

CLIMATE CHANGE IMPACT
ASSESSMENT FOR THE
**UPPER MISSOURI
RIVER BREAKS**
NATIONAL MONUMENT

Photo: Bureau of Land Management

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NORTH CENTRAL
Climate Adaptation
Science Center



University of Colorado
Boulder

Overview

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I. STATEMENT OF PURPOSE

The purpose of this assessment is to inform future climate change-related decisions and research in the **Upper Missouri River Breaks National Monument (UMRBNM)** by laying out the potential impacts of climate change on the monument as a whole, and on select prioritized objects. This assessment, which includes a synthesis of the best available scientific literature and consultations with subject experts, can be used to inform the creation of a science plan for the monument. The specific goals of this assessment are to (right):

Goal #1:

Synthesize historic and projected climate trends in UMRBNM

Goal #2:

Detail the effects of climate change on important habitats in the monument

Goal #3:

Describe the vulnerabilities of select objects of interest for the monument under climate change

Goal #4:

Identify where future research is needed to understand how climate change may affect the monument or its prioritized objects

II. CLIMATE CHANGE IN UMRBNM

Since 1979 the average annual temperature on the monument has increased by 0.2°F per decade, and the average annual precipitation has increased by 0.4 inches per decade (Hegewisch & Abatzoglou, 2024). Overall, the region has experienced a **2°F warming and 10% increase in**



Photo: Bureau of Land Management

precipitation in the last four decades. Temperature is expected to increase during the 21st century as the concentration of atmospheric greenhouse gasses increases. The yearly precipitation is expected to increase as well, although there is large variation across different climate projections. The magnitude of projected changes in temperature and precipitation depends on future concentrations of greenhouse gasses, climate variability, and differences among climate models.

To account for this uncertainty, all climate change projections provided in this assessment, unless stated otherwise, are the aggregated mean of 20 different climate models used by the Intergovernmental Panel on Climate Change. Downscaled climate data and relevant quantitative information is obtained from <https://ClimateToolbox.org>. Each of these models have a unique

