J. W. Long, L. Quinn-Davidson, and C. N. Skinner, editors. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. Gen. Tech. Rep. PSW-GTR-247. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Law BE, Waring RH. 2015. Carbon implications of current and future effects of drought, fire and management on Pacific Northwest forests. Forest Ecology and Management **355**:4–14.

Lemly JM. 2007. Fens of Yellowstone National Park, USA: regional and local controls over plant species distribution. Master of Science thesis. Colorado State University, Fort Collins, CO.

Lemly JM, Cooper DJ. 2011. Multiscale factors control community and species distribution in mountain peatlands. Botany **89**:689–713.

Leng G, Huang M, Voisin N, Zhang X, Asrar GR, Leung LR. 2016. Emergence of new hydrologic regimes of surface water resources in the conterminous United States under future warming. Environmental Research Letters **11**:114003.

Loheide SP, Deitchman RS, Cooper DJ, Wolf EC, Hammersmark CT, Lundquist JD. 2009. A framework for understanding the hydroecology of impacted wet meadows in the Sierra Nevada and Cascade Ranges, California, USA. Hydrogeology Journal **17**:229–246.

Loheide SP, Gorelick SM. 2007. Riparian hydroecology: a coupled model of the observed interactions between groundwater flow and meadow vegetation patterning. Water Resources Research **43**:W07414.

Long JW, Davis J. 2016. Erosion and restoration of two headwater wetlands following a severe wildfire. Ecological Restoration **34**:317–332.

Long JW, Pope K. 2014. Wet meadows. Pages 341–372 in J. W. Long, L. Quinn-Davidson, and C. N. Skinner, editors. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. Gen. Tech. Rep. PSW-GTR-247. US Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Lowry CS, Loheide SP, Moore CE, Lundquist JD. 2011. Groundwater controls on vegetation composition and patterning in mountain meadows. Water Resources Research **47**:W00J11.

Mann ML, Batllori E, Moritz MA, Waller EK, Berck P, Flint AL, Flint LE, Dolfi E. 2016. Incorporating anthropogenic influences into fire probability models: effects of human activity and climate change on fire activity in California. PLoS ONE **11**:e0153589.

Marchetti MP, Moyle PB. 2001. Effects of flow regime on fish assemblages in a regulated California stream. Ecological Applications **11**:530–539.

Mass C, Skalenakis A, Warner M. 2010. Extreme precipitation over the west coast of North America: is there a trend? Journal of Hydrometeorology **12**:310–318.

Mayer TD, Naman SW. 2011. Streamflow response to climate as influenced by geology and elevation. Journal of the American Water Resources Association **47**:724–738.

McMenamin SK, Hadly EA, Wright CK. 2008. Climatic change and wetland desiccation cause amphibian decline in Yellowstone National Park. Proceedings of the National Academy of Sciences **105**:16988–16993.

Merriam KE, Markwith SH, Coppoletta M. 2018. Livestock exclusion alters plant species composition in fen meadows. Applied Vegetation Science **21**:3–11.

Micheli ER, Kirchner JW. 2002. Effects of wet meadow riparian vegetation on streambank erosion. 2. Measurements of vegetated bank strength and consequences for failure mechanics. Earth Surface Processes and Landforms **27**:687–697.

Middleton B, van Diggelen R, Jensen K. 2006. Seed dispersal in fens. Applied Vegetation Science **9**:279–284.

Millar CI, Rundel PW. 2016. Subalpine forests. Pages 579–611 in H. Mooney and E. Zavaleta, editors. Ecosystems of California. University of California Press, Oakland, CA.

Miller JD, Skinner CN, Safford HD, Knapp EE, Ramirez CM. 2012. Trends and causes of severity, size, and number of fires in northwestern California, USA. Ecological Applications **22**:184–203.

Mills E. 2012. The carbon footprint of indoor Cannabis production. Energy Policy **46**:58–67.

Minshall GW. 2003. Responses of stream benthic macroinvertebrates to fire. Forest Ecology and Management **178**:155–161.

Mohr JA, Whitlock C, Skinner CN. 2000. Postglacial vegetation and fire history, eastern Klamath Mountains, California, USA. The Holocene **10**:587–601.

Morelli TL, Daly C, Dobrowski SZ, Dulen DM, Ebersole JL, Jackson ST. 2016. Managing climate change refugia for climate adaptation. PLoS ONE **11**:e0159909.

Moritz C, Beissinger SR, Maher SP, Morelli TL. 2013. Determining landscape connectivity and climate change refugia across the Sierra Nevada. California Landscape Conservation Cooperative, Sacramento, CA.

Naiman RJ, Johnston CA, Kelley JC. 1988. Alteration of North American streams by beaver. BioScience **38**:753–762.

Norgaard KM, Vinyeta K, Hillman L, Tripp B, Lake F. 2016. Karuk Tribe climate vulnerability assessment: assessing vulnerabilities from the increased frequency of high severity fire. Karuk Tribe Department of Natural Resources, Orleans, CA.

Null SE, Viers JH, Mount JF. 2010. Hydrologic response and watershed sensitivity to climate warming in California's Sierra Nevada. PLoS ONE **5**:e9932.

Panno SV, Nuzzo VA, Cartwright K, Hensel BR, Krapac IG. 1999. Impact of urban development on the chemical composition of ground water in a fen-wetland complex. Wetlands **19**:236–245.

Parks SA, Miller C, Abatzoglou JT, Holsinger LM, Parisien M-A, Dobrowski SZ. 2016. How will climate change affect wildland fire severity in the western US? Environmental Research Letters **11**:035002.

Parks SA, Miller C, Parisien M-A, Holsinger LM, Dobrowski SZ, Abatzoglou J. 2015. Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere **6**:1–13.

Patterson L, Cooper DJ. 2007. The use of hydrologic and ecological indicators for the restoration of drainage ditches and water diversions in a mountain fen, Cascade Range, California. Wetlands **27**:290–304.

Perry LG, Andersen DC, Reynolds LV, Nelson SM, Shafroth PB. 2012. Vulnerability of riparian ecosystems to elevated CO_2 and climate change in arid and semiarid western North America. Global Change Biology **18**:821–842.

Pierce DW, Kalansky JF, Cayan DR. 2018. Climate, drought, and sea level rise scenarios for the Fourth California Climate Assessment. California’s Fourth Climate Change Assessment. Publication Number: CNRA-CEC-2018-006. California Energy Commission, Sacramento, CA.

Pollock MM, Beechie TJ, Wheaton JM, Jordan CE, Bouwes N, Weber N, Volk C. 2014. Using beaver dams to restore incised stream ecosystems. BioScience **64**:279–290.

Pollock MM, Lewallen G, Woodruff K, Jordan CE, Castro JM. 2015. The beaver restoration guidebook: working with beaver to restore streams, wetlands, and floodplains. Version 1.02. U.S. Department of the Interior, Fish and Wildlife Service, Portland, OR.

Prein AF, Liu C, Ikeda K, Trier SB, Rasmussen RM, Holland GJ, Clark MP. 2017. Increased rainfall volume from future convective storms in the US. Nature Climate Change **7**:880–884.

Purdy SE, Moyle PB. 2006. Mountain meadows of the Sierra Nevada. University of California, Center for Watershed Sciences, Davis, CA.

Rapacciuolo G et al. 2014. Beyond a warming fingerprint: individualistic biogeographic responses to heterogeneous climate change in California. Global Change Biology **20**:2841–2855.

Ratliff RD. 1985. Meadows in the Sierra Nevada of California: state of knowledge. Gen. Tech. Rep. PSW-GTR-84. U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, CA.

Rauscher SA, Pal JS, Diffenbaugh NS, Benedetti MM. 2008. Future changes in snowmelt-driven runoff timing over the western US. Geophysical Research Letters **35**:L16703.

Rejmánková E, Rejmánek M, Djohan T, Goldman CR. 1999. Resistance and resilience of subalpine wetlands with respect to prolonged drought. Folia Geobotanica **34**:175–188.

Rundel PW, Keeley JE. 2016. Dispersal limitation does not control high elevational distribution of alien plant species in the southern Sierra Nevada, California. Natural Areas Journal **36**:277–287.

Ryan ME, Palen WJ, Adams MJ, Rochefort RM. 2014. Amphibians in the climate vise: loss and restoration of resilience of montane wetland ecosystems in the western US. Frontiers in Ecology and the Environment **12**:232–240.

Safford HD, Van de Water KM. 2014. Using fire return interval departure (FRID) analysis to map spatial and temporal changes in fire frequency on national forest lands in California. Res. Pap. PSW-RP-266. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Sanders SD, Flett MA. 1989. Montane riparian habitat and Willow Flycatchers: threats to a sensitive environment and species. Pages 262–266 in D. L. Abell, editor. Proceedings of the California Riparian Systems Conference: protection, management, and restoration for the 1990s; 1988 September 22-24; Davis, CA. Gen. Tech. Rep. PSW-GTR-110. U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, CA.

Sawaske SR, Freyberg DL. 2014. An analysis of trends in baseflow recession and low-flows in rain-dominated coastal streams of the Pacific coast. Journal of Hydrology **519**:599–610.

Schenck SM, Gifford EW. 1952. Karuk ethnobotany. University of California Press, Berkeley, CA.

Seavy NE, Gardali T, Golet GH, Griggs FT, Howell CA, Kelsey R, Small SL, Viers JH, Weigand JF. 2009. Why climate change makes riparian restoration more important than ever: recommendations for practice and research. Ecological Restoration **27**:330–338.

Semlitsch RD. 2000. Principles for management of aquatic-breeding amphibians. The Journal of Wildlife Management **64**:615–631.

Sette CM, Vredenburg VT, Zink AG. 2015. Reconstructing historical and contemporary disease dynamics: a case study using the California slender salamander. Biological Conservation **192**:20–29.

Shakesby RA, Doerr SH. 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews **74**:269–307.

Shields CA, Kiehl JT. 2016. Simulating the Pineapple Express in the half degree Community Climate System Model, CCSM4. Geophysical Research Letters **43**:7767–7773.

Sikes K, Cooper DJ, Weis S, Keeler-Wolf T, Barbour M, Ikeda D, Stout D, Evens J. 2013. Fen conservation and vegetation assessment in the National Forests of the Sierra Nevada and adjacent mountains, California. Version 2. California Native Plant Society, Davis, CA and U.S. Forest Service, Pacific Southwest Region, Vallejo, CA.

Sikes K, Roach-McIntosh D, Stout D, Nelson J. 2012. Vegetation assessment and ranking of fen and wet meadow sites of the Shasta-Trinity National Forest, California. California Native Plant Society, Davis, CA, and U.S. Department of Agriculture, Forest Service, Shasta-Trinity National Forest, Davis, CA.

Skinner C. 2003. Fire history of upper montane and subalpine glacial basins in the Klamath Mountains of northern California. Pages 145–151 in R. C. Galley, R. C. Klinger, and N. G. Sugihara, editors. Proceedings of Fire Conference 2000: the first national congress on fire ecology, prevention, and management. Miscellaneous Publication No. 13. Tall Timbers Research Station, Tallahassee, FL.

Skinner CN, Taylor AH, Agee JK. 2006. Klamath Mountains bioregion. Pages 170–194 in N. G. Sugihara, J. W. van Wagtendonk, J. Fites-Kaufmann, K. E. Shaffer, and A. E. Thode, editors. Fire in California's ecosystems. University of California Press, Berkeley, CA.

Snyder KU. 2000. Debris flows and flood disturbance in small, mountain watersheds. Master of Science thesis. Oregon State University, Corvallis, OR.

Spence JR, Shevock JR. 2012. *Ptychostomum pacificum* (Bryaceae), a new fen species from California, Oregon, and western Nevada, USA. Madroño **59**:156–162.

Stephens SL, Martin RE, Clinton NE. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands, and grasslands. Forest Ecology and Management **251**:205–216.

Stewart IT, Cayan DR, Dettinger MD. 2004. Changes in snowmelt runoff timing in western North America under a `business as usual' climate change scenario. Climatic Change **62**:217–232.

Stewart IT, Cayan DR, Dettinger MD. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate **18**:1136–1155.

Stillwater Sciences. 2012. A guide for restoring functionality to mountain meadows of the Sierra Nevada. Prepared for American Rivers, Nevada City, CA. Stillwater Sciences, Berkeley, CA.

Swain DL, Langenbrunner B, Neelin JD, Hall A. 2018. Increasing precipitation volatility in twenty-first-century California. Nature Climate Change **8**:427.

Syphard AD, Keeley JE, Pfaff AH, Ferschweiler K. 2017. Human presence diminishes the importance of climate in driving fire activity across the United States. Proceedings of the National Academy of Sciences **114**:13750–13755.

Tague C, Seaby L, Hope A. 2009. Modeling the eco-hydrologic response of a Mediterranean type ecosystem to the combined impacts of projected climate change and altered fire frequencies. Climatic Change **93**:137–155.

Taylor AH, Trouet V, Skinner CN, Stephens S. 2016. Socioecological transitions trigger fire regime shifts and modulate fire–climate interactions in the Sierra Nevada, USA, 1600–2015 CE. Proceedings of the National Academy of Sciences **113**:13684–13689.

Thorne JH, Boynton RM, Flint LE, Flint AL. 2015. The magnitude and spatial patterns of historical and future hydrologic change in California's watersheds. Ecosphere **6**:1–30.

Thorne JH, Boynton RM, Holguin AJ, Stewart JAE, Bjorkman J. 2016. A climate change vulnerability assessment of California's terrestrial vegetation. California Department of Fish and Wildlife, Sacramento, CA.

Tolman DA. 2007. Soil patterns in three *Darlingtonia* fens of Southwestern Oregon. Natural Areas Journal **27**:374–384.

Turner NJ, Deur D, Mellott CR. 2011. "Up on the mountain": ethnobotanical importance of montane sites in Pacific Coastal North America. Journal of Ethnobiology **31**:4–43.

Viers JH, Purdy SE, Peek RA, Fryjoff-Hung A, Santos NR, Katz JVE, Emmons JD, Dolan DV, Yarnell SM. 2013. Montane meadows in the Sierra Nevada: changing hydroclimatic conditions and concepts for vulnerability assessment. University of California, Center for Watershed Sciences, Davis, CA.

Wahl ER, Zorita E, Trouet V, Taylor AH. 2019. Jet stream dynamics, hydroclimate, and fire in California from 1600 CE to present. Proceedings of the National Academy of Sciences **116**:5393–5398.

Weixelman DA, Hill B, Cooper DJ, Berlow EL, Viers JH, Purdy SE, Merril AG, Gross SE. 2011. A field key to meadow hydrogeomorphic types for the Sierra Nevada and southern Cascade Ranges in California. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, Vallejo, CA.

Welsh HH, Pope KL, Boiano D. 2006. Sub-alpine amphibian distributions related to species palatability to non-native salmonids in the Klamath mountains of northern California. Diversity and Distributions **12**:298–309.

Westerling AL. 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. Philosophical Transactions of the Royal Society B: Biological Sciences **371**:20150178.

Westerling AL. 2018. Wildfire simulations for California's Fourth Climate Change Assessment: projecting changes in extreme wildfire events with a warming climate. California's Fourth Climate Change Assessment. Publication Number: CCCA4-CEC-2018-014. California Energy Commission, Sacramento, CA.

Westerling AL, Bryant BP, Preisler HK, Holmes TP, Hidalgo HG, Das T, Shrestha SR. 2011. Climate change and growth scenarios for California wildfire. Climatic Change **109**:445–463.

Wolf EC, Cooper DJ. 2015. Fens of the Sierra Nevada, California, USA: patterns of distribution and vegetation. Mires and Peat **15**:1–22.

Woltemade CJ. 2000. Ability of restored wetlands to reduce nitrogen and phosphorus concentrations in agricultural drainage water. Journal of Soil and Water Conservation **55**:303–309.

Yarnell SM, Viers JH, Mount JF. 2010. Ecology and management of the spring snowmelt recession. BioScience **60**:114–127.

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),[8] 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.[9]

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

[9] 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:

Vulnerability = [(Climate Exposure*0.5) x Sensitivity] - 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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Literature Cited

EcoAdapt. 2014a. A climate change vulnerability assessment for aquatic resources in the Tongass National Forest. EcoAdapt, Bainbridge Island, WA. 124 pp.

EcoAdapt. 2014b. A climate change vulnerability assessment for resources of Nez Perce-Clearwater National Forests. Version 3.0. EcoAdapt, Bainbridge Island, WA. 398 pp.

Glick P, Stein BA, Edelson NA. 2011. Scanning the conservation horizon: A guide to climate change vulnerability assessment. National Wildlife Federation, Washington, D.C.

Gregg RM, editor. 2018. Hawaiian Islands climate vulnerability and adaptation synthesis. EcoAdapt, Bainbridge Island, WA. 284 pp.

Hutto SV, Higgason KD, Kershner JM, Reynier WA, Gregg DS. 2015. Climate change vulnerability assessment for the north-central California coast and ocean. Marine Sanctuaries Conservation Series ONMS-15-02. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Office of National Marine Sanctuaries, Silver Spring, MD. 473 pp.

Intergovernmental Panel on Climate Change (IPCC). 2007. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Pages 617–652 in M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson, editors. Cambridge University Press, Cambridge, UK.

Kershner JM, editor. 2014. A climate change vulnerability assessment for focal resources of the Sierra Nevada. Version 1.0. EcoAdapt, Bainbridge Island, WA. 418 pp.

Lawler J. 2010. Pacific Northwest Climate Change Vulnerability Assessment. From http://climatechangesensitivity.org

Manomet Center for Conservation Sciences and National Wildlife Federation. 2012. The vulnerabilities of fish and wildlife habitats in the Northeast to climate change. A report to the Northeastern Association of Fish and Wildlife Agencies and the North Atlantic Landscape Conservation Cooperative. Manomet Center for Conservation Sciences, Plymouth, MA.

Moss R, Schneider S. 2000. Towards consistent assessment and reporting of uncertainties in the IPCC TAR. In R. Pachauri and T. Taniguchi, editors. *Cross-Cutting Issues in the IPCC Third Assessment Report*. Global Industrial and Social Progress Research Institute (for IPCC), Tokyo.