# Extremes to Ex-Streams: Ecological Drought Adaptation in a Changing Climate

## 2019

Cover photos: Jonny Armstrong

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## Table of Contents

## List of Tables



## Table of Figures

# Executive Summary

Climate change is one of the most pressing challenges on natural and cultural resource management and conservation practice. Resource managers and conservation planners are addressing these challenges by revising current plans and practices with increased attention on potential climate impacts to natural resources, communities, and socioeconomic values to better meet long-term goals. However, decision-making is complicated by uncertainty in terms of which adaptation actions are best suited for different implementation conditions and supported by scientific evidence (Sutherland et al. 2004; Cook et al. 2009; Eriksen et al. 2011; Bayliss et al. 2012; Cross et al. 2012). The purpose of this and other EcoAdapt adaptation science assessments is to evaluate the body of scientific knowledge supporting specific climate adaptation actions to determine the conditions under which particular actions may be most effective for achieving management goals.

The Northwest United States is vulnerable to climatic stressors, including warming air temperatures and variable precipitation patterns that will likely result in significantly altered snowpack, stream flows, and water availability (Snover et al. 2013; Mote et al. 2014). These factors will combine to affect ecological drought conditions throughout the Northwest Climate Adaptation Science Center (NWCASC) region. Ecological drought is a deficit in naturally available water supplies that creates multiple stresses across ecosystems (Crausbay et al. 2017). Increases in the frequency and severity of ecological droughts under changing climate conditions may drive ecosystems beyond thresholds of vulnerability (e.g., transition from mesic to xeric [moist to dry] habitats). Consequently, natural and managed systems may shift into alternative stable states or transform into a new system, requiring reconsideration and possible modifications to current management practices. **Section 1** of this report presents an overview of ecological drought impacts in the Northwest region.

**Section 2** describes ecological drought adaptation options available to and in use by resource managers as well as potential future options for management. Hundreds of regionally relevant ecological drought adaptation strategies and actions were identified by reviewing the options generated and prioritized by managers during regional adaptation workshops (e.g., Nez Perce-Clearwater National Forests, North Cascadia Adaptation Partnership) and other adaptation planning efforts (e.g., Lummi Nation Climate Change Mitigation and Adaptation Plan, Northern Institute of Applied Climate Science Climate Change Response Framework). This list underwent a second screening and editing process to remove duplicates and combine similar actions, resulting in a pared down list of 72 actions.

Documenting the ecological and socioeconomic conditions under which specific ecological drought management strategies and actions (e.g., planting drought-resistant trees) are most suited will help identify when and how traditional responses to ecological drought may need to be modified to respond to changing climatic conditions. We reviewed the scientific and gray literature (e.g., agency reports, project websites, tools) to identify available evidence behind the factors that contribute to the success and longevity of these adaptation actions. **Section 3**

presents the findings of this literature review and synthesis by different adaptation goals and strategies, including:

- Goal: Retain ecologically available water in the natural system
    - Restore habitats by maintaining native vegetation cover and removing invasive species
    - Restore and reconnect floodplains to allow for groundwater recharge
    - Enhance natural water storage (e.g., beaver dams/dam analogs, snow fencing, large woody debris, green infrastructure, rock structures)
    - Maintain and enhance infiltration, water storage capacity, and/or health of soils (e.g., fuel treatments, biochar application, restricting access to minimize soil compaction)
- Goal: Facilitate species persistence under drought conditions
    - Consider species type, timing, and location in management activities
    - Create or enhance water supply (e.g., wildlife water developments, constructed wetlands, forest canopy manipulation, snow fencing and harvesting)
    - Protect vulnerable species through assisted migration and improved habitat connectivity
    - Identify and protect drought refugia
- Goal: Reduce stress on ecosystems and species
    - Reduce tree density and fuel loads through thinning and prescribed burns
    - Use exclosures and fences to protect groundwater-dependent habitats and associated species
    - Use livestock rotation and diversification to reduce pressure on vegetation and soils
    - Enhance ecologically available water supply via environmental watering
    - Reduce water withdrawals
    - Promote water conservation through collaborative agreements (e.g., water banking, water trading)
- Goal: Increase understanding of ecological drought
    - Improve understanding of ecological drought impacts and adaptation options through synthesis, research, monitoring, and evaluation

When selecting adaptation actions for implementation, managers should consider both *effectiveness* (action reduces vulnerability) and *feasibility* (action capable of being implemented). Implementation feasibility considers technical (e.g., financial, staff, data) and socio-political (e.g., social, political, institutional, legal) barriers. To complement the literature review, we classified each adaptation action according to its implementation feasibility and effectiveness in reducing ecological drought vulnerabilities to support informed decision-making. Rankings are based on evidence from the literature and/or expert opinion (e.g., natural resource managers provided rankings during workshops that were a part of other adaptation planning projects). **Section 4** presents these evaluations in a table to help managers prioritize actions for implementation, better target management efforts toward specific challenges, and/or evaluate whether to proceed with implementation.

Knowing which adaptation actions can be best implemented at different scales and in various ecosystems will help resource managers to identify and leverage funding opportunities, create new or enhance existing partnerships, and communicate and coordinate with other agencies and organizations to prioritize on-the-ground ecological drought responses. This project directly supports the expressed goal of the Northwest Climate Adaptation Science Center to provide scientific research and synthesis to support natural resource management in a changing climate.

# 1. An Overview of Ecological Drought in the Northwest

## Defining ecological drought

Ecological drought is an "episodic deficit in water availability that drives ecosystems beyond thresholds of vulnerability, impacts ecosystem services, and triggers feedbacks in natural and/or human systems" (Crausbay et al. 2017). Historically, drought analyses have focused more narrowly on drought impacts to human communities (e.g., water supply, agriculture), either minimally or failing to consider ecological impacts (Crausbay et al. 2017; McEvoy et al. 2018). Comparatively, ecological drought holistically examines how natural processes and human actions interact to affect water available to ecological systems, as well as ecological responses to water scarcity, over longer time frames and larger spatial scales (Crausbay et al. 2017).

> Types of Drought
>
> *Meteorological*: period of dry weather
>
> *Hydrological*: low water supplies in rivers and reservoirs following meteorological drought
>
> *Agricultural*: water shortage causing damage to agricultural crops and food supply
>
> *Ecological*: water shortage causing damage to habitats, species, and ecosystem services
>
> *Socioeconomic*: water shortage affecting the supply and demand of goods, such as water and food.
>
> *Source:* Crausbay et al. 2017; Zaniolo et al. 2018

The water available to ecological systems in a given location is influenced by several natural and human-caused factors (Crausbay et al. 2017):

- *Local and sub-regional meteorological conditions (e.g., local weather conditions)*: Local meteorological conditions are controlled by atmosphere-ocean-land interactions, such as sea surface temperatures, the El Niño Southern Oscillation, and land-atmosphere feedbacks (Cook et al. 2016). A dry weather pattern and the associated lack of rainfall is termed "meteorological drought" (National Oceanic and Atmospheric Administration 2018).
- *Local and regional landscape characteristics*: Landscape characteristics such as topography and soil types influence rainfall, water storage, and water movement across the landscape (Crausbay et al. 2017). For example, topography is particularly important in the Northwest United States, where longitudinal mountain ranges drive significant differences in annual rainfall from west to east (NWCASC 2017).
- *Human-caused climate change*: Increasing air temperatures associated with global climate change elevate evaporative demand and drive changes in precipitation form

from snow to rain, affecting water delivery and residence time (e.g., as snowpack) in natural systems (Trenberth et al. 2014). As a result, climate change is contributing to more frequent, longer, and severe droughts and increases the likelihood of multidecadal mega-droughts (Cook et al. 2016).

- *Human landscape and hydrological modifications*: Human modifications of the landscape and hydrological network (e.g., reservoirs, irrigation) alter water movement and storage (Van Loon et al. 2016).

The sensitivity of ecological systems to water scarcity, and their capacity to accommodate or adapt to drought conditions, is similarly influenced by both natural and human-caused factors (Glick et al. 2011; Crausbay et al. 2017). Most ecosystems and species have evolved with some exposure to drought, which has contributed to the evolution of drought adaptations (Halofsky et al. 2018b). For example, in response to low water availability, individual species may temporarily migrate to areas with more abundant water, and ecosystems may exhibit temporary shifts in community productivity, composition, structure, and distribution (Halofsky et al. 2018b). Sensitivity and responses can vary within similar species and ecosystems depending on genetics, life history, phenotypic and behavioral plasticity, location, and other factors (Glick et al. 2011; Crausbay et al. 2017). However, human landscape alterations and management decisions can also alter the inherent sensitivity and response to drought conditions by natural landscapes and native species (Glick et al. 2011; McEvoy et al. 2018). For example, habitat loss and fragmentation can affect the ability of individual species to temporarily or permanently disperse to more mesic microhabitats (e.g., climatic refugia; McGuire et al. 2016). Similarly, high basal stand density (which is currently common in many managed forest stands) can increase tree competition for soil moisture, effectively enhancing individual tree sensitivity to drought-induced moisture stress (Bradford & Bell 2017).

Ecological drought occurs when water scarcity exceeds ecological tolerance thresholds. Ecological drought can lead to a range of impacts in ecological communities and affect the provisioning of ecosystem services, which can lead to additional impacts and responses in human systems. Ecological and socioecological drought repercussions vary in scope, severity, and permanence. For example, impacts may occur predominately in natural systems, in human uses of natural systems (e.g., ecosystem services), or both. Impacts may be relatively minimal and/or short-lived (e.g., seasonal rangeland productivity loss) or be quite severe and long-lasting (e.g., vegetation type conversion and subsequent loss of original ecosystem service suite) (Crausbay et al. 2017).

## Ecological drought impacts in the Northwest

The Northwest United States exhibits significant climatic variability depending on location. Numerous mountain ranges running north to south intercept eastward maritime air flow, resulting in large latitudinal climatic gradients from the moist, temperate Oregon and Washington coasts to the arid, dry interior areas of eastern Washington and Oregon, Idaho, and western Montana. Local weather patterns further interact with diverse topography of mountain ranges, river basins, and plains to create a diversity of regional habitat types, ranging from dense temperate rainforest to arid coniferous forest to arid grasslands and high desert shrubland. The diversity of habitat types and historic climates mean that the Northwest will likely exhibit variable responses to increasing frequency and severity of ecological drought (NWCASC 2017). Moisture-limited habitats are acutely vulnerable to ecological drought; these include, for example, low-elevation areas in the Klamath, Siskyou, Blue, and Wallowa mountains as well as the Columbia Highlands, northern Rocky Mountains, and eastern foothills of the Cascade Mountains (Chmura et al. 2011). Areas such as the Olympic Mountains and mid-elevations of the Cascades, which are not currently moisture limited, may still be vulnerable to drought as a result of the combined effects of rainfall, snowmelt, surface runoff, subsurface flow, and evapotranspiration (Chmura et al. 2011). General ecological drought impacts are summarized by major ecosystem type (Table 1).

Table 1. Potential impacts of ecological drought on ecosystems and ecosystem services.

| | |
|---|---|
| **FORESTS AND WOODLANDS** | - Altered tree growth and productivity<br>    o Lower-elevation forests and woodlands, especially dry forest types, are likely to experience growth and productivity declines (e.g., Douglas fir [*Pseudotsuga menziesii*] growth likely to decline on east and west sides of Cascades)<br>    o Higher-elevation forests may experience increased productivity as less snowpack will facilitate longer growing seasons<br>- Impaired regeneration and recruitment via reduced seedling establishment and survival, particularly at lower elevations<br>- Decreased moisture leading to increased fuel flammability and tree mortality<br>- Increased wildfire frequency, size, and severity, and longer fire season, affecting stand age, structure, composition, and increasing risk of vegetation type conversion<br>- Increased vulnerability to disease and insect outbreaks<br>- Increased invasive and non-native species abundance<br>- Local shifts in species composition (e.g., declines in drought-intolerant species), particularly at lower elevations and/or ecotones<br>- Regional shifts in habitat distribution (e.g., upwards in elevation)<br>- Potential vegetation type conversions when paired with increasing fire disturbance (e.g., from mesic forests to dry forest types, or from dry forests to woodland, shrubland, or grassland)<br><br><u>Sources</u>: Rahel & Olden 2008; Klos et al. 2009; Latta et al. 2010; Littell et al. 2010; Millar & Stephenson 2015; Sun et al. 2015; Abatzoglou & Williams 2016; Harvey et al. 2016; Vose et al. 2016a; Restaino et al. 2016; Halofsky & Peterson 2017; USDA Forest Service 2017a, 2017b; Halofsky et al. 2018a, 2018b; Hudec et al. 2018; Southwest Oregon Adaptation Partnership 2018; USDA Forest Service 2018a; Davis et al. 2019 |

| | |
|---|---|
| **SHRUBLANDS, GRASSLANDS, AND RANGELANDS** | - Impaired recruitment by reducing seedling germination and survival<br>- Shifts in habitat distribution: some potential expansions with the decline of woodland and forested vegetation types, but also some habitat losses in driest areas (e.g., low elevations)<br>- Reduced productivity and biodiversity<br>- Shifts in species composition toward more dry-adapted species, including dry-adapted invasive species, particularly at lower elevations<br>- Potential grassland habitat expansion with the decline of forest and woodland vegetation types<br>- Increased survival of non-native annual grasses, which may lead to increased fire frequency and extent<br><br><u>Sources</u>: Schlaepfer et al. 2014; Yung et al. 2015; Vose et al. 2016a; Halofsky & Peterson 2017; USDA Forest Service 2017a, 2017b; Halofsky et al. 2018a, 2018b; USDA Forest Service 2018a |
| **ALPINE AND SUBALPINE HABITATS** | - Altered plant growth, recruitment, and phenology<br>- Potential alpine habitat contraction if lower elevation forests move upward<br>- Reduced seedling germination and survival, particularly at lower-elevation limits<br>- Reduced tree growth, particularly at lower-elevation limits<br><br><u>Sources</u>: Albright & Peterson 2013; Halofsky & Peterson 2017; Halofsky et al. 2018b; Hudec et al. 2018; Southwest Oregon Adaptation Partnership 2018 |
| **WETLANDS, MARSHES, AND BOGS** | - Accelerated habitat drying, particularly for precipitation-dependent systems<br>- Habitat contraction away from drying edges and/or complete summer drying<br>- Hydroperiod shifts from permanent to ephemeral ponds and reducing wet periods in ephemeral ponds<br>- Enhanced mortality of wet-adapted plant species<br>- Shifts in species composition toward more xeric and drought-tolerant species<br>- Tree encroachment into wetlands and wet meadows<br><br><u>Sources:</u> Poff et al. 2002; Rogers et al. 2006; Halofsky & Peterson 2017; Raymond et al. 2017; Halofsky et al. 2018b; Hudec et al. 2018 |
| **RIVERS, STREAMS, AND RIPARIAN HABITATS** | - Reduced streamflows, particularly summer baseflows: flow reductions will be most acute in snowmelt-dependent streams relative to groundwater-controlled streams<br>- Lower streamflows will likely reduce stream connectivity and/or habitat suitability (e.g., dissolved oxygen, water temperature) for aquatic organisms, potentially reducing fitness and elevating mortality<br>- Changes in the amount and timing of precipitation may affect water levels and flow regimes of coastal rain-dominated rivers as they receive most or all of their precipitation during the winter and experience steep recession rates during the summer dry period<br>- Reduced riparian vegetation germination, driving altered species composition<br>- Reduced riparian tree and shrub growth and productivity<br>- Reduced riparian width in response to declining summer baseflows; wider valley bottoms may be most susceptible to streamside drying |

| | |
|---|---|
| | - Increased fire frequency<br><br>Sources: Luce & Holden 2009; Safeeq et al. 2014; Sawaske & Freyberg 2014; Power et al. 2016; Vose et al. 2016b; Halofsky & Peterson 2017; Klein et al. 2017; Li et al. 2017; USDA Forest Service 2017a; Halofsky et al. 2018b; Hudec et al. 2018; Southwest Oregon Adaptation Partnership 2018 |
| **ESTUARIES AND COASTAL WETLANDS** | - Reduced freshwater inputs, resulting in higher salinities, warmer water temperatures, and lower dissolved oxygen levels<br>- Reduced plant biomass<br>- Potential changes in the extent of saltwater intrusion<br>- Drying and compaction of coastal soils<br>- Drought may expose estuarine vegetation to salinity extremes and potential vegetation type shifts<br>- Drought may exacerbate saltwater intrusion by allowing saline water further inland; hotter, drier, and more saline conditions may compromise dune vegetation and cause instability<br><br>Sources: Poff et al. 2002; Desantis et al. 2007; Greaver & Sternberg 2010; Nolan et al. 2016; Buffington et al. 2018; Conrads et al. 2018 |
| **FISH AND WILDLIFE** | - Dewatering of channels and lower flows, which may expose fish eggs, larvae, and juveniles to lethal water temperatures, dissolved oxygen levels, or desiccation<br>- Limited migration and dispersal opportunities for anadromous fish<br>- Changes in behavioral traits (e.g., movement, diet)<br>- Increased physiological stress, such as altered survival and growth rates<br>- Altered species interactions (e.g., competition, predation)<br>- Reduced diversity and abundance in hyporheic zones of streams<br>- Changes in trophic structure and food resources<br><br>Sources: Lake 2003; Marques et al. 2007; Rahel & Olden 2008; National Marine Fisheries Service 2015, 2016; Klein et al. 2017; Prugh et al. 2018; Aspin et al. 2019 |
| **ECOSYSTEM SERVICES** | - Altered recreational and cultural opportunities (e.g., fishing, hunting, swimming, skiing, snowshoeing, wildlife viewing)<br>- Reduced water supply and quality<br>- Reduced property values and aesthetic qualities<br>- Reduced habitat extent and forage<br>- Reduced livestock carrying capacity due to increased plant mortality, reduced vegetation cover, and increased soil erosion<br><br>Sources: National Marine Fisheries Service 2015, 2016; USDA Forest Service 2017a, 2017b; USDA Forest Service 2018a; Raheem et al. 2019 |

## 2. Ecological Drought Adaptation

Documenting the ecological and socioeconomic conditions under which specific ecological drought management strategies and actions (e.g., planting drought-resistant trees) are most suited will help identify when and how traditional responses to ecological drought may need to be modified to respond to changing climatic conditions. We reviewed the scientific and gray literature (e.g., agency reports, project websites, tools) to identify evidence behind the factors that contribute to the success and longevity of ecological drought adaptation actions, and documented on-the-ground examples of climate-informed ecological drought management in action. The scientific and gray literature relevant to the topic of ecological drought spans a number of disciplines and fields, from resource management and engineering to agriculture and wildlife, and we looked for supporting evidence from the Northwest as well as relevant studies from outside the project geography. Synthesizing what has worked and how and what has not worked and why can help identify potential modifications to current practices and facilitate understanding of consequences of management decisions. Similar adaptation science assessments have been conducted on wildfire (Gregg et al. 2016) and sea level rise (Gregg et al. 2018) in the Northwest.

We identified over 260 regionally relevant ecological drought adaptation strategies and actions by reviewing the literature and options generated and prioritized by managers during regional adaptation workshops (e.g., Nez Perce-Clearwater National Forests, North Cascadia Adaptation Partnership) and other adaptation planning efforts (e.g., Lummi Nation Climate Change Mitigation and Adaptation Plan, Stillaguamish Tribe of Indians Natural Resources Climate Change Adaptation Plan, Northern Institute of Applied Climate Science Climate Change Response Framework). These adaptation strategies and actions include those already being implemented throughout the Northwest as well as potential future options for management. Evidence from the literature behind these actions were categorized according to their sectoral/ecosystem relevance (e.g., forests, shrublands, agricultural/rangelands, wildlife, fish), co-benefits and conflicts with other actions and resources, and effectiveness in reducing drought vulnerabilities. Literature was primarily identified using Google Scholar, TreeSearch, and TACCIMO (Template for Assessing Climate Change Impacts and Management Options). In total, we located and reviewed 498 documents of relevance from 1957–2019 (Figure 1).

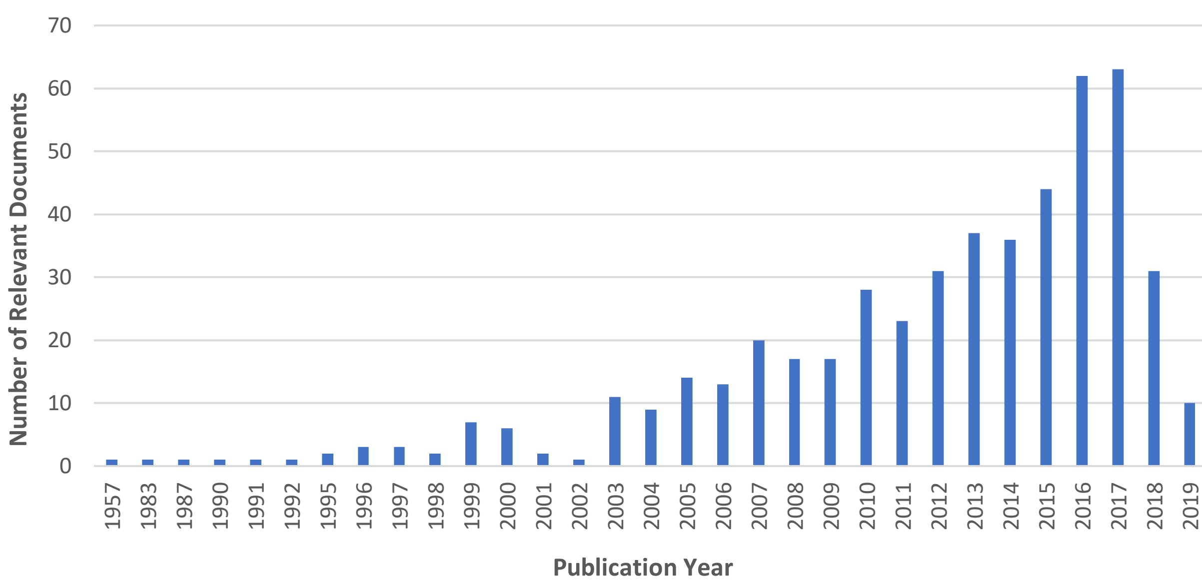


**Figure 1. Publication year of relevant literature reviewed.**

Many of these actions are already in use by managers across the Northwest to meet different management goals and objectives. However, these actions, whether anticipatory or responsive, can also be leveraged to enhance overall resilience to ecological drought. Additionally, managers may choose to alter the implementation of existing management actions to adapt to changing conditions, or develop novel approaches to respond to the challenges of ecological drought (Figure 2).


**Figure 2. Example continuum of climate-informed management options for forests.**

Where a manager's choices land on the continuum is frequently determined by management values and culture, risk tolerance, and regulatory restrictions. Spreading risk across different adaptation approaches may help managers maximize the outcomes of their investments (Aplet & McKinley 2017). For example, decision-makers may opt to manage resources for persistence in a changing climate by resisting climate change impacts or promoting resilience, or facilitate a transition to some future altered state (Millar et al. 2007). Similarly, Aplet & Cole (2010) present three options for managing for change in an uncertain future—resist change (e.g., active restoration to retain historical ecosystem function and processes), accept change (e.g., observe ecosystem shifts without manipulation), and guide change (e.g., active facilitation and transformation of systems). We use the following classification for adaptation strategies and actions (Kershner & Gregg 2013):

- **Resistance.** These strategies help to limit the effects of climate change on a resource and/or bolster a resource's ability to retain fundamental processes and functions by maintaining relatively limited changes over time. *Near-term approach*
- **Resilience.** These strategies help a resource withstand the impacts of climate change by absorbing the effects of or recovering from changes in order to enable a return to prior conditions. *Near- to mid-term approach*

- **Response.** These strategies intentionally accommodate change and/or enable resources to transition to changing and new conditions through adaptive responses. *Long-term approach*
- **Knowledge.** These strategies are aimed at gathering more information about climatic changes, impacts, or the effectiveness of management actions in addressing climate change. *Near- to long-term approach*
- **Collaboration.** These strategies may help coordinate efforts and/or capacity across landscapes and agencies. *Near- to long-term approach*

The overarching goals of ecological drought adaptation are to retain ecologically available water in the system (e.g., creating or maintaining ecosystem water supply and limiting or preventing withdrawal of water), reduce the sensitivity of ecosystems to water deficits (e.g., reducing stress on habitats and species, facilitating species persistence under drought conditions), and respond to post-drought conditions (e.g., promptly revegetating disturbed sites, managing altered streamflows). Additional strategies and actions fall under a broader category of co-benefits—those that are taken for some other primary purpose but have ancillary benefits towards limiting ecological drought effects—as well as those geared towards facilitating research to better inform management. For the purposes of this report, we have simplified the list of relevant ecological drought adaptation strategies and actions (Table 2).[1]

Table 2. Ecological drought adaptation goals, strategies, and actions.

| Adaptation Approach | Adaptation Goal | Adaptation Strategies (and Actions) |
|---|---|---|
| Resistance | Reduce stress on ecosystems and species | Reduce water withdrawals |
| | | Use exclosures and fences to protect groundwater-dependent habitats and associated species |
| | | Enhance ecologically available water supply via environmental watering |
| | | Reduce tree density and fuel loads through thinning and prescribed burns |
| Resilience | Retain ecologically available water in the natural system | Restore habitats by maintaining native vegetation cover and removing invasive species |
| | | Enhance natural water storage (e.g., beaver dams/dam analogs, large woody debris, green infrastructure, rock structures) |
| | | Maintain and enhance infiltration, water storage capacity, and/or health of soils (e.g., fuel treatments, biochar application, restricting access to minimize soil compaction) |
| | | Restore and reconnect floodplains to allow for groundwater recharge |

[1] For complete list of ecological drought adaptation strategies and actions sourced from the scientific and gray literature, please see page 46.

| Adaptation Approach | Adaptation Goal | Adaptation Strategies (and Actions) |
|---|---|---|
| | Reduce stress on ecosystems and species | Use livestock rotation and diversification to reduce pressure on vegetation and soils |
| | Facilitate species and habitat persistence under drought conditions | Consider species type, timing, and location in management activities |
| | | Create or enhance water supply (e.g., wildlife water developments, constructed wetlands, canopy manipulation, snow fencing and harvesting) |
| Response | Facilitate species and habitat persistence under drought conditions | Identify and protect drought refugia |
| | | Actively plant and store seed from drought-tolerant species and individuals |
| | | Protect vulnerable species through assisted migration and improved habitat connectivity |
| Knowledge | Increase understanding of ecological drought | Improve understanding of ecological drought impacts and adaptation options through synthesis, research, monitoring, and evaluation |
| Collaboration | Reduce stress on ecosystems and species | Promote water conservation through collaborative agreements (e.g., water banking, water trading, voluntary reductions) |

# 3. Literature Findings

## Goal: Retain ecologically available water in the natural system

### Restore habitats by maintaining native vegetation cover and removing invasive species

Increasing native cover and habitat heterogeneity and decreasing the presence of invasive species is linked to increased ecosystem health for forests and rivers and streams (Palmer et al. 2009; van Kleunen et al. 2010). In general, invasive plant species consume more water than native plants as invasives typically exhibit faster growth rates (van Kleunen et al. 2010) and higher rate of leaf-level water use (Leishman et al. 2007; Cavaleri & Sack 2010). For example, the invasive salt cedar (*Tamarix* spp.) consumes 10–20 times the amount of water used by native species (McCormick et al. 2010), and is better able to tolerate water stress than the native plants it replaces (Kerns et al. 2009). While this aggressive invasive is already found in parts of the Pacific Northwest, a recent study found that over 20% of land east of the Cascade Mountains will support suitable habitat by the end of the century (Kerns et al. 2009). Harms & Hiebert (2006) found that cutting and burning (both followed by herbicide application) reduced tamarisk cover by up to 95%, although González et al. (2017) found that these same treatments have been linked to invasions of exotic forbs in the Southwest post-tamarisk removal. Biocontrol agents (e.g., central Asian salt cedar leaf beetle, *Diorhabda elongata*) have been successfully used to facilitate tamarisk defoliation along 1,000 km of riparian habitat along the Colorado River (Hultine et al. 2010), although the impact of *Diorhabda* spp. on *Tamarix* and riparian plant communities across large landscapes is uncertain (Hultine et al. 2015; González et al. 2017). Active revegetation of native riparian species (e.g., seeding, transplanting) in sites proximal to perennial water sources with coarser soils post-tamarisk removal was linked to increased native cover at 28 restoration sites in Arizona, Nevada, and New Mexico (Bay & Sher 2008).

Western juniper (*Juniperus occidentalis*) is also known to be a high water-use species (Deboodt et al. 2008) that is drought-tolerant. Fire suppression has enabled western juniper dominance and expansion in the Northwest over the last century, and management efforts to reduce densities have yielded several projects designed to intentionally convert juniper to grass- or sagebrush-dominated communities. Kuhn et al. (2007) evaluated the effects of western juniper removal on water yields within the water-stressed Klamath River Basin, and found limited evidence that juniper removal would lead to a substantial increase in water yield basinwide; however, the authors note that more field research on managing juniper for local-level effects on surface flows and soil moisture is needed. Juniper removal has been linked to higher soil moisture levels compared with sites still dominated by the trees in Oregon (Mollnau et al. 2014) and the Great Basin (Roundy et al. 2014). However, this removal has also increased site susceptibility to invasion by exotic annual grasses, such as cheatgrass (*Bromus tectorum*), Japanese brome (*Bromus japonicus*), and medusahead wildrye (*Taeniatherum caput-medusae*) (Davies et al. 2019). While juniper is known to be a drought-tolerant species, intense drought conditions have caused substantial juniper mortality in the Southwest (Clifford et al. 2011), central Oregon (Soulé & Knapp 2007), and southern Utah (Stapleton 2019).

### Restore and reconnect floodplains to allow for groundwater recharge

Maintaining existing and reconnecting and restoring floodplains increases water retention and storage potential. Floodplains provide pervious space for groundwater recharge but have lost their ability to store water where floodplains are disconnected from rivers by engineered structures such as levees, dams, and channel straightening (Fleckenstein et al. 2004; Opperman et al. 2010; Opperman et al. 2011; Loos & Shader 2016). Restoring floodplain connectivity can increase water storage capacity (Opperman et al. 2009; Isaak et al. 2017). By increasing water supply storage, floodplain reconnection also increases flexibility in reservoir management (Opperman et al. 2009). Hunt et al. (2018) determined that the 2012 restoration of 0.1 $km^2$ of meadow floodplain significantly increased baseflow and groundwater storage in the Sierra Nevada Mountains. After restoration, summer baseflows increased 5–12 times over pre-restoration volumes between 2012–2015, and groundwater levels rose at four out of five sites near the stream channel.

### Enhance natural water storage (e.g., beaver dams/dam analogs, snow fencing, large woody debris, green infrastructure, rock structures)

Retaining water in ecosystems through natural water storage is key to avoiding the effects of ecological drought (Crausbay et al. 2017), therefore strategic measures to improve the ability of natural systems to capture and store water will reduce drought vulnerability (Holmes 2016). Natural water storage systems such as floodplains, riparian areas, and wetlands absorb water and promote groundwater recharge and release (Holmes et al. 2017). Techniques used to facilitate natural water retention include beaver dams/dam analogs, snow fencing, large woody debris, green infrastructure, and rock structures (e.g., one-rock dams, media luna structures).

#### *Beaver dams/beaver dam analogs*

One nature-based approach to retaining natural water storage is creating and maintaining beaver dams and beaver dam analogs (BDAs). Both tactics impound water and retain sediment, promoting higher water tables and groundwater recharge (Westbrook et al. 2006; Hood & Bayle 2008; Pollock et al. 2015), and increase water storage in surrounding soils (Wild 2011). They are increasingly being used to achieve wetland, stream, and floodplain restoration goals (Dwire et al. 2018). For example, the presence of active beaver lodges significantly increased open water areas in shallow boreal wetlands despite variations in temperature, precipitation, and drought (Hood & Bayley 2008). Beaver dams and ponds enhanced the depth, extent, and duration of inundation during floods, and elevated the water table along the riparian corridor of the Colorado River during both high-flow and low-flow periods, indicating that beaver can support wetland formation and persistence (Westbrook et al. 2006). The dams attenuated water supply declines during drier periods as both surface runoff and alluvial groundwater seepage; the seepage, caused by overbank flooding across the floodplain, raised water tables up to 600 m downstream of one dam's location (Westbrook et al. 2006). Using beaver dams to capture sediment and rebuild streambeds in Oregon increased the area within 0.5 m elevation of the stream by five times, allowing for the reestablishment of riparian vegetation, and creating pockets of cool water behind the dams (Pollock et al. 2007). In Alaska, beaver damming of floodplain spring brooks produced larger coho and Chinook juvenile salmon and more total biomass than beaver-free spring brooks, although the latter had greater survival and densities, demonstrating that the presence of beavers increases habitat variability and a range of potential growth conditions (Malison et al. 2015). The Methow Beaver Project has relocated beaver populations to riparian areas where their dams have created over 260 hectares of wetland habitat (Holmes 2016).

For areas lacking beaver or with low beaver population numbers, there is increasing management interest in mimicking beaver dams through man-made structures—BDAs—to achieve similar ecological and societal benefits and/or to facilitate beaver population recolonization (Pollock et al. 2014). The addition of BDAs increased the number of natural beaver dams in Bridge Creek, Oregon, which were associated with increased juvenile steelhead habitat, increased water residence time, groundwater levels, and summer flow, decreased water temperature in some areas and lower daily temperature fluctuations, increased habitat complexity, and a 175% increase in juvenile production, without impacting upstream migration (Bouwes et al. 2016). Because BDAs modify upstream and downstream landscapes, long-term maintenance is required to maintain ecosystem function and service benefits (Pollock et al. 2015).

BDAs are being used in the Scott River Basin of Northern California to improve instream habitat for endangered coho salmon by raising groundwater levels and surface water flows, and reducing channel incision (Charnley 2018). Since 2014, over ten BDAs have been installed and beavers have begun to take over and make improvements to many of the structures. As part of this project, managers are monitoring the effects of BDAs on species by tracking surface water elevation, groundwater levels and recharge, and fish movement and passage. To date, the following observations have been made: higher surface water levels above the BDAs, increased groundwater levels near BDAs, salmon using habitat upstream of the BDAs at all life stages, restored perennial streamflow above and below BDAs, and cooler water temperatures in ponds created by BDAs (Yokel et al. 2018 *in* Charnley 2018).

### *Large woody debris*

Maintaining or adding large woody debris (LWD) in streams slows water flows, raises water tables, and improves channel complexity (Gippel 1995; Gippel et al. 1996), which facilitates overbank flooding and alluvial groundwater infiltration (Wohl 2014). Placing LWD in low-velocity zones along river channel margins or inside of meanders maximizes increases in water levels (Gippel et al. 1996). The effect of LWD varies depending on the size of the debris relative to the size of the water body. Its accumulation affects the abundance and distribution of fish by providing mesohabitats such as pools and riffles (Crook & Robertson 1999) and its presence creates variety in flow velocity and water depth (Sundbaum & Näslund 1998). Greater amounts of LWD are linked to higher frequency and depth of pools (Beechie & Sibley 1997). The addition of LWD to two streams in coastal Oregon increased overwinter survival of juvenile coho salmon and downstream migrant numbers the following spring, demonstrating that the treatment increased winter habitat (Solazzi et al. 2000).

### *Green infrastructure*

In human communities, green infrastructure approaches—rain gardens, bioswales, grass buffers—can promote groundwater infiltration and recharge and reduce water demand (Gregg et al. 2018). For example, grass buffer strips have been observed to retain more moisture in soil compared to areas without buffers (Cardinali et al. 2014) by decreasing flow and velocity and lengthening water residence time (Evans et al. 1996; Ducros & Joyce 2003). Avellaneda et al. (2017) modeled the hydrologic performance of green infrastructure in Ohio and found that rain gardens, bioretention cells, and rain barrels reduced surface runoff (9%) and increased soil infiltration (7.6%). An ancillary benefit of these green infrastructure techniques is effective pollutant removal and water quality control (Akram et al. 2015; Jiang et al. 2015).

### *Rock structures*

Bill Zeedyk pioneered the use of rock structure techniques, such as one-rock dams and media lunas, derived from Zuni practices to restore hydrologic processes and increase soil moisture (Zeedyk 2009). For example, one-rock dams slow water flows and retain soil, enhancing natural water storage and promoting vegetation growth (Montano Allred 2009), while media luna structures evenly spread sheet flow, reduce erosion, and allow for soil infiltration (Maestas et al. 2018). Wilson & Norman (2018) evaluated the effectiveness of rock structures in restoring vegetation productivity and biodiversity in the arid Cienega San Bernardino wetland in southeastern Arizona and northern Sonora, Mexico over a 33 year period. Vegetation recovery, signified by increased greenness and soil moisture, was positively correlated with restoration sites equipped with these erosion control structures, including up to 5 km downstream and 1 km upstream of the sites. Nichols et al. (2012) also evaluated soil moisture response to rock structures over a three-month period in 2006 in a degraded semiarid grassland in southeastern Arizona. The authors found that both loose rock and wire-bound structures increased soil moisture by increasing the residence time of ponded water, but note that results may vary elsewhere depending on site and species characteristics. Similar results have been documented in Arizona's Santa Rita Mountains (Nichols et al. 2016) and Chiricahua Mountains (Norman et al. 2016), and New Mexico's Valles Caldera National Preserve (New Mexico Environment Department 2017) and Torreon Wash watershed (Matherne et al. 2018).

### Maintain and enhance infiltration, water storage capacity, and/or health of soils (e.g., fuel treatments, biochar application, restricting access to minimize soil compaction)

The ability of soil to retain water depends on several interacting factors including organic matter content, soil texture and depth, and biological activity (Bot & Benites 2005). Maintaining organic carbon in soils improves water retention and water use efficiency, and therefore recovery potential from the effects of ecological drought (Lal 2016; USDA Forest Service 2017b). Some options for improving soil health include fuels treatments, the use of biochar, and restricting access to minimize soil compaction.

Masticated materials from fuels treatments can be used as mulch and layered to help reduce soil water evaporation, increase water availability for deeply rooted trees, and enhance seedling survival (Grant et al. 2013), and cool soils (Owen et al. 2009). Higher soil moisture was detected in masticated sites compared to untreated sites in pinyon-juniper (Owen et al. 2009; Rhoades et al. 2012) and ponderosa pine (*Pinus ponderosa*) forests (Rhoades et al. 2012). Mulching may be particularly feasible and effective as a post-burn treatment within high-valued forests (Grant et al. 2013) and to retain snowpack in alpine and subalpine habitats (Osterhuber et al. 2007).

Interest in applying biochar to soils as a means to both improve soil health and sequester carbon is increasing. Among other benefits, biochar can increase soil water retention (Lehmann et al. 2003; Kammann et al. 2011; Sharma et al. 2018) as it is highly porous and capable of storing water unavailable to plants (Abel et al. 2013). The application of biochar has been linked to enhanced growth and nutrient uptake of drought-stressed plants via improved soil water holding capacity and soil structure (Ali et al. 2017). The addition of biochar may also increase the resistance and resilience of soil microbial habitats to drought (Liang et al. 2014).

The optimal capture and storage of water in soils is determined by infiltration, permeability, and water-holding capacity. Activities such as recreation, forest operations, and grazing can cause soil compaction, restricting the movement and storage of water. For example, Lei (2004) found that repeated recreational activities, such as human trampling and biking, significantly increased soil compaction in a blackbrush (*Coleogyne ramosissima*) shrubland in Nevada's Spring Mountains. Cole (1987) conducted a three-year experimental project across five forest communities and one grassland in Montana to evaluate the effect of trampling on soil health, vegetation cover loss, and species loss, noting that restricting the timing of access may not yield large benefits as maximum impact of trampling occurs rapidly and rotating use between sites may only limit impacts in low-use areas.

## Goal: Facilitate species persistence under drought conditions

### Consider species type, timing, and location in management activities

Restoration of habitat structure, function, and processes continues to be one of the best ways to address both climate and non-climate stressors. However, it is not enough to engage in restoration activities as we have done in the past and, in fact, "restoring" ecosystems to some former state will likely make them ill-equipped to deal with the challenges of climate change. Although habitat restoration activities generally enhance resilience by recovering critical ecosystem functions and services degraded by human activities and natural processes (e.g., storms) (Elmqvist et al. 2003; Gregg et al. 2011, 2017), these activities also need to be reflective of projected climate change impacts (Harris et al. 2006; Choi 2007). For example, planting drought-tolerant native species in areas projected to get drier is a more climate-informed strategy than planting the species that have historically been there under wetter conditions. This paradigm shift in natural resource management includes intentionally considering climate change in the determination of when, where, and how to best implement habitat restoration and protection. For example, climate-driven phenological shifts may cause temporal and spatial mismatches between plants and pollinators (Burkle et al. 2013), and management approaches may therefore need to be adjusted. Increasing the pace and scale of restoration efforts is critical to effectively preparing for, responding to, and recovering from climate change.

Habitat protection efforts should consider representation (or diversity) and redundancy (or multiples) of ecosystems across the landscape to reduce the risk of habitat loss from climate change impacts (Biringer 2003; Joyce et al. 2008) as well as biological and genetic diversity, habitat connectivity, and refugia. Promoting biological and genetic diversity in habitat restoration and protection efforts maximizes the ability of species to survive in different climates and environments (Chmura et al. 2011).

#### *Species diversity*

Scientists and managers are considering the ability of particular species and genotypes to survive under changing climate conditions, including drought. For example, Petrie et al. (2017) modeled ponderosa pine regeneration potential under warmer and drier conditions. The model indicated a projected increase by 50% ± 106% across the West by 2020–2059 due to more seed production and germination. By 2060, regeneration potential is projected to decline by 50% ± 62% due to decreased seedling production and survival, particularly in the Pacific Northwest. In general, species diversity is expected to yield increased resilience to climate change (Heller et al. 2015; Pires et al. 2018). Highlights from the literature include:

- Managing for species diversity, particularly favoring communities that may be better adapted to future conditions (e.g., more drought-adapted species or those known to have broad tolerance ranges), may reduce stand vulnerability to drought and insect outbreaks (Vose et al. 2016a).
- Greater diversity of plant life on rangelands has been linked to improved adaptive capacity to drought (Howery 1999).
- Promoting higher species richness may ameliorate the effects of drought and reduce mortality rates (Klos et al. 2009; Richardson et al. 2010).
- Mixed species forests with high genetic diversity and high percentage of broadleaf species may reduce drought risk in temperate forests in Europe (Spiecker 2003).
- Pretzsch et al. (2013) determined that species mixing of beech (*Fagus sylvatica*) and oak (*Quercus petraea*) have potentially higher tolerance of and resilience to drought than monocultured stands.
- Increasing genetic and species diversity in riparian habitats can provide shade and water storage to support resilience to climate change (Halofsky et al. 2016).
- Species diversity increased biomass yield regardless of management intensity (e.g., mowing, grazing) and generally reduced grassland vulnerability to drought (Vogel et al. 2012).
- Seed collection and storage from different populations across a broad geographic range may conserve genetic diversity (Williams & Dumroese 2013).
- Considering species from a range of climatic and altitudinal gradients broadens the source of seeds and seedlings that may be better adapted to drier conditions (Bernazzani et al. 2012).
- Restoring plant communities through high-density seeding using species-rich seed mixtures increased the success of restoration efforts, enhanced cover of native forbs, and reduced cover of invasive species in a large Nebraska grassland; however, there was little evidence that these treatments influenced drought resistance, recovery, or resilience (Carter & Blair 2012).
- Plant diversity can increase resistance to extreme events (e.g., wet or dry, moderate or extreme, brief or extended) by stabilizing biomass production, particularly during moderate and extreme climate events in grassland ecosystems (Isbell et al. 2015).

Three studies that directly tested how species diversity may influence drought resistance and recovery are Mariotte et al. (2013), Grossiord et al. (2014), and Kreyling et al. (2017). Mariotte et al. (2013) tested the hypothesis that more diverse communities are more likely to be resistant to drought, and found that the presence of a single species enhanced overall community resistance in a grassland community in the Swiss Jura Mountains. Resistance was calculated as the ratio of regrowth biomass during drought to pre-drought biomass. The study tested the effects of drought combined with the removal of subordinate species—smaller species that grow under dominant canopy that are found frequently but with low relative cover (e.g., *Trifolium pretense*, *Achillea millefolium*, *Leontodon* sp., *Veronica chamaedrys*)—and found a tenfold decline in resistance, indicating that the presence of subordinate species can facilitate regrowth of dominant neighbors. Grossiord et al. (2014) evaluated the relationship between tree species diversity and forest habitat resistance and resilience to drought in 160 forest stands from monocultures to mixed forests across five major forest types in Europe (e.g., Mediterranean, hemi-boreal, temperate beech, mountainous beech, thermophilous deciduous). The authors found that higher species diversity did not uniformly improve drought resistance across all forest types, noting that diversity may enable drought resistance only in drought-prone sites, and that local climatic conditions and individual species traits may be more critical to appropriately creating drought-resilient forests. Finally, Kreyling et al. (2017) conducted a biodiversity and drought manipulation experiment to evaluate drought resistance and recovery of aboveground biomass production across five grassland sites in Europe. The study found that species richness did not affect resistance but did improve the recovery of biomass production affected by drought in low-productive, slow-growing plant communities.

In general, managers are advised to consider diversity in life stages, traits, and processes in species collection (Chmura et al. 2011). Age structures of plant communities may confer resilience in forests where more mature trees may be less vulnerable to drought but more vulnerable to pests (Millar et al. 2007). Maintaining a mix of older and younger trees as well as diverse sizes may reduce habitat loss from drought, pests, and diseases (Bernazzani et al. 2012; Baker & Williams 2015). Maintaining species with diverse phenologies so that the timing of life cycle events (e.g., germination, migration) are staggered may improve resilience to climate-driven changes (Schwartz 1999 *in* Bernazzani et al. 2012).

Drought-informed species selection may include prioritizing phenotypic traits that may confer resilience, including water use efficiency, root development, and rapid seedling emergence (Vose et al. 2016a). Plants may exhibit traits associated with drought escape (e.g., early flowering and reproduction), avoidance (e.g., high water use efficiency, high root-to-shoot ratios), and/or tolerance (e.g., root growth, osmotic adjustment, accumulation of sugars) (Kooyers 2015). For example, Laughlin et al. (2017) compared efforts to enhance resilience in southwestern U.S. mixed-conifer forests to wildfire and drought based on different phenotypic trait combinations of thick bark, dense wood, and leaf nitrogen concentration. The authors developed a model to identify and select species based on traits that enable high survival rates in a changing climate. For example, drought resistance may be enhanced by restoring species with dense wood and low to moderate leaf nitrogen concentrations as these species may be able to resist cavitation and tolerate moisture stress. Another computer model—the Restoring Ecosystem Services Tool (REST)—was designed to assist managers in species selection for restoration based on functional traits that best match specific management objectives, such as drought tolerance, successional facilitation, and carbon storage (Rayome et al. 2019).

### *Investigating drought tolerance*

The wide-ranging current and potential future impacts of ecological drought in the Northwest have led to an increase in research efforts aiming to better understand drought tolerance of vegetation present in the region. One major branch of research includes investigating how genetic variation influences drought tolerance across the range of a given species. Researchers are using genecological studies and common garden experiments to investigate how drought-tolerance traits vary among different populations of the same species across latitudinal, longitudinal, and elevation gradients, as well as how different genotypes interact with environmental conditions to affect drought resilience (Bansal et al. 2015b, 2016; Prendeville et al. 2016; Merz et al. 2017).

The most robust drought tolerance investigations to date in the Northwest are for coast Douglas-fir (*P. menziesii* var. *menziesii*). Common garden experiments have shown that while coast Douglas-fir trees from different source populations all exhibit some ability to adapt to drought conditions (e.g., by lowering minimum transpiration), populations differ in their degree of drought tolerance (Bansal et al. 2015b). Specifically, populations from climates with cool winters and arid summers exhibit the greatest drought tolerance (Bansal et al. 2015b), and in general, populations from cool locations (e.g., more northerly latitudes and higher elevations) exhibit both high drought tolerance (Bansal et al. 2015b) and cold-hardiness (Bansal et al. 2015a). Researchers hypothesize that cellular and tissue adaptations evolved in response to winter desiccation stress in cooler environments help trees from these populations better deal with summer drought stress, meaning drought impacts may be less severe at higher elevations and more northerly locations than previously believed (Bansal et al. 2015b). In addition to winter temperature, summer precipitation has also played an important role in the evolution of stress tolerance between different populations, demonstrating how multiple environmental drivers can apply selection pressure that influences future climate resilience (Bansal et al. 2015a, 2015b, 2016). Compared to populations from cooler locations, populations from warmer and drier climates exhibited superior drought tolerance only under the warmest and driest experimental conditions (Bansal et al. 2015b), but these drought adaptations resulted in trade-offs with cold-hardiness (Bansal et al. 2016). These findings indicate that while populations on the trailing edge of current coast Douglas-fir species distribution may be more resilient to increasing drought stress than previously believed (Bansal et al. 2015b), individuals sourced from these populations could experience some cold damage if transplanted to cooler environments (Bansal et al. 2016).

Similar common garden and genecology experiments are currently underway for bluebunch wheatgrass (*Pseudoroegneria spicata*), a native perennial bunchgrass species commonly used in restoration activities across the Intermountain West (Prendeville et al. 2016, 2017; USDA Forest Service 2018b). Bluebunch wheatgrass populations are known to genetically differ in several traits that affect adaptation to temperature and precipitation (St. Clair et al. 2013). Current experiments are examining how seeds from different areas respond to environmental conditions across a range of regional climates (e.g., hot/dry conditions on the Columbia Plateau vs. cool/wet conditions in Blue Mountains vs. cool/dry conditions east of the Blue Mountains) (Prendeville et al. 2016). This research intends to identify traits and climatic factors that affect establishment, growth, and reproduction, and how responses vary between different bluebunch wheatgrass populations, which can ultimately be used to make climate-informed restoration decisions, as well as to inform assisted migration efforts (Prendeville et al. 2016).

There have also been preliminary investigations into the drought tolerance of Garry oak (*Q. garryana*) seedlings in Washington State. Researchers have found that seedlings sourced from different sites exhibit variable responses to imposed drought stress. However, because the genetic relatedness of Garry oaks from different sites is unknown, researchers have not yet been able to determine whether differences in drought response are due to genetic variation between seedlings or plastic responses by individual trees to environmental stress (Merz et al. 2017).

Mclaughlin & Zavaleta (2012) and Mclaughlin et al. (2014) evaluated the tolerances of valley oak (*Q. lobata*) and blue oak (*Q. douglasii*) in California to drought conditions. Oak distribution is closely linked to precipitation amount and timing, which affects seedling recruitment, growth, and germination of oak species. Drought conditions are likely to increase adult tree mortality and may decrease seedling recruitment outside of cool, wet microenvironments, potentially leading to range contractions (Mclaughlin et al. 2014). Saplings are acutely sensitive to drought and their narrower window of suitable climatic conditions (e.g., more sensitive to warmer temperatures and moisture stress) may prevent their regeneration in areas projected to remain suitable for adults (Mclaughlin & Zavaleta 2012).

Information generated from genetic studies may help inform a variety of future adaptation efforts, such as climate-informed stock selection for restoration (Bansal et al. 2015b; Prendeville et al. 2016; Merz et al. 2017) or timber stand replanting efforts (Watts 2015), as well as assisted migration (Bansal et al. 2015a, 2015b, 2016; Prendeville et al. 2016). Better understanding of genetic variation in drought tolerance can also provide finer-scale pictures of vulnerability; for example, research on coast Douglas-fir has shown that populations at both the leading and trailing edges of current species distribution are more drought-resilient than previously believed, indicating that range shifts may occur at a slower rate than initially projected (Bansal et al. 2015b).

### *Location and timing*

Limited studies discussed changing the location and timing of seeding and planting. In terms of location, Poulos et al. (2007) found a close association between the drought response and elevational distribution of two oak species (*Q. laceyi* and *Q. sideroxyla*) in the Sierra del Carmen, Coahuila, Mexico. While both species exhibit drought tolerance, *Q. laceyi* is found at hotter, drier sites at lower elevations while *Q. sideroxyla* avoids drought stress at higher elevations. Likewise, Leverkus et al. (2015) found that oak seedlings planted at higher elevation sites in Spain's Sierra Nevada had increased survival due to reduced drought stress in cooler conditions. On moisture-limited sites, tree shelters may be used to improve seedling survival potential (Taylor et al. 2009 *in* Vose et al. 2016b). Managers may also need to alter the timing of planting to ensure optimal growing conditions for species. This includes adjusting the prioritization of management and restoration activities so as to first repair the most vulnerable habitats (Palmer et al. 2009) and to not disturb vulnerable areas during drought (Vose et al. 2016b).

## Create or enhance water supply

### *Wildlife water developments*

Wildlife water developments help to ameliorate the loss of naturally-occurring water sources for the benefit of wildlife, game species, and livestock. Examples of wildlife water developments include natural rock basins (tinajas), artificial catchments (e.g., guzzlers, earthen reservoirs), developed springs, and wells (Rosenstock et al. 1999; Krausman et al. 2006). These structures reduce the distance animals need to travel for water access (Bailey 2005) and their distribution can influence species' ranges (Bailey 2005; Whiting et al. 2009). For example, mule deer (*Odocoileus hemionus*) in Arizona were able to use previously unsuitable habitat after the construction of water developments (Rosenstock et al. 1999). Bleich et al. (2010) evaluated the effect of wildlife water developments on habitat quality for mountain sheep (*Ovis canadensis*) in three mountain ranges in the Sonoran Desert, and determined that additional surface water provisioning could increase the availability of high-quality habitat. Mountain sheep populations may therefore be able to persist in arid and low-elevation sites that are most vulnerable to drought.

Water developments are also used to benefit livestock during drought, particularly constructed ponds developed from groundwater sources (Wallander et al. 2013), and concentrate livestock and grazing pressure within specific areas (Bailey 2005). For example, cattle altered their grazing patterns when a solitary water development was constructed within a pasture in eastern Oregon (Ganskopp 2001 *in* Bailey 2005), and when water was pumped into a tank further away from a stream in northeastern Oregon (Porath et al. 2002 *in* Bailey 2005).

In some areas, wildlife water developments are fenced to exclude feral species and livestock; however, these exclosures also limit use by wild ungulates such as pronghorn (*Antilocapra americana*) and mule deer, and Larsen et al. (2011) recommend avoiding their use where possible. Additional concerns about these developments include potential increased competition between wild and domestic animals, direct mortality of animals that become trapped, and reduced flushing resulting in degraded water quality, although these negative effects are poorly studied and understood (Rosenstock et al. 1999; Whiting et al. 2009; Simpson et al. 2011). In addition, these structures require capital investment and long-term maintenance (Bailey 2005; Krausman et al. 2006).

### *Constructed wetlands*

Constructed or artificial wetlands are typically created to treat wastewater and improve water quality (Greenway 2005; Vymazal 2010), however animals such as birds and amphibians also use these wetlands as opportunistic habitat (Gelt 1997; Levy 2015). Studies from North Carolina (Petranka et al. 2007) and Oregon (Pearl & Bowerman 2006) demonstrate that amphibians can quickly colonize constructed waterbodies and support population persistence, particularly during periods of environmental stress. Artificial habitats have also been constructed to support threatened fish populations in the Murray-Darling Basin of Australia, particularly within heavily modified, water-limited reaches (Hammer et al. 2013).

### *Forest canopy manipulation, snow fencing and harvesting*

Snowpack is an important natural water storage reservoir, supplying water for streamflow, soil moisture, and groundwater recharge, and snow droughts can have severe ecological consequences (Dierauer et al. 2019). Measures to reduce snowmelt rates or increase snow accumulation on the ground or in forested watersheds include forest canopy manipulation as well as snow fencing and harvesting.

Forest canopy structure controls snow accumulation by intercepting snowfall and attenuating wind speeds (Woods et al. 2006; Vose et al. 2016b; Roth & Nolin 2017). If the snowfall sublimates—or converts from solid to gas (or ice to water vapor)—before falling to the ground, less water is available in the snowpack (Montesi et al. 2004). Fire management activities may increase or decrease snow accumulation. For example, fire suppression increases or maintains canopy cover, which can reduce snow accumulation on the ground (Matheussen et al. 2000 *in* Woods et al. 2006), while thinning treatments have been linked to enhanced snow accumulation in snow-dominated coniferous forests in the West (Meiman 1987 *in* Vose et al. 2016b). Woods et al. (2006) examined the hydrologic effects of thinning on lodgepole pine (*P. contorta*) stands in Montana and found that the spatial arrangement of trees also affects snow accumulation. The study used two treatments replicating the effects of wildfires and the removal of ~50% of the stand basal area; the first treatment mimicked a low-intensity mixed-severity wildfire wherein many trees survived in an even distribution, while the second treatment replicated a stand-replacing wildfire where trees were killed in large swaths while others remained unburned in distinct groups. Snow accumulation rates increased in the first treatment but no significant change was observed in the second treatment, likely due to differences in canopy interception and sublimation. Creating forest openings in the Rocky Mountains increased snow accumulation and reduced snowmelt rates by over two weeks (Troendle 1983).

Snow fences are structures that force drifting snow to accumulate in a specific area. These barriers can be structural (e.g., horizontal or vertical slats) or natural (e.g., grasses, shrubs, trees) (USDA National Agroforestry Center 2011). Snow fences keep snowdrifts from accumulating on roadways while collecting snow to improve localized water sources and to recharge soil water content (Jairell & Schmidt 1991). Vegetation barriers can enhance snow retention, increase soil moisture, and extend water yield availability; sudangrass (*Sorghum* x *drummondii*), sorghum (*Sorghum bicolor*), and tall wheatgrass (*Thinopyron ponticum*) have all been used to trap snow (Renton et al. 2015). Snow fencing and harvesting can maintain soil moisture and high live fuel moisture, which may reduce wildfire risk (Grant et al. 2013).

Studies have linked snow fencing to increased soil moisture in Wyoming sagebrush communities and enhanced water supply for lakes in Alaska. David (2013) assessed how snow fencing could increase soil moisture and improve native species re-establishment at the Jonah Natural Gas Field in southwest Wyoming. Efforts to restore sagebrush (*Artemisia tridentata*) were significantly improved with the use of the Hollow Frame Fence System (HFFS) to accumulate snow and increase soil moisture. Within the fenced area, significant increases in snowpack density, spring soil moisture, and *Artemisia* establishment were observed. The HFFS maximizes snow capture, minimizes sublimation, and evenly distributes subsequent moisture supply across the fenced area. Sturges (1992) evaluated the effects of snow fencing on a sagebrush rangeland watershed in south central Wyoming and determined that snow water storage and snowmelt runoff increased by 78% and 129%, respectively. Snowmelt also increased soil infiltration and groundwater recharge. Finally, Stuefer & Kane (2016) examined the use of snow fencing to augment water supply for lakes on Alaska's Coastal Plain. A fence was placed adjacent to an experimental lake for two winter seasons (2009-2010 and 2010-2011) and the snowdrift provided meltwater, resulting in an increase of 21–29% of lake water volume, which offset summer evaporation losses.

### Protect vulnerable species through assisted migration and improved habitat connectivity

Warming air temperatures and changes in precipitation are already driving species outside of their historic ranges (Parmesan & Yohe 2003; Pounds & Puschendorf 2004; Rosenzweig et al. 2007; Crimmins et al. 2011; Monleon & Lintz 2015). Assisted migration—also known as assisted translocation—is a tool to facilitate the intentional movement of an organism from its historic range to more favorable climatic habitat conditions (Mueller & Hellmann 2008; Ricciardi & Simberloff 2009; Camacho 2010; Vitt et al. 2010; Chmura et al. 2011; Lawler & Olden 2011). Assisted migration allows for the movement of species with slow dispersal rates (e.g., snails, some plants) or those unable to move on their own with no suitable bridge or migration corridor to other potentially favorable habitats (e.g., species on remote islands or isolated lakes). Improving habitat connectivity to facilitate species movement across landscapes in response to climate change is also an important adaptation strategy (Littlefield et al. 2017), particularly for fish species (Lyon et al. 2010). For example, removing barriers to fish passage in streams or creating fish-friendly water crossing structures allows fish to move out of drought-stressed areas (Palmer et al. 2009; Beechie et al. 2013; Wilhere et al. 2017).

#### *Assisted migration*

Several papers discuss efforts to translocate species out of drought-stressed areas. With respect to plant species, current seed transfer guidelines lack climatically-informed temporal and spatial considerations to assist decision makers (Chmura et al. 2011; Williams & Dumroese 2013; Vose et al. 2016a). One terrestrial example from Gray et al. (2011) notes three conditions for assisted migration in aspen reforestation programs: (1) evidence of a species-specific adaptational lag (characterized by a mismatch in rapid environmental change and slow evolutionary response), (2) observed biological impacts on forest health and productivity, and (3) robust bioclimate envelope projections to support translocation decisions. One of the authors' key recommendations is for aspen reforestation efforts to focus on drought-tolerant species and genotypes in a changing climate. McLane & Aitken (2012) successfully tested the ability of whitebark pine (*P. albicaulis*) to establish outside of its current range by planting seeds from seven Oregon and Washington populations in eight different locations in northwestern British Columbia. Isaac-Renton et al. (2018) evaluated the drought tolerance of lodgepole pine, by comparing the results of an extensive provenance trial and transplant experiment in western Canada with tree-ring analyses to evaluate genetic responses between different populations to extreme events, including a 2002 drought. The study found that northern populations of lodgepole pine are highly vulnerable to water stress due to thin xylem cell walls and the inability to modify stomatal behavior in response to drought, both of which increase the risk of cavitation, while southern edge populations demonstrated the highest drought tolerance and could be good candidates for northward transplanting.

A nationwide series of experiments are currently underway to evaluate adaptation options (resistance, resilience, and response treatments) across different forest types—the Adaptive Silviculture for Climate Change project (https://www.adaptivesilviculture.org). One such site is the western larch (*Larix occidentalis*)-dominated forests located at the Flathead National Forest/Coram Experimental Forest in northwestern Montana. The response treatment that will be tested at this site aims to create forest stands that are more tolerant of a warmer, drier future climate by introducing ponderosa pine, which is currently found at warmer, drier locations at different elevations than the experimental sites and is more drought-tolerant (Crotteau et al. 2019). The treatment plan involves active management via thinning and cutting to reduce risks associated with wildfire and blister rust, while adopting an adaptive management approach to make adjustments as the climate changes. These adjustments may include introducing additional species or actively facilitating a transition to a hardwood forest or woodland if western larch can no longer survive.

The majority of assisted migration examples can be found with respect to aquatic habitats under drought stress. Fish rescue and reintroduction programs have been conducted to varying degrees of success and some general principles are accepted; for example, translocation of fish species are most effective if done during wet periods (Crook et al. 2010). Two well-known examples evaluating the assisted migration of freshwater fish in response to drought come from the southern Murray-Darling Basin in Australia. Three translocations have been undertaken for southern pygmy perch (*Nannoperca australis*), hardyhead (*Craterocephalus fluviatilis*), and river blackfish (*Gadopsis marmoratus*) (Hammer et al. 2013). Pygmy perch were transferred from a shallow pool to a deeper pool upstream, however, subsequent monitoring indicates that the species has been extirpated from the site. Hardyhead were translocated to a managed wetland with high initial success in terms of recruitment and survival, although the site was subsequently inundated during a flood. The third translocation of river blackfish was deemed to be successful as the donor pool site dried while the translocation site five kilometers upstream retained water. The limited effectiveness of these translocations was likely due to a combination of the rapid onset and scale of critical water shortages in the region as well as a lack of prior planning.

Hammer et al. (2015) also reviewed efforts to recover the southern-purple spotted gudgeon (*Mogurnda adspersa*) in the Murray-Darling Basin. The species was considered to be completely extirpated until 2002, when a single remnant population was discovered in a small wetland. Soon after, the wetland habitat completely dried as a result of extensive drought, flow regulations, and water diversions. In response, a fish rescue operation captured 50 individuals before the habitat completely dried out with the intention of reintroduction to the river after the drought. A long-term captive breeding program was required to rescue the fish from extinction due to the multi-year drought and subsequent desiccation and limited recovery of aquatic plants. In general, captive breeding and reintroduction should be viewed as a last resort as on-site restoration and other improvements better protect intrinsic functional and evolutionary links between species and habitat (Frankham et al. 2010 *in* Hammer et al. 2013). The reintroduction plan prioritizes sites that maintain natural freshwater flows during long-term low-flow periods and sites that will likely respond positively to environmental watering efforts as required (Hammer et al. 2012).

Since 2014, the California Department of Fish and Wildlife (CDFW) has engaged in fish rescue and reintroduction efforts to protect native populations from drought (CDFW 2019). For example, endangered coho salmon (*Oncorhynchus kisutch*) populations in California are at increased risk of extirpation due to severe drought. Fish rescue was first proposed in 2012 to save the population from Redwood Creek and efforts began in August 2014 to capture, rear, and release juvenile fish. Juvenile coho were captured, genetically analyzed, and reared at the Don Clausen Fish Hatchery on Lake Sonoma, and between 2016 and 2018, the hatchery-reared adult coho salmon were released back into Redwood Creek to spawn. Success of this and other fish rescue efforts in response to drought are being monitored by CDFW staff.

Other monitoring efforts have demonstrated less success for fish translocations in California. Arriaza et al. (2017) evaluated size-conditional smolting and marine survival of the Carmel River steelhead (*Oncorhynchus mykiss*) population, which has been under intensive restoration for over 20 years. The authors determined that translocations likely reduced smolt success and had minor effects on juvenile growth rates, and that recovery efforts would be more effective by focusing on conditions to improve growth in the river (e.g., riparian restoration to lower water temperatures and slow flows) rather than in captivity.

The legal feasibility of assisted migration depends upon the interaction of three factors—species type, responsible party, and land ownership of current and future species locations. For example, the introduction of non-native species such as cows and sheep are allowed on Bureau of Land Management lands for grazing, while the introduction of non-native plants on these same lands is more closely regulated and native species are preferred (Camacho 2010).

Risks associated with assisted migration include potentially high administrative costs, translocation failure, harm to biodiversity and ecosystem health at new sites (e.g., *Pueraria lobata* [kudzu vine] was introduced as an ornamental plant and to support erosion control in the Southeast United States but crowded out native species [Everest et al. 1999]; hybridization of non-native and native species and the introduction of diseases and parasites [Hoegh-Guldberg et al. 2008; Ricciardi & Simberloff 2009]), and species extirpation (Williams & Dumroese 2013). These risks tend to increase with distance of transfer (Mueller & Hellmann 2008).

### *Habitat connectivity*

Two studies discussed habitat connectivity and species persistence in the context of drought. Epps et al. (2006) evaluated how elevation, habitat connectivity, and climate change influence genetic diversity of desert bighorn sheep (*Ovis canadensis nelsoni*). Lower-elevation populations were found to have lower genetic diversity and were also subject to higher temperatures, lower precipitation, and fewer stable spring sources when compared with higher-elevation populations, and decreased population connectivity was linked to lower genetic diversity. Therefore, genetically diverse populations of desert bighorn sheep in high-elevation habitats with high connectivity may represent genetic refugia during droughts. Measures to restore connectivity include translocations and installment of wildlife corridors across highways.

O'Farrill et al. (2014) evaluated how climate-driven changes in water availability affect habitat connectivity for three large endangered species in the Greater Calakmul Region of the Yucatan Peninsula, Mexico. The three species assessed include Baird's tapir (*Tapirus bairdii*), white-lipped peccary (*Tayassu pecari*), and jaguar (*Panthera once*), all of which rely on freshwater from waterholes and seasonal streams. The authors compared the effects of 10 drought scenarios on waterhole connectivity for each species inside and outside of the Calakmul Biosphere Reserve, and found that drought significantly affected spatial distribution and connectivity of waterholes and therefore potential suitable habitat for all three species. The study determined that maintaining a network of small waterholes inside and adjacent to the reserve will best support species under water stress.

### Identify and protect drought refugia

Identifying and protecting climate refugia—areas likely to maintain more stable climatic conditions over time—supports species persistence in a changing world. Temperature and moisture are two factors that dictate the spatial and temporal distribution of climate change refugia (Morelli et al. 2016). Refuge types depend on the habitat and species in question; for example, in aquatic systems, pools, waterholes, ephemeral and permanent streams, logs, riffles, and riparian vegetation all can sustain moisture or water levels to combat the effects of drought on aquatic species (Bond et al. 2008).

Cartwright (2018) evaluated how specific landscape topography and soil features can create refugia in lodgepole and whitebark pine (*P. albicaulis*) forests. Topographically shaded slopes, thinner stands, areas of low bulk-density soils, valley bottoms, and riparian areas were associated with higher and longer soil moisture retention rates compared with other sites. Mackey et al. (2012) mapped vegetated areas that likely functioned as drought and fire refugia between 2000–2010 in Australia, finding that areas with topographic sheltering and shading and reliable surface or subsurface water flow created or maintained moister conditions. Mesic areas within dry forest habitats, such as gullies, serve as drought and fire refugia for stressed species (Mac Nally et al. 2000). Elevated water supply alone is not sufficient to identify drought refugia as hydrologic characteristics of mesic sites must also meet biological requirements; for example, how much water, the timing of availability, and its form dictate where a given species can persist (Mclaughlin et al. 2017). Managers should also consider balancing other effects of climate change, including thermal stress, wildfire, and floods (Mackey et al. 2012; Mac Nally et al. 2010; Mclaughlin et al. 2017).

Intact floodplains may act as drought refugia as they typically feature cool, mesic microclimates with stable water availability from groundwater sources and flooding. Selwood et al. (2015) evaluated how floodplains may serve as refugia by assessing the effects of a multi-year drought on terrestrial birds in floodplains and adjacent areas in Australia. The authors found that fewer species declined in floodplains (19%) than non-floodplains (29%) with specific species—white-naped honeyeater (*Melithreptus lunatus*), grey currawong (*Strepera versicolor*) and golden whistler (*Pachycephala pectoralis*)—using floodplain forests as refugia over the course of the 13-year drought.

Hyporheic zones—saturated areas of streambeds and banks where surface water and groundwater mix—are habitat for microbes, invertebrates, and fish (Woessner 2017) and may also be drought refugia for macroinvertebrates (Wood et al. 2010). Wood et al. (2010) found that during a severe drought on the Little Stour River in the United Kingdom, benthic fauna used the hyporheic zone as a refugium from the combined effects of low flows, increased water temperatures, and decreased dissolved oxygen levels. Similar studies have resulted in conflicting findings about the use of hyporheic zones as refugia; for example, no evidence was found on benthic taxa use of the hyporheic zone under low-flow conditions in South Carolina, California, and the Sonoran Desert (Stubbington 2012). However, this lack of consistency may be due to different flow characteristics (e.g., intermittent vs. perennial, floods vs. droughts) and physical characteristics of the substrates (e.g., volume of fine sediment) in the streams within different study areas. Sedell et al. (1990) also discuss the role of hyporheic zones as drought and thermal refugia, noting that diapausing stonefly nymphs have been observed as deep as 25 cm in the substrate of a dry streambed.

Sedell et al. (1990) recommend identifying and conserving refugia in river systems based on a hierarchical geomorphic and spatial system. For example, refugia at smaller scales (e.g., side channels) may be less resistant to disturbance than refugia at larger scales (e.g., wide floodplain areas, oxbow lakes), which may be more resistant and resilient to disturbances. However, small-scale refugia such as deep pools have been observed to allow fish to persist during drought conditions in an intermittent stream in Iowa (Pauloumpis 1958 *in* Sedell et al. 1990).

The lack of geospatial data on certain refugia, such as springs, may limit conservation measures. Cartwright & Johnson (2018) used remote sensing and climate data to identify and assess the potential of different types of springs to act as drought refugia. Springs are sources of habitat, surface water for wildlife, and soil moisture, and can vary widely in terms of hydrology and chemistry within a watershed. This study assessed surface-moisture zones associated with springs in a montane sage-steppe habitat, and found that the most resilient sites featured cool temperatures and topographic shading and sheltering common among higher elevation, north- and northeast-facing slopes. Additional resilient surface-moisture zones were identified immediately below snowbanks that provide stable recharge for high-elevation springs.

Man-made water bodies—drainage ditches, irrigation pipes, canals, and wastewater treatment and decommissioned industrial ponds—could be particularly beneficial as refugia if natural water systems are insufficient to support fish and wildlife. For example, fish in the Rio Grande

Basin have been observed using irrigation canals as drought and thermal refugia when adjacent rivers are drying and warm; these canals also serve as corridors through which fish and amphibians can migrate, including both native and non-native species (Cowley et al. 2007). In evaluating the potential of man-made water storage bodies to serve as refugia, Halliday et al. (2015) point to three main criteria: the water regime; water quality; and species' habitat requirements. With respect to the water regime, man-made water storage sites are relatively permanent and stable compared to natural systems that are more susceptible to drying. Water quality in some man-made sites (e.g., wastewater treatment ponds) may limit the ability of species to utilize these areas as habitat. In a comparison of man-made and natural water bodies, few differences were identified in the potential ability of both areas to support fish, frogs, and zooplankton during drought and other disturbances, however, habitat enhancement via increased vegetation and substrate diversity would be beneficial (Halliday et al. 2015). In addition, the utility of these sites may be constrained by natural dispersal corridors and human interventions and translocations may be necessary.

## Goal: Reduce stress on ecosystems and species

### Reduce tree density and fuel loads through thinning and prescribed burns

Water shortages are driving increased moisture stress, tree mortality, and fuel flammability in forests (Gregg et al. 2016; Littell et al. 2016). Thinning and prescribed fire are common tools used to reduce tree stand densities and fuel loads in forests. Thinning increases tree vigor by strategically reducing competition and therefore increasing resistance and resilience to moisture stress and infection by pests and pathogens (Bréda & Badeau 2008; Klos et al. 2009; Swanston and Janowiak 2012; Scott et al. 2013; Kershner et al. 2015; Halofsky et al. 2016; Bradford & Bell 2017; Gleason et al. 2017). Halofsky et al. (2016) note that managers can alter thinning treatments to be more effective by focusing on areas where summer drought impacts will be most prominent, in high-value habitats (e.g., riparian zones), and in high-risk locations (e.g., Wildland Urban Interface).

Prescribed fire reduces the risk of catastrophic or stand-replacing fire by targeting and reducing surface and ladder fuels, allows for the re-introduction of natural fire regimes, and prepares the landscape for the re-establishment of fire-tolerant native species that may be better adapted to shifting fire regimes (Spies et al. 2010; Gregg et al. 2016). However, extended droughts can reduce opportunities to use prescribed fire as particularly dry conditions can make fires difficult to control. In general, more heterogeneous forests are typically more resilient to drought, wildfire, and disease (Ahlqvist & Shortridge 2010; Spies et al. 2010; Peterson et al. 2011; Scott et al. 2013; Kane et al. 2017; USDA Forest Service 2017b).

In general, reducing stand density can limit drought stress, fire risk, and susceptibility to disease and insect outbreaks (Chmura et al. 2011), although there are some inconsistent findings in the literature regarding the effects of thinning and prescribed fire on individual tree health and mortality, stand health and competition for water, soil moisture and water yield, and understory vegetation.

*Effects on individual tree health and mortality*

- Thinned trees typically require less water and may therefore be less vulnerable to water stress (Ruzicka et al. 2017).
- Zausen et al. (2005) studied the long-term effects of thinning and combined thinning and prescribed fire treatments on ponderosa pine tree and stand susceptibility to water stress and bark beetles. Thinning in particular was effective in increasing resistance and reducing tree mortality to drought and beetles by increasing individual tree growth and resin production, increasing leaf nitrogen concentration and photosynthetic capacity, and reducing tree water stress.
- Thinning may increase leaf-to-sapwood area ratios, which can cause increased individual tree water demand (Kolb et al. 2007; Carnwath et al. 2012).
- Neither growth nor drought vulnerability were significantly affected by repeated burns in forests dominated by red pine (*P. resinosa*) in northern Minnesota (Bottero et al. 2017).

*Effects on stand health and competition*

- Reducing the density of fire-suppressed stands may also reduce the risk of drought-induced mortality (Young et al. 2017).
- Prescribed fire reduced tree mortality and stand density and increased resistance to drought in a comparison of burned and unburned low-elevation mixed-conifer forests in Yosemite, Kings Canyon, and Sequoia National Parks (van Mantgem et al. 2016).
- Basal area reduction via thinning and other forest management activities may be particularly useful over the near-term in limiting drought-induced mortality (Bradford & Bell 2017). By maintaining basal area at low rates (20% of current area), managers may be able to prevent projected mortality increases of:
    - 70–86% by 2050s and 31–54% by 2080s of Douglas-fir;
    - 42–70% by 2050s and 25–51% by 2080s of ponderosa pine; and
    - 30–55% by 2050s and 16–34% by 2080s of pinyon pine.
- Thinning generally increased drought resistance and resilience, resulting in larger trees, in a long-term experiment in Minnesota's red pine forests; however, over the long-term, lower-density stands with large trees exhibited reduced drought tolerance due to higher water demands (D'Amato et al. 2013).
- Thinning supports drought resistance and resilience by increasing growing space for trees (Larson et al. 2012), including larger root systems (Andre 1957; Kohler et al. 2010), which allow for greater access to groundwater (Dawson 1996; Wang et al. 2018).
- Thinning reduced competition for moisture but did not sufficiently reduce tree mortality in mixed-conifer and ponderosa pine forests in northern Arizona during a severe 10-year drought (Ganey & Vojta 2011).
- Thinning and prescribed fire have reduced competition in overcrowded ponderosa pine forests (Gitlin et al. 2006).
- Thinning to limit crown competition can also reduce canopy interception, which increases the amount of precipitation that is capable of reaching the forest floor (Aussenac 2000).

- Thinning and prescribed burns to reduce stand density and competition may promote drought resistance in ponderosa pine and Douglas-fir forests, although success appears to be dependent on treatment intensity, drought persistence, tree characteristics, and species (Vernon 2017). For example, basal area was only reduced by 34% and some competitors were left to maintain species diversity in the Whiskeytown National Recreation Area of northern California. Overall growth responses to treatments were not strongly significant, but may have been more so if thinning treatment intensity had been more intensive. As drought conditions persisted, resistance appeared to decline in both treated and untreated stands. The study also determined that lower drought resistance was more common amongst larger trees, trees experiencing higher competition, and Douglas-fir species.
- Many thinning projects conducted to resist moderate- to high-severity wildfires have reduced the abundance of small trees and diverse tree species (Baker & Williams 2015). Ensuring the maintenance of both abundance and diversity of trees with small trees dominant may best support dry-forest drought resilience in the western United States rather than just retaining larger, more mature trees (Baker & Williams 2015). For example, Carnwath et al. (2012) concluded that thinning small-diameter trees that grow beneath the dominant canopy increased stand vulnerability to drought by removing the most resilient canopy class.

### *Effects on soil moisture and water yield*

- Thinning can increase soil moisture per tree or per leaf area (Adams et al. 2012).
- Thinning of ponderosa pines in the Coconino National Forest increased soil water content from May to August and improved the condition of remaining trees by increasing canopy growth and nutrient uptake (Stone et al. 1999).
- Thinning increased soil moisture levels in balsam fir (*Abies balsamea*) stands in Canada (Thibodeau et al. 2000).
- Removing pinyon and juniper trees encroaching on sagebrush habitats in the Great Basin with prescribed fire or mechanical thinning was successful in lowering fuel loads, and increased the availability of soil moisture up to four years post-treatment (Roundy et al. 2014).
- In a comparison of prescribed burns and thinning treatments over 30+ years for western juniper control across 77 sagebrush steppe habitats in Idaho, Oregon, and northern California, Davies et al. (2019) found that prescribed burning was more effective at controlling juniper encroachment and encouraging sagebrush dominance over the long term than thinning, but results in more immediate loss of sagebrush and sage-grouse habitat. Both prescribed burning and thinning are effective at increasing soil nutrient and water availability, although this may lead to invasion of exotic annual grasses, such as cheatgrass (*Bromus tectorum*), Japanese brome (*Bromus japonicus*), and medusahead wildrye (*Taeniatherum caput-medusae*).
- Clark et al. (2018) evaluated data from a prescribed fire conducted in 2002 in the Reynolds Creek Experimental Watershed in southwestern Idaho, and found that vegetation type, tree height, percent bare ground, and fire type were the most

significant factors influencing western juniper mortality from fire and could reliably be used when planning fire prescriptions.

- Prescribed burning had no significant effect on soil moisture in a comparison of soil greenhouse gas exchange between burned and unburned plots in a southeastern Australia eucalypt forest (Zhao et al. 2015).
- Prescribed burns generate soil heat, which can reduce water infiltration and may create hydrophobic soils; managers are advised to burn when soil volumetric moisture content is >20% to limit heat penetration (Busse et al. 2014).
- Thinning of at least 15-30% of the forest canopy is required to see any measurable effect on peak flows and water yields (Troendle & Leaf 1980 and Bosch & Hewlett 1982 *in* MacDonald & Stednick 2003). Thinning of 25-30% of the forest canopy may also slightly increase snowmelt peak flows and water yields in alpine and subalpine areas (MacDonald & Stednick 2003).
- Burning treatments decreased soil moisture and total combined above-ground biomass for trees and shrubs but increased herbaceous cover and production in Wyoming sagebrush habitat (Davies et al. 2007).
- Burning increased soil moisture and native perennial biomass and seed production at low-elevation sites in the Great Basin, limiting *Bromus* invasions (Chambers et al. 2007).

### *Effects on understory vegetation*

- Regeneration treatments may indirectly increase stand vulnerability to drought by increasing evaporative losses and understory competition for soil moisture (Nilsen et al. 2001 *in* Vose et al. 2016a).
- Thinning to reduce overstory density increased cover and diversity of understory species in a study of the relationship between thinning and food availability for wildlife in Douglas-fir forests in western Oregon (Neill & Puettmann 2013). These understory species were also found to be drought-, fire-, and heat-tolerant, indicating that some wildlife habitat services can be maintained under changing climate conditions.
- Declines in understory vegetation cover and diversity were observed post-thinning of Douglas-fir and ponderosa pine forest stands in British Columbia (Page et al. 2005). Drought conditions likely contributed to the observed declines by preventing the re-establishment and recovery of understory vegetation. The authors conclude that two or more full growing seasons are needed to recover undergrowth cover and biomass following drought, and recommend limiting fuels treatments during drought years or conducting treatments on snow-covered soils.
- Thinning and burning increased species richness in understory vegetation in a study of restoration treatments in central Washington dry conifer forests (Dodson et al. 2008).

### *Maintaining organic debris*

Maintaining organic debris on forest floors, such as tree limbs and trunks, snags, and leaves, limits soil moisture loss and soil erosion, promotes nutrient cycling, and provides food and habitat for wildlife (Schnepf et al. 2009). A study in a temperate Australian woodland found that coarse woody debris supports understory plant growth and survival by promoting higher moisture content; larger diameter woody debris had a greater effect on maintaining moisture content and protecting understory vegetation from the effects of drought (Goldin & Brookhouse 2015). Woody debris may also act as seedbed substrate in forests that have shallow soils with limited water-holding capacity. In a study of subalpine Norway spruce (*Picea abies*), woody debris water stores were able to provide sufficient moisture for seedling recruitment (Pichlerová et al. 2013).

### Use exclosures and fences to protect groundwater-dependent habitats and associated species

Wildlife exclosures and livestock fences can be used to protect highly sensitive groundwater-dependent habitats (USDA Forest Service 2017b). Groundwater-dependent habitats, such as seeps, springs, and riparian areas, support functional hydrologic regimes (Dwire et al. 2018) and high biodiversity (Gibert et al. 2009), including rare species (Frost et al. 2004), mosses and invertebrates (Virtanen et al. 2009), and springsnails (Frest & Johannes 1999), and provide cool water for spawning and rearing of salmon (Ziemer & Lisle 1998). These habitats are highly sensitive to ecological drought as well as browsing and grazing intensity (Jackson & Allen-Diaz 2006). Wildlife exclosures and livestock fencing reduce access to groundwater-dependent habitats and help reduce damage in and adjacent to water sources (Renton et al. 2015; Dwire et al. 2018).

Areas with livestock exclosures may be able to capture and store more water than grazed sites. Kauffman et al. (2004) compared the effects of livestock exclosures on dry (grass and forb-dominated) and wet (sedge-dominated) meadow communities along the Middle Fork John Day River in Oregon. In exclosed sites, total belowground biomass, soil pore space, and mean infiltration rates were higher in both dry and wet meadows. The study determined that if livestock were excluded from the entire area, the surface 10 cm of soils in the meadows could store $16.6 \times 10^6$ L more of water.

Fencing can reduce grazing and browsing pressure on drought-stressed species. In the Book Cliffs area of Utah, aspen forests (*Populus tremuloides*) exhibit low resilience to moisture stress and other environmental changes, which are exacerbated by elk and livestock browsing intensity (Rogers & Mittanck 2014). Fencing has been effective in reducing browsing impacts in smaller aspen stands, riparian habitats, and recreational locations in the region, but fencing across larger areas is less feasible and cost-prohibitive. In these cases, allowances for increased hunting opportunities and/or the reintroduction of predators (e.g., wolves) may be more feasible than widespread fencing.

Studies show that exclosures and fences are effective in increasing species survival. Wetland species abundance was higher in riparian areas with livestock exclosures than those where cattle were allowed to graze in a comparison of 14 sites on Forest Service and Bureau of Land Management lands in Idaho and Oregon (Coles-Ritchie et al. 2007). Grazing exclosures yielded higher recruitment, survival, and population growth rate of the Columbia spotted frog (*Rana luteiventris*) near stock ponds created to ameliorate drought-induced population declines in the Northern Great Basin (Pilliod & Scherer 2015).

### Use livestock rotation and diversification to reduce pressure on vegetation and soils

Livestock rotation and diversification help to spread risk within drought-affected areas. For example, livestock congregating near water sources trample and overgraze streambanks, causing damage to around 80% of stream and riparian habitats in the Western United States (Belsky et al. 1999). Overgrazing of vegetation can reduce soil water infiltration (Trimble & Mendel 1995), and plant cover and root growth (Finch et al. 2016).

Livestock rotational systems, including flexible stocking methods, facilitate reductions in grazing pressure (Wallander et al. 2013; Renton et al. 2015), which can expedite post-drought vegetation recovery (Finch et al. 2016). For example, under moderate grazing intensity, rangelands may experience limited loss of root growth and may continue to use soil moisture at levels as low as 1–2%, but under heavy grazing, vegetation can experience permanent wilting even with moderate soil moisture loss (levels at 6–8%) (Howery 1999). Livestock rotation during post-drought recovery is also important; pastures may need to be completely rested for several seasons and grazing may need to be delayed until after herbaceous forage plants have produced mature seed (Howery 1999). Economic losses may be offset by opening up rested pastures for hunting or wildlife viewing recreational opportunities (Fox 2008). Emerging tools, such as Grass-Cast, can support flexible stocking decisions on rangelands by integrating climate projections into forage productivity outlooks to better match forage supply with animal demand before, during, and after periods of drought (Peck et al. 2019).

Grassbanking is increasingly used to reduce grazing pressure on rangelands in the western United States. These cooperative conservation agreements incentivize ranchers to rest areas of pasture on their property in exchange for grazing access on other properties (Gripne 2005). The practice was created by the Arizona-New Mexico collaborative Malpai Borderlands Group to support ranchers suffering from drought in the 1990s. In addition to temporary reductions in grazing pressure, grassbanks have been used to preserve space for other conservation benefits, including wildlife habitat and conservation easements prohibitive of development (Gripne 2005; White & Conley 2007).

Shifting the breed, class, or species of livestock on rangelands may ameliorate the effects of drought while maximizing production capacity under drought conditions. For example, *Bos indicus* cattle are more tolerant of heat than *Bos taurus* cattle, while sheep and goats are more heat tolerant and require less water than cattle (Joyce et al. 2013). Drought can also reduce forage nutritional quality (Wallander et al. 2013); bison, sheep, and goats are less selective in foraging than cattle (Fox 2008; Joyce et al. 2013).

### Enhance ecologically available water supply via environmental watering

Managing environmental flows includes the allocation, timing, and quality of flows needed to preserve freshwater ecosystems (Richter et al. 2003; Almazán-Gómez et al. 2018). Environmental flow management is undertaken to reestablish higher minimum flows, stable daily flows, high-flow characteristics, and multiple environmental flow components (Higgins & Konrad 2012). Environmental watering is the active delivery or use of water to achieve ecological objectives, such as conservation of biodiversity and restoration of ecological processes and functions (Beesley et al. 2014; Bond 2016).

Environmental flows include those waters “remaining in rivers after extraction for human use,” however, most of these flows are currently insufficient to sustain ecological benefits (Kingsford 2011). Recovery of these flow regimes requires coordination with the very operations that restrict them in the first place (e.g., dams) (Palmer et al. 2009). For example, dams can be modified to accommodate fish passage and mimic natural flow regimes to support the restoration of downstream floodplain habitats (Richter & Thomas 2007). Potential complications associated with engineered environmental flows include reduced access to critical habitats for fish species, water quality degradation, and loss of flow diversity (Bond et al. 2014).

Environmental watering has been used to maintain post-restoration sites in highly valued and rare habitats (Hanson & Weltzin 2000) and to improve seedling survival post-fire in a semi-arid montane watershed in Spain (via fog water capture and redistribution; Estreal et al. 2009 *in* Grant et al. 2013). Core examples from the literature on environmental watering come from aquatic environments. The delivery of environmental water helped to maintain core refuge habitat for several species stressed by low-flow conditions in the Murray-Darling Basin (Hammer et al. 2013). For example, environmental watering was instrumental in maintaining water depth and dissolved oxygen levels in a stream refuge pool in Rodwell Creek. In 2005, the largest ever environmental water delivery in Australia was made to the Barmah-Millewa Forest, a river red gum (*Eucalyptus camaldulensis*) forest featuring a variety of ephemeral and permanent aquatic habitats (King et al. 2010). The decline of native fish species abundance and distribution in the region has been attributed to drought and flow alterations. To maintain the ecological integrity of the area, managers use environmental flows and environmental water allocations to supplement water supply. Overall, the 2005 environmental watering event yielded enhanced native fish spawning and recruitment, enhanced growth and health of native vegetation, and successful breeding of several waterbird and frog species. However, there were several unintended and undesirable effects, including improved dispersal of invasive fish species and the spread of the exotic weed arrowhead (*Sagittaria graminea*).

Key considerations for environmental watering include where the water is sourced from, as well as when and how it is delivered, although the relative importance differs among species (Beesley et al. 2014). In one study, fish abundance was highest when watering was sourced from natural rivers and lowest when sourced from artificial irrigation channels (Beesley et al. 2014). Fish recruitment and abundance may be optimized if natural water sources are used and delivered during target species’ spawning periods.

### Reduce water withdrawals

Limiting water withdrawals can reduce the effects of drought on wildlife and habitats (Bond et al. 2008; Petes et al. 2012; Vose et al. 2016b). For example, water withdrawals, combined with severe drought conditions, decreased freshwater input to Florida’s Apalachicola Bay, which raised salinity levels and significantly increased disease-related mortality of oysters (Petes et al.

2012). Water withdrawals have significantly contributed to the desiccation and salinization of Lake Abert, Oregon, designated as critical habitat under the Western Hemisphere Shorebird Reserve Network, resulting in limited shorebird use and reduced brine shrimp and alkali fly populations (Moore 2016). Without these withdrawals, the lake would have maintained water volume and optimal salinity limits. In Québec's Yamaska River, Trudel et al. (2016) evaluated vulnerability to low flows associated with climate change as well as water withdrawals. The project established five low-flow alert levels for decision makers (e.g., watchfulness, target low flow, alert, reinforced alert, and crisis) and created water use and withdrawal restriction criteria based on the alert level (e.g., 5–10% reduction per level). In the Gulf Coast region, striped bass (*Morone saxatilis*) populations have sharply declined due to a combination of drought and groundwater withdrawals that have limited their access to cool-water springs (Baker & Jennings 2005). While natural resource managers from state and federal agencies attempt to reintroduce striped bass to local rivers, limiting groundwater withdrawals, particularly during drought periods, has been identified as a key strategy to retain cool-water springs.

### Promote water conservation through collaborative agreements (e.g., water banking, water trading)

Water conservation measures play critical roles in preparing for and recovering from the ecological effects of drought. Water banking is the practice of allocating water for current use or storing it for later use (Green & O'Connor 2001). Water trading is the process of buying, leasing, and selling water rights, which are the rights a particular user has to access and use water from a source (e.g., river, stream, groundwater) (Arnold 2009; Benson 2012). Improving irrigation efficiency may limit water loss at diversions and ditches, maximize water absorption by plants and soil, and restore flows to benefit wildlife (Howell 2003); however, this may lead to increased water consumption, ultimately exacerbating water shortages (Scott et al. 2014; Grafton et al. 2018; Linstead 2018).

Climate change, coupled with increasing population growth and subsequent water demand, requires the reevaluation of how water use and management mechanisms operate. Water banking for environmental purposes has occurred in Idaho, Montana, Oregon, and Washington (Montilla-López et al. 2016). In Idaho, the state government developed a bank to acquire water rights—both temporary and permanent—from parts of the Upper Snake, Payette, Boise, and Lemhi rivers to address ecological impacts from hydroelectric activity in the Columbia Basin. The Montana and Oregon Water Trusts uses water banking to restore water flows of each state's rivers. The Washington Water Trust aims to restore natural water flows to the Yakima and Dungeness rivers by purchasing temporary water rights in drought years as well as permanent rights. For example, the Washington Water Trust bought water rights on First Creek in the upper Yakima River Basin to increase the amount of water available for steelhead in First and Swauk creeks (PRISM 2019).

Water rights have been critical to the effective management of drought in the Bear River Migratory Bird Refuge (BRMBR), located at the delta between the Bear River and the Great Salt Lake in Utah. The Bear River is heavily regulated, flowing through Idaho, Utah, and Wyoming and subject to multiple uses and rights along its course (Downard et al. 2014). The BRMBR have a portfolio of over 25 junior water rights to maintain wetlands and species, but the majority of its water supply comes from the Bear River, and as such, the complex depends on whatever water is left after senior water rights have been accommodated upstream during the height of the irrigation season. Managers have developed management plans and designed infrastructure to maintain wetlands during periods of drought and flooding. The plan establishes management alternatives and objectives for each refuge wetland unit to maintain water depth and distribution to accommodate variations in water flows and minimize consecutive drought years. Managers collaborate with nearby water users, including canal and hydroelectric dam companies, to assess how development and water uses may affect the basin, and participate in water rights allocations meetings. Canals and dikes divert and impound water from the Bear River to supplement refuge supply, and managers use water control structures to supply water to wetlands in winter and spring and aim to maintain these water levels for as long as possible through the fall (Downard & Endter-Wada 2013; Downard et al. 2014).

There are several examples from the literature of how collaborative agreements have been used to engage in water conservation. Ranchers in Montana's Blackfoot and Big Hole Valleys have engaged in drought management collaboratives at the local level that have conserved water during drought years, improved irrigation and well systems, and restored critical riparian habitats (Yung et al. 2015). The Blackfoot Drought Response Plan aims to balance multiple water uses through a "shared sacrifice" framework wherein a consensus-driven, voluntary approach to managing low flows across the basin is prioritized (Smith 2012). Water users can pool their water rights and use water banking to maintain instream flows during periods of drought.

In the Big Hole Valley, the native Arctic grayling (*Thymallus arcticus arcticus*) has declined due to a combination of low flows and high water temperatures driven by decreasing precipitation and growing irrigation demands (Smith 2012). The Big Hole Watershed Committee (BHWC) facilitated conversations between ranchers, farmers, municipalities, anglers, and conservation practitioners to develop a drought management plan in 1997 that balances multiple needs. The plan provides guidance on voluntary measures to prepare for and recover from drought, including limiting fishing and water extraction, as well as triggers for river use restrictions enforced by Montana Fish, Wildlife, and Parks should drought conditions become severe (e.g., flows at or below 20 cfs or daily maximum water temperatures at or above 73°F over some period of time for three consecutive days [BHWC 2017]).

McEvoy et al. (2018) evaluated the degree to which drought management plans from the Big Hole, Blackfoot, Boulder-Jefferson, Gallatin, Madison, Ruby, and Beaverhead-Red Rock watersheds address non-human water needs. The authors note that the majority of plans consider the ecological impacts of drought as triggered by water temperatures and streamflow variation; focusing on streamflows encourages managers to prioritize reactive measures rather than developing long-term, proactive strategies.

## Goal: Increase understanding of ecological drought

### Improve understanding of ecological drought impacts and adaptation options through synthesis, research, monitoring, and evaluation

There is evidence supporting the use of several drought adaptation strategies and actions in use by managers and recommended in adaptation guidance documents. However, there are still knowledge gaps and areas for future research to support more climate-informed drought management. This includes more targeted research on drought interactions with habitats and species as well as increased monitoring and evaluation to detect the effectiveness of specific actions under different conditions. Some specific examples of future research needs identified in the reviewed literature include:

- Evaluation of the ecohydrological variables that influence how drought affects forest species (Littell et al. 2016);
- Identification of genetic provenances that thrive in warmer and drier climates (Spittlehouse 2003);
- Improved long-term monitoring to detect climate change-driven drought-induced vegetation shifts and the effects of drought-induced mortality (and the role of non-drought co-drivers of mortality) on vegetation dynamics (Martínez-Vilalta & Lloret 2016);
- Studies to identify which fuels treatments most effectively reduce intraspecific tree density and basal area and increase diversity in tree size and age to better survive drought and insects (Kane et al. 2014);
- Tests of the application of ecologically-informed indices such as the Normalized Ecosystem Drought Index (NEDI) (Chang et al. 2016) and the Drought Effect of Habitat Loss on Invertebrates (DEHLI) (Chadd et al. 2017). The DEHLI, for example, quantifies the effects of drought on riverine species through stages of channel drying, ecological thresholds and relative tolerance to the potential loss of habitat at each stage, and subsequent recovery. For example, Stage 1 includes species that require fast-flowing, cool, well-oxygenated water while at Stage 5 free surface water is lost leaving behind only moist sediment or complete desiccation;
- Investigation of the effects of drought on forest regeneration (Klos et al. 2009);
- Identification of the genetic variation and adaptation potential of non-commercial and non-threatened vegetation species (Chmura et al. 2011);
- Identification of how factors affecting forest growth may vary depending on the relationships between competitive intensity and climate (e.g., cold, arid) (Gleason et al. 2017);
- Improved understanding of how belowground processes interact with drought (e.g., root dynamics, soil water access) (Vose et al. 2016b); and
- Developing drought indicators for coastal managers (Nolan et al. 2016).

## 4. Ecological Drought Adaptation Decision Support Table

In an initial search, we identified over 260 ecological drought adaptation strategies and actions by reviewing the literature and options generated and prioritized by managers during regional adaptation workshops and other adaptation planning efforts. This list underwent a second screening and editing process to remove duplicates and combine similar actions; this pared down list includes 72 actions. These actions are sourced directly or modified from the Adaptation Partners' Climate Change Adaptation Library for the Western United States[2]; Northern Institute of Applied Climate Science Adaptation Workbook[3]; Stillaguamish Tribe of Indians Natural Resources Climate Change Adaptation Plan[4]; Puyallup Tribe of Indians Climate Change Impact Assessment and Adaptation Options[5]; Climate Adaptation Plan for the Territories of the Yakama Nation[6]; Jamestown S'Klallam Tribe: Climate Vulnerability Assessment and Adaptation Plan[7]; Lummi Nation Climate Change Mitigation and Adaptation Plan[8]; EPA Climate Change Adaptation Resource Center (ARC-X)[9]; Climate Change Adaptation Planning for Resources of Nez Perce-Clearwater National Forests[10]; Southern California Climate Adaptation Project[11]; and the Hawaiian Islands Climate Synthesis Project.[12]

To complement the literature review, we classified the full list of identified ecological drought adaptation actions according to adaptation approach, adaptation strategy, implementation feasibility, and effectiveness in reducing ecological drought vulnerabilities. When selecting adaptation actions for implementation, managers should consider both *effectiveness* (action reduces vulnerability) and *feasibility* (action capable of being implemented). Implementation feasibility considers technical (e.g., financial, staff, data) and socio-political (e.g., social, political, institutional, legal) barriers. An adaptation action with high effectiveness is very likely to reduce associated vulnerabilities and may benefit additional management goals or resources whereas an action with low effectiveness is unlikely to reduce vulnerability and may have negative impacts on other resources. An adaptation action with high feasibility has no obvious barriers and a high likelihood of implementation whereas an action with low feasibility has obvious and/or significant barriers to implementation that may be difficult to overcome. The implementation feasibility and effectiveness of each action are ranked from Low to High (Figure 3; Table 3).

[2] http://adaptationpartners.org/library.php
[3] https://adaptationworkbook.org/niacs-strategies
[4] https://cig.uw.edu/wp-content/uploads/sites/2/2014/11/SNRD-Adaptation-Plan.pdf
[5] http://www.puyallup-tribe.com/tempFiles/PuyallupClimateChangeImpactAssessment_2016_FINAL_pages.pdf
[6] http://www.critfc.org/wp-content/uploads/2016/05/Yakama-Nation-Climate-Adaptation-Plan-.pdf
[7] http://www.jamestowntribe.org/programs/nrs/nrs_climchg.htm
[8] http://lnnr.lummi-nsn.gov/LummiWebsite/userfiles/360_Climate%20Change%20Assessment%20FINAL.pdf
[9] https://www.epa.gov/arc-x
[10] https://www.cakex.org/documents/climate-change-adaptation-strategies-resources-nez-perce-clearwater-national-forests
[11] http://ecoadapt.org/programs/adaptation-consultations/socal-asproducts
[12] http://ecoadapt.org/programs/awareness-to-action/hawaiianislands/products

**Feasibility of Implementing the Action**

- *High*: There are no obvious barriers and it has a high likelihood of being implemented
- *Moderate*: It may be possible to implement the action, although there may be challenges or barriers
- *Low*: There are obvious and/or significant barriers to implementation that may be difficult to overcome

**Action Effectiveness at Reducing Vulnerabilities**

- *High*: Action is very likely to reduce vulnerability and may benefit additional goals or habitats
- *Moderate*: Action has moderate potential to reduce vulnerability, with some limits to effectiveness
- *Low*: Action is unlikely to reduce vulnerability

**Figure 3. Description of action feasibility and effectiveness rankings.**

Effectiveness rankings are based on evidence in the scientific literature. Feasibility rankings are based on expert opinion (e.g., natural resource managers provided rankings during workshops that were a part of other adaptation planning projects, such as Climate Change Adaptation Planning for Resources of Nez Perce-Clearwater National Forests, Northern Rockies Adaptation Partnership, Southern California Climate Adaptation Project, Hawaiian Islands Climate Synthesis Project, and scientist-manager expert panels on prescribed fire and sea level rise in the Northwest). The table also provides notes on any known caveats or limitations to feasibility and effectiveness.

This table can help managers prioritize actions for implementation (e.g., actions with high feasibility and high effectiveness), better target management efforts toward specific challenges (e.g., actions with low or moderate feasibility but high effectiveness), and/or evaluate whether to proceed with implementation (e.g., actions with high feasibility but low effectiveness). For the latter two purposes, managers may consider the following questions:

- **Low or Moderate Feasibility/High Effectiveness Actions**: What steps can be taken to increase the likelihood of this action being implemented in the future?
  - *Example*: Would improving public outreach and education or enhancing public/private collaboration facilitate increased management access and activity on private lands (e.g., to remove invasive species)?
- **High Feasibility/Low or Moderate Effectiveness Actions**: Does this action still make sense given projected climate changes and impacts?
  - *Example*: If conditions are projected to become drier, should groundwater pumping still continue to support lowland wetland hydrology?

Alternatively, there may be some actions that do not directly reduce vulnerability but could provide important information, tools, or support to address vulnerability down the line. For example, actions aimed at increasing knowledge through monitoring or modeling could provide key information for future restoration activities (e.g., creating detailed species genetic profiles to select genetically and ecologically suitable plant species for future conditions). Managers may want to weigh the costs and benefits of implementing actions with the timeframe required to reduce vulnerability directly. Additionally, actions focused on coordination and collaboration may not directly address vulnerabilities, but these remain important steps toward better planning and management.

Table 3. Ecological drought adaptation actions classified according to adaptation approach, adaptation strategy, implementation feasibility, and effectiveness in reducing ecological drought vulnerabilities.

Implementation feasibility considers technical (e.g., financial, staff, data) and socio-political (e.g., social, political, institutional, legal) barriers. Both implementation feasibility and effectiveness are ranked from Low to High. Feasibility rankings were provided by natural resource managers (e.g., during workshops that were a part of other adaptation planning projects). Effectiveness rankings are based on information in the scientific literature. Any caveats or limitations to feasibility and effectiveness are listed in the Notes column. Effectiveness rankings for *italicized* adaptation actions are based on expert opinion, as no supporting literature could be found. Action relevance to habitats is indicated in parentheses: Forest (FO), Grassland/Shrubland (G/S), Freshwater (FW), Marine/Coastal (M/C).

| Adaptation Strategy | Associated Actions | Feasibility | Effectiveness | Notes |
|---|---|---|---|---|
| | **RESISTANCE APPROACHES** | | | |
| Enhance ecologically available water supply via environmental watering | Consider alternative water supplies (e.g., importing water from other areas) for federal lands to retain instream flows, particularly during the dry season (FW, M/C) | **Low-Mod** | **High** | |
| | Install berms or dikes to divert surface water to water-stressed lowland areas (FW) | **Mod** | **Mod-High** | Feasibility: social barriers likely greater than technical barriers |
| | Maintain water in wetlands using float valves, diversion valves, and hose pumps (FW, M/C) | **Mod** | **Mod** | |
| | Manage water levels to maintain hydrologic function and supply proper soil moisture to vegetation adjacent to the stream during critical periods (e.g., by manipulating existing dams and water control structures or through restoration of natural dynamic water fluctuations) (FW) | **Mod-High** | **High** | |
| Reduce tree density and fuel loads through thinning and prescribed burns | Design burn prescriptions that consider soil moisture requirements (FO, G/S) | **Mod** | **High** | Feasibility: political and social barriers (e.g., wariness of fire; air quality concerns); funding skewed toward suppression |
| | Promote age class, species, structural, and/or spatial (e.g., forest gaps) diversity across the landscape using a variety of management tools (e.g., prescribed and wildland fire, regeneration harvest, thinning) (FO) | **Mod-High** | **High** | Feasibility: low at the landscape scale, high at the stand scale |

| Adaptation Strategy | Associated Actions | Feasibility | Effectiveness | Notes |
|---|---|---|---|---|
| | Reduce density via variable means (e.g., thinning, prescribed fire, wildfire use, girdling, falling and leaving trees), with density and structural goals based on projected future conditions (FO) | **Mod-High** | **Mod-High** | Feasibility: low at the landscape scale, high at the stand scale |
| | Use herbicide or mechanical thinning to prevent the encroachment of woody competitors and invasive species and conserve soil moisture (FO, G/S) | **Mod-High** | **Mod** | Feasibility: technical barriers (e.g., labor- and time-intensive) Effectiveness: limited by treatment frequency and scale |
| Reduce water withdrawals | Reduce water withdrawals to retain instream flows, particularly during the dry season and/or critical low flow periods (FW) | **Low** | **High** | Feasibility: political and social barriers |
| Use exclosures and fences to protect water resources | Use exclosures and fences to protect groundwater-dependent habitats and associated species (FW) | **Mod** | **High** | Feasibility: social and financial barriers |
| **RESILIENCE APPROACHES** | | | | |
| Consider species type, timing, and location in management activities | Consider using genetically improved (e.g., insect- or disease-resistant) seedling stock to increase resilience to disturbance such as drought (FO, G/S) | **Mod-High** | **High** | Feasibility: social barriers |
| | Plant larger vegetation (e.g., saplings) to help ensure establishment and survival, particularly in disturbed sites where dry conditions are expected (FO) | **High** | **Mod** | Effectiveness: saplings of some species are acutely sensitive to drought |
| | Favor or establish more drought- and heat-tolerant species on narrow ridge tops, south-facing slopes with shallow soils, or other sites that are expected to become warmer and drier (FO, G/S) | **High** | **Mod-High** | Feasibility: limited by funding and willingness to move toward different species (e.g., perception of economic loss; traditional conifer management goals) |
| | Protect trees that exhibit adaptation to water stress (e.g., trees with low leaf area:sapwood ratio) (FO) | **High** | **Mod-High** | |

| Adaptation Strategy | Associated Actions | Feasibility | Effectiveness | Notes |
|---|---|---|---|---|
| | *Prioritize removal of stressed, damaged, or unhealthy trees in order to promote the survival of those expected to fare better (FO)* | **Mod-High** | **Mod-High** | |
| | *Reduce abundance of mesic species on drought-prone sites (e.g., western hemlock, western red cedar) (FO)* | **Mod** | **Mod** | |
| | *Retain some survivors of a die-back event (e.g., drought-induced mortality, pathogenic blight) when conducting salvage harvests in the affected area (FO)* | **Mod** | **Mod-High** | |
| Create or enhance water supply | Construct artificial wetlands to support threatened fish, wildlife, and birds (FW) | **Mod** | **High** | |
| | Enhance water-retaining areas, such as abutting wetland projects with agricultural areas (FW) | **Mod** | **High** | |
| | Manage gaps and forest openings to increase snow catch accumulation, and use techniques to shade snow such as mulching with wood chips to extend the retention of snowpack and enhance water availability during the growing season (FO) | **High** | **High** | |
| | Use wildlife water developments to ameliorate loss of naturally-occurring water sources for the benefit of wildlife, game species, and livestock (FW) | **High** | **Mod-High** | Feasibility: social/political barriers, requires long-term maintenance |
| | Use snow fences and reflective tarps to retain snowpack and enhance water availability during the growing season (FO, G/S, FW) | **High** | **High** | Effectiveness: more effective at smaller scales |
| Use livestock rotation and diversification to reduce pressure on vegetation and soils | Manage livestock grazing to restore ecological function of riparian vegetation and maintain streambank conditions (FW) | **High** | **High** | Feasibility: political and social barriers; some technical barriers (e.g., providing offsite water) |
| | Shift the breed, class, or species of livestock on rangelands to ameliorate the effects of drought (FW, FO, G/S) | **Mod** | **Mod** | |
| | Use grassbanking cooperative agreements to ease grazing pressures (G/S, FO) | **Mod** | **Mod-High** | Effectiveness: more effective at smaller scales |
| | Rotate pastures to ease grazing pressure during drought (G/S, FO) | **Mod** | **Mod-High** | |
| Enhance natural water storage | Add wood to streams to enhance natural water storage (FW) | **Mod-High** | **Mod-High** | |
| | Enhance natural water storage through the use of beaver dam analogs (FW) | **Mod-High** | **High** | Feasibility: likely less on private lands compared |

| Adaptation Strategy | Associated Actions | Feasibility | Effectiveness | Notes |
|---|---|---|---|---|
| | | | | with public lands, requires maintenance |
| | Slow water flows and increase soil infiltration with rock structures (e.g., one-rock dams, media lunas) (FW, G/S) | **High** | **Mod-High** | |
| | Reintroduce, enhance, and maintain beaver populations to improve water storage (FW) | **Mod-High** | **High** | Feasibility: likely less on private lands compared with public lands; may be some regulatory obstacles; to date, not well tested in Washington |
| | Utilize green infrastructure (e.g., bioswales, permeable pavement) (FW) | **Mod-High** | **High** | Feasibility: can require long-term maintenance |
| Maintain and enhance infiltration, water storage capacity, and/or health of soils | Increase soil organic matter and/or use soil amendments to restore or improve soil quality, water-holding capacity, soil structure, and water infiltration, and to reduce erosion (e.g., biochar) (FO, G/S) | **Mod** | **High** | Feasibility: more likely if tied to existing management actions (e.g., riparian management, salvage logging) |
| | Maintain soil productivity through appropriate silvicultural practices (e.g., fuels treatments) (FO) | **High** | **High** | |
| | Plant deep-rooted perennials to reduce runoff and improve infiltration (FO) | **High** | **Mod-High** | |
| | Restrict access and/or alter timing of use (e.g., recreation, forest operations, grazing) to minimize soil compaction and reduce potential impacts on soil water retention (FO, G/S, FW, M/C) | **Mod** | **Low-Mod** | |
| | Retain coarse woody debris in the uplands and riparian areas to maintain moisture, soil quality, and nutrient cycling (FO, FW) | **Mod-High** | **High** | Feasibility: more likely if tied to existing management actions (e.g., riparian management, salvage logging) |
| Restore and reconnect floodplains to | Increase aquatic habitat structure, complexity, and connectivity to refugia in stream channels, off-channels, channels fed by wetlands, and floodplains (FW, M/C) | **High** | **High** | |

| Adaptation Strategy | Associated Actions | Feasibility | Effectiveness | Notes |
|---|---|---|---|---|
| allow for groundwater recharge | Reconnect streams to floodplains/alluvial fans to improve hyporheic and base flow conditions (FW, M/C) | **Mod-High** | **High** | Feasibility: technical barriers (e.g., costly to implement) |
| | Remove and/or modify roads to control erosion and runoff and restore floodplain and hydrological connectivity (FW, M/C) | **Mod-High** | **Mod-High** | Feasibility: some social/ political and financial barriers |
| | Remove or modify dams (particularly if defunct or those with little hydroelectric or irrigation value), dikes, and levees (FW) | **Mod** | **High** | Feasibility: some social/political and financial barriers (e.g., land-use conflicts, potential short-term increases in flood and erosion risk) |
| Restore habitats by maintaining native vegetation cover and removing invasive species | Interplant to supplement natural regeneration and genetic diversity (FO, G/S) | **Mod-High** | **Mod** | |
| | Monitor, remove, control, and prevent the spread of non-native species as well as introduction/dispersal vectors (FO, FW, G/S, M/C) | **Mod-High** | **Mod-High** | Feasibility: technical barriers (e.g., labor- and time-intensive); generally difficult to eradicate non-native species at larger spatial scales<br>Effectiveness: limited by treatment frequency and scale |
| | Plant potential microsites with a mix of native species (FO, FW, G/S, M/C) | **High** | **Mod** | |
| **RESPONSE APPROACHES** | | | | |
| Actively plant and store seed from drought-tolerant | Collect seed from trees that exhibit adaptation to water stress for future regeneration (FO) | **High** | **High** | |
| | Create novel drought-adapted communities "from scratch" in areas that have been severely affected by natural or human disturbance (FO, G/S) | **Mod** | **Mod** | |
| | Plant native species that are well-adapted to drought and/or have a broader moisture tolerance range (FO, FW, G/S, M/C) | **High** | **Mod-High** | Feasibility: limited by mindset/willingness to |

| Adaptation Strategy | Associated Actions | Feasibility | Effectiveness | Notes |
|---|---|---|---|---|
| species and individuals | | | | move toward different species (e.g., perception of economic loss; traditional conifer management goals) |
| | Plant stock from seeds collected from healthy trees in warmer or drier locations in the region (FO) | **High** | **Mod-High** | |
| | Seed or plant drought-resistant genotypes of commercial species where increased drought stress is expected (FO) | **High** | **Mod-High** | |
| Identify and protect drought refugia | Identify and protect a network of sheltered mountain slopes, valleys, or forests with continuous shading canopy (FO) | **Mod-High** | **Mod-High** | |
| | Identify, protect, and restore headwaters and source waters (e.g., groundwater recharge areas, seeps and springs, mid- and high-elevation wetlands) (FW) | **Mod** | **High** | |
| | Minimize or eliminate disturbances in areas that may be buffered from climate change (FO, FW, G/S, M/C) | **Mod** | **Mod** | |
| Protect vulnerable species through assisted migration and improved habitat connectivity | Actively assist aquatic species movement/migration in times of drought or extreme low flows (FW, M/C) | **Mod** | **High** | Feasibility: legal and social barriers, dependent on targeted species |
| | Manage for culturally important species in areas where temperature and hydrologic conditions may be most suitable in the future (FO, FW, G/S, M/C) | **Mod** | **Mod-High** | |
| | Plant vulnerable species in suitable habitat outside their current range (FO, FW, G/S, M/C) | **Mod** | **Mod-High** | Feasibility: legal and social barriers |
| | Maintain and/or create a network of waterholes to support species under water stress (FO, FW, G/S) | **Low** | **High** | |
| **COLLABORATION APPROACHES** | | | | |
| Promote water conservation through collaborative agreements | Acquire water rights/instream flow rights (FW) | **Low-Mod** | **High** | |
| | Coordinate with downstream partners on water conservation education (FW) | **Mod-High** | **Mod** | |
| | Explore opportunities for increasing irrigation and water use (e.g., stock ponds) efficiency, and returning diverted water back into stream channels during critical low flow periods (FW) | **Mod-High** | **Mod-High** | |

| Adaptation Strategy | Associated Actions | Feasibility | Effectiveness | Notes |
|---|---|---|---|---|
| (e.g., water banking, water trading) | Explore opportunities for water trading with others (FW) | **Mod** | **Mod** | |
| | Implement facility and operational improvements to increase efficiencies in water diversion, conveyance, and delivery (FW) | **Mod** | **High** | |
| | Promote conjunctive use of water (surface and ground) and water banking (FW) | **Mod** | **High** | |
| **KNOWLEDGE APPROACHES** | | | | |
| Improve understanding of ecological drought impacts and adaptation options through synthesis, research, monitoring, and evaluation | Consider information from surveys of warmer basins as indicators of potential future vulnerability (FW) | **Mod** | **Low-Mod** | |
| | Use models to improve seasonal water supply and demand forecasts in order to inform irrigation allocation, drought declaration, and planning and regulation of instream flows (FW) | **Mod** | **Mod-High** | |
| | Continue to inventory, identify, and prioritize stream reaches, floodplains, riparian areas, and wetlands for protection and restoration, taking into account areas that may be particularly resilient or vulnerable to drought conditions (FW, M/C) | **High** | **Mod** | |
| | Develop watershed models to describe forestry and climate change (e.g., snowpack, precipitation, temperature) interactions in order to identify ways to maximize water retention (FO, FW) | **Mod-High** | **High** | |
| | Evaluate the long-term adequacy of water delivery infrastructure to ensure that changes in hydrological patterns can be anticipated and managed effectively (FW) | **High** | **Mod-High** | |
| | Examine how restoration project maintenance may need to be restructured in drought years (FO, FW, G/S, M/C) | **High** | **Mod-High** | |
| | Identify, map, and monitor groundwater and surface water sources and conditions. Incorporate monitoring data into models, along with information on projected climate changes, to forecast future water supply and quality changes and trends (FW) | **Low-Mod** | **Mod-High** | Feasibility: technical barriers (e.g., staff, funding, locating groundwater sources) |
| | Improve the network of real-time water and weather stations, which are fundamental for drought forecasting, water supply forecasting and monitoring, improving hydrologic models, and long-term water resources planning (FW) | **Mod** | **Mod-High** | Feasibility: technical barriers (e.g., staff, funding) |

| Adaptation Strategy | Associated Actions | Feasibility | Effectiveness | Notes |
|---|---|---|---|---|
| | | | | Effectiveness: dependent on number and frequency of monitoring stations |
| | Monitor food web dynamics and species distributions for changes with warming and drying (FO, FW, G/S, M/C) | **Mod** | **Mod** | |
| | Utilize existing models to better understand potential climate change impacts (e.g., forecasted stream temperatures and changing hydrologic regimes) on fishery resources (FW, M/C) | **High** | **Mod** | |

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