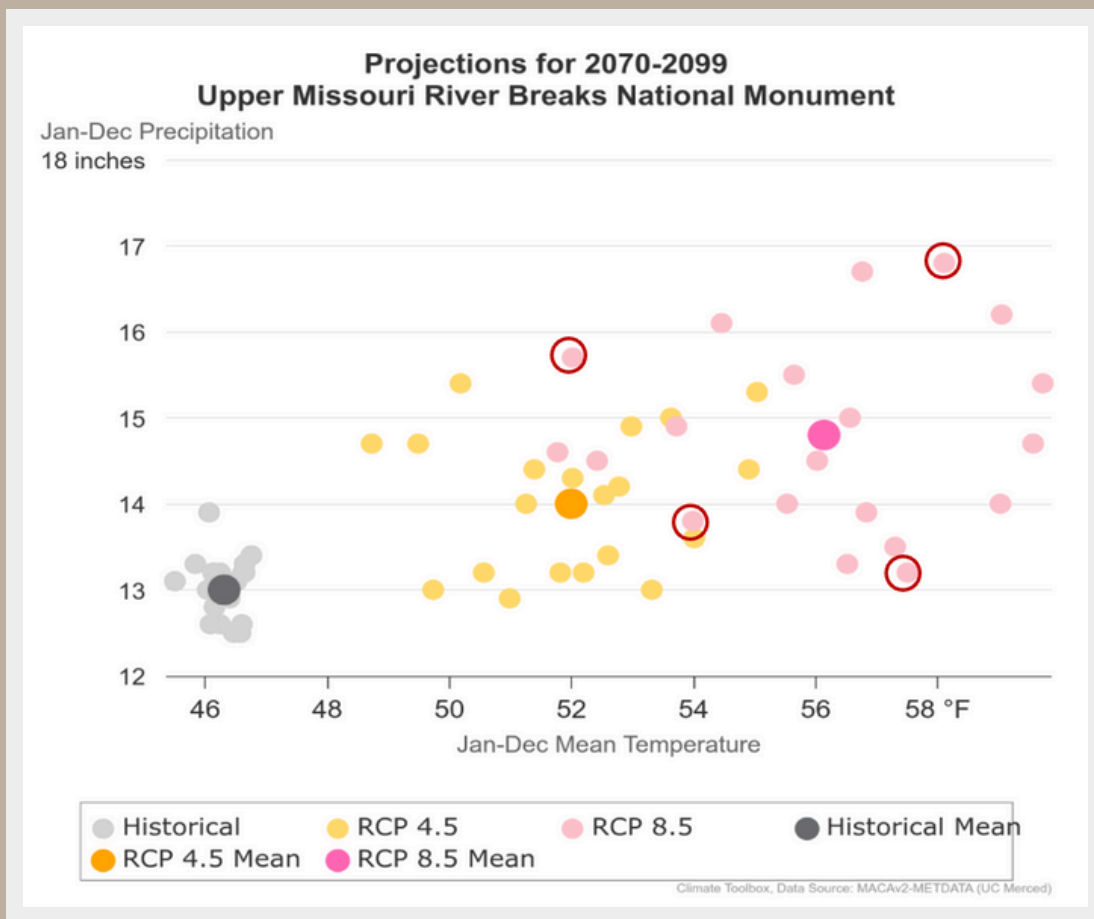


Figure 1. Projections of total annual precipitation and mean annual temperature for 2070-2099 by 20 climate models and 2 emission scenarios, relative to historical (1971-2000) values. Figure plotted from Climate Toolbox (Hegewisch & Abatzoglou, 2024). Table 2 shows projections in various different climate metrics for the four climate scenarios identified by red circles.

projection for different scenarios of future atmospheric greenhouse gas emissions. This assessment considers two such emission scenarios, called the **representative concentration pathways (RCPs)**. In the RCP 4.5 scenario, emissions peak around 2040 then decline. It would likely result in global warming of approximately 3-6°F (1.7-3.3°C; IPCC, 2019). In the RCP 8.5 scenario, emissions continue to rise throughout the 21st century. It would likely result in global warming between 5.5-10°F (3.2-5.4°C; IPCC, 2019).

See Figure 1 (above) for an illustration of the variability in climate projections between different models and RCPs.

An alternative way of accounting for uncertainty that is used by many

practitioners is to examine a select set of divergent model projections (typically 3-5) that, taken together, could capture a large range of possible climate change outcomes (Table 2).

All models project mean annual and seasonal temperatures in UMRBNM to increase substantially by the end of the century (Figure 1; Table 1).

Climate Metric	Historical Value	RCP 4.5	RCP 8.5
Annual Mean Temperature (°F) (change relative to historical value)	46.29	52 (5.71)	56.12 (9.83)
Winter Mean Temperature (°F) (change relative to historical value)	23.2	29.06 (5.86)	33.42 (10.22)
Spring Mean Temperature (°F) (change relative to historical value)	46.16	51.48 (5.32)	54.41 (8.25)
Summer Mean Temperature (°F) (change relative to historical value)	69.3	75.42 (6.12)	80.34 (11.04)
Fall Mean Temperature (°F) (change relative to historical value)	46.51	52.04 (5.53)	56.31 (9.80)

Table 1. Projections of mean annual and seasonal temperatures based on the mean of 20 climate models for 2070-2099 relative to 1971-2000 under RCP 4.5 and RCP 8.5. Data obtained from Climate Toolbox (Hegewisch & Abatzoglou, 2024).

Climate Metric	CanESM2 (Hot, Wet)	IPSL-CM5A- LR (Hot, Dry)	GFDL-ESM 2M (Warm, Wet)	bcc-csm1-1-m (Warm, Dry)	Model Mean	Historical Value
Winter Mean Temp (°F) (change relative to historical by °F)	34.95 (11.75)	34.62 (11.42)	29.43 (6.23)	30.62 (7.42)	33.42 (10.22)	23.2
Winter Precipitation (in) (% change relative to historical)	2.17 (65.65)	1.5 (14.50)	1.47 (12.21)	1.82 (38.93)	1.7 (29.77)	1.31
Spring Mean Temp (°F) (change relative to historical by °F)	56.94 (10.78)	55.01 (8.85)	50.92 (4.76)	52.72 (6.56)	54.41 (8.25)	46.16
Spring Precipitation (in) (% change relative to historical)	5.35 (28.30)	4.97 (19.18)	5.66 (35.73)	5.09 (22.06)	5.57 (33.57)	4.17
Summer Mean Temp (°F) (change relative to historical by °F)	83.24 (13.94)	82.23 (12.93)	74.98 (5.68)	79.65 (10.35)	80.34 (11.04)	69.3
Summer Precipitation (in) (% change relative to historical)	5.82 (12.14)	4.4 (-15.22)	6.07 (16.96)	3.7 (-28.71)	4.87 (-6.17)	5.19
Fall Mean Temp (°F) (change relative to historical by °F)	57.27 (10.76)	58.18 (11.67)	52.68 (6.17)	52.92 (6.41)	56.31 (9.80)	46.51
Fall Precipitation (in) (% change relative to historical)	3.44 (43.93)	2.37 (-0.84)	2.5 (4.60)	3.22 (34.73)	2.7 (12.97)	2.39
Annual Mean Temp (°F) (change relative to historical by °F)	58.1 (11.81)	57.51 (11.22)	52 (5.71)	53.98 (7.69)	56.12 (9.83)	46.29
Annual Precipitation (in) (% change relative to historical)	16.8 (28.54)	13.25 (1.38)	15.71 (20.20)	13.85 (5.97)	14.86 (13.70)	13.07
Coldest Winter Day (°F) (relative to historical by °F)	-1.61 (20.11)	-4.3 (17.41)	-11.59 (10.13)	-10.82 (10.90)	-5.36 (16.36)	-21.72
Hottest Summer Day (°F) (relative to historical by °F)	114.84 (14.07)	113.61 (12.84)	107.93 (7.16)	114.03 (13.26)	113.32 (12.55)	100.77
Day of First Fall Freeze (days) (relative to historical by days)	Oct. 7 (10)	Oct. 8 (11)	Sept. 24 (-3)	Sept. 30 (3)	Oct. 7 (10)	Sept. 27
Day of Last Spring Freeze (days) (relative to historical by days)	Apr. 12 (-22)	Apr. 14 (-20)	Apr. 20 (-14)	Apr. 23 (-11)	Apr. 13 (-21)	May 4
Length of Growing Season (days) (relative to historical by days)	178 (31.90)	177 (30.90)	157 (10.90)	160 (13.90)	176.99 (30.89)	146.1
Days with Heat Index ≥100 °F (days) (relative to historical by days)	44.6 (44.10)	33.1 (32.60)	13.5 (13.00)	31.4 (30.90)	30.9 (30.40)	0.5

Table 2. Projections in various climate metrics for four different climate scenarios (identified in Figure 1) and multi-model mean (20 climate models) for 2070-2099 under RCP 8.5. Values in parentheses are change from the historical (1971-2000) values. Table obtained from Climate Toolbox (Hegewisch & Abatzoglou, 2024).

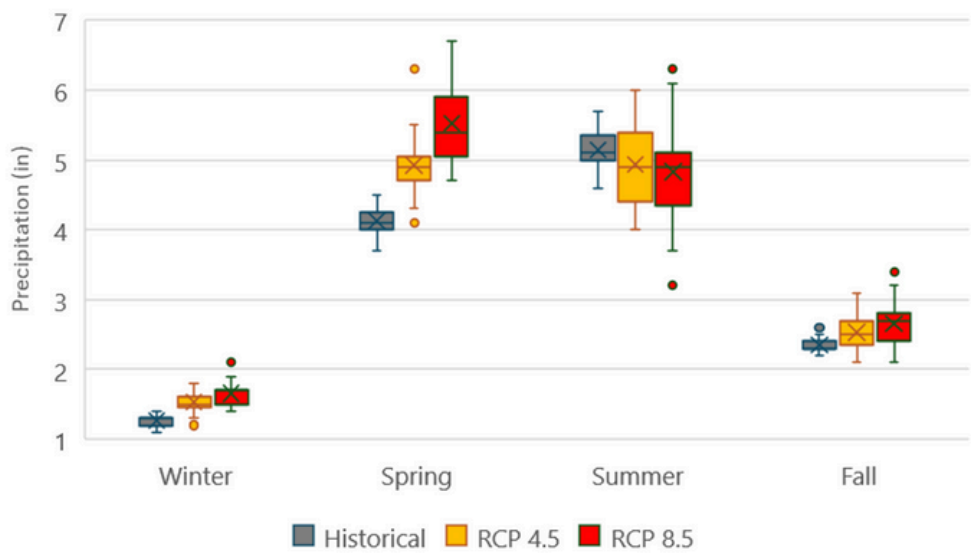


Figure 2. Projections for total seasonal precipitation for 2070-2099 from 20 climate models, relative to historical (1971-2000) values. Data obtained from Climate Toolbox (Hegewisch & Abatzoglou, 2024)



Mean annual precipitation is expected to increase by the end of the century across most of the models, but the magnitude of that increase varies substantially across models, and most models project a decrease in summer precipitation (Figure 2).

The amount of interannual variability in precipitation is also expected to increase, though it is unclear how much of the variability is due to wet years getting wetter, dry years getting drier, or both (Whitlock et al., 2017). A greater amount of yearly precipitation is also expected in the form of intense precipitation events (Knapp et al., 2023).



The increase in temperatures predicted by all the climate models would lengthen the growing season in the monument and increase potential evapotranspiration, which is the maximum amount of water that would evaporate if it was available (Hegewisch & Abatzoglou, 2024).

These two factors would cause soils to dry out more rapidly in the summer, and to stay dry for longer, even if summer precipitation remains constant (Palmquist et al., 2016b). Ultimately, this would lead to **more frequent, longer periods of drought** as climate change progresses (Whitlock et al., 2017; Knapp et al., 2023).

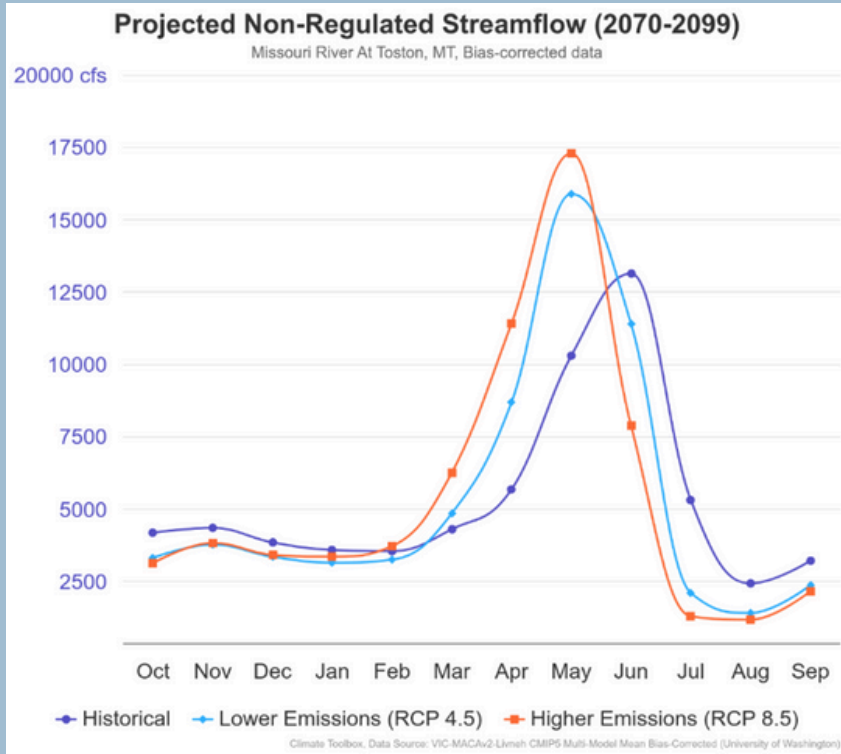


Figure 3. Projected natural streamflow for the Missouri River at Toston, upstream of UMRBNM. Figure obtained from Climate Toolbox (Hegewisch & Abatzoglou, 2024).

While soil moisture in UMRBNM is dependent on local precipitation, the amount of water that flows in the rivers and major streams of the monument is more affected by precipitation at the source of the Missouri River and its tributaries.

These headwaters are located in the Rocky Mountains, or other, smaller island mountain ranges closer to UMRBNM, and are fed primarily by winter snowpack (MT-DNRC, 2015).

Projected increases in temperature would cause:

- More winter precipitation to fall as rain;
- Snowmelt to start earlier; and
- A greater likelihood of rain-on-snow events that can quickly reduce the snowpack and cause flooding (Whitlock et al., 2017).

This would cause **peak streamflows to occur earlier in the year, and summer and fall streamflows would decrease** (Figure. 3; Whitlock et al., 2017). Lower summer streamflow, combined with higher temperatures, would drive up water temperature in the monument (Sinokrot & Gulliver, 2010). Modeling from Isaak et al. (2016) found that mean August water temperatures in the section of Missouri River on the monument could rise to over 75°F by the end of the century, more than 4 degrees above the historical average.



Photo: Bureau of Land Management

An increase in extreme precipitation events may mean a **greater likelihood of microbursts**, which have been known to trigger flash floods in smaller drainages of UMRBNM. These floods can cause slope failures on the monument, in which an entire hillside loses structural integrity and rolls downhill in a landslide (Zhang et al., 2011).

More flooding on the monument, as well as a **greater possibility of wildfire** that would destroy stabilizing plant roots, may lead to a greater frequency of slope failures.

III. IMPACTS ON HABITAT



Photo: Bureau of Land Management

PONDEROSA PINE WOODLANDS

42% of UMRBNM is covered by forest, most of which is dominated by ponderosa pine (*Pinus ponderosa*). Ponderosa pine is a relatively drought-resilient conifer species commonly found in low-elevation dry forests (Fryer, 2018), and has the potential to persist on the monument under the hotter, drier average summer conditions that climate change will bring.

Ponderosa responds to these conditions by reducing growth (Peltier & Ogle, 2019) and shifting biomass allocation from leaves to sapwood (McCullough et al., 2017), potentially

altering canopy structure. Additionally, years with extremely dry conditions make ponderosa pine **more susceptible to insect disturbances** (Negrón et al., 2009), **wildfire** (Sherriff and Veblen, 2008), and **drought-induced mortality** (Ganey & Vojta, 2011).

Juvenile recruitment is an important process for maintaining ponderosa pine forest on the monument. Ponderosa pine seedlings require greater amounts of moisture to establish than their adult counterparts need to survive (Petrie et al., 2017). Davis et al. (2020) found that the average summer vapor pressure deficit (the difference between the amount of moisture in

in the air and the amount the air can hold when saturated), along with the maximum temperature of the warmest month of the year, was strongly correlated with ponderosa pine recruitment after fires. However, water availability is most critical for seedlings during establishment in the spring and in the following fall through spring (League & Veblen, 2006), when precipitation is expected to increase (Fig. 2). Given the episodic nature of ponderosa recruitment (Brown & Wu, 2005), and a projected increase in interannual rainfall variability, future ponderosa recruitment might be restricted to abnormally wet years.

At the same time, seedling recruitment would likely become more critical for forest persistence in the future as more fire-conducive climate conditions make stand-replacing wildfires more likely (Parks & Abatzoglou, 2020), leaving forests vulnerable to vegetation shifts if

ponderosa cannot regenerate after fires (Davis et al., 2020).

Research conducted on ponderosa pine stands suggests that their vulnerability to both drought and stand-replacing fire can be mitigated through thinning. Studies on ponderosa stands in drought conditions found those that were mechanically thinned beforehand continued to grow, were less water stressed, and had more access to soil moisture than un-thinned stands (Sankey & Tatum, 2022; Young et al., 2023).

Thinning also reduces the amount of fuel available for wildfires, lessening potential fire intensity (Pollet & Omi, 2002; Parks et al., 2023). However, fuel reduction is most effective using prescribed burns, which can remove more surface-level fuel than mechanical thinning (Davis et al., 2024).





Overall, **increased summer water stress may result in ponderosa pine forests in UMRBNM having less biomass and more limited recruitment** under future climate change. Related stressors, most notably wildfire, may also become more persistent as a result of drier conditions. However, elevated CO₂ concentrations in the atmosphere, which have been shown to increase water-use efficiency in plants (McDowell et al., 2022), may mitigate these impacts, though to what extent is unclear.

SAGEBRUSH STEPPE

40% of UMRBNM is covered by sagebrush steppe, most of which is taken up by Wyoming big sagebrush (*Artemisia tridentata wyomingensis*), a subspecies of big sagebrush. Studies predict that projected climate change will have both positive and negative effects on big sagebrush in the area of the monument. An earlier growing season would allow individuals more time to grow (Perfors et al., 2003), increased winter and spring precipitation would allow water storage in deeper soil layers (Palmquist et al., 2016a), and a warmer, wetter winter and spring, combined with reduced snowpack, would provide better conditions for seed germination (Palmquist et al., 2016a).

However, longer, drier summers would put more stress on adult sagebrush, and hinder the ability of seedlings to establish (Palmquist et al., 2016b). More favorable fire conditions also pose a risk to sagebrush, as big sagebrush's lack of capacity for dispersal and germination post-fire (Young & Evans, 1989) means it takes over 50 years for sagebrush habitat on the monument to be restored (Baker, 2006).

Though cooler conditions and greater annual precipitation makes sagebrush habitat in the northern Great Plains less vulnerable to climate change than other parts of the species' range (Chambers et al., 2023), it is still unclear whether climate change will be a net positive or negative for big sagebrush in UMRBNM. Palmquist et al. (2021), using a plant simulation model based on competition for water availability, found that changes in soil water availability in the northern Great Plains would, by the end of the century, lead to no change in sagebrush biomass under the RCP 8.5 scenario, and a decrease in biomass under the RCP 4.5 scenario.

However, Renwick et al. (2018), using four different models, two based on spatial and temporal relationships with bioclimatic variables and two based on mechanistic approaches to simulate sagebrush recruitment and growth, found that sagebrush performance in the region would benefit from increased temperatures. Even if sagebrush biomass decreases as Palmquist et al. (2021) predicts, sagebrush density may still increase, as mortality of large individuals reduces competition and

resource availability for remaining smaller individuals, leading to shifts in stand structure.

Though there is disagreement in literature on the direct effects of climate change on big sagebrush species, **climate change is very likely to benefit the invasive species cheatgrass** (*Bromus tectorum*), and other invasive grasses. Cheatgrass' winter dormancy period is shorter than native plants, allowing it to take advantage of greater precipitation in the late fall and early spring (Taylor et al., 2014), and giving it a competitive advantage over native species in the hot, dry summer months (Bradley et al., 2009; Brummer et al., 2016). Cheatgrass also provides fine fuels for wildfire, which may create a positive feedback loop in which post-fire habitats are more susceptible to future cheatgrass invasion (Shinneman, 2020). Light to moderate livestock grazing might hinder this relationship by reducing potential fuel loads (Davies et al., 2009).

RIPARIAN HABITAT

12% of UMRBNM is riparian habitat. **This habitat is sensitive to irregularity in streamflow, which makes it especially vulnerable to climate change** (Fig. 3). Lower summer streamflow would lower the water table and reduce water availability to riparian vegetation, leading to decreases in growth, recruitment, and survival of riparian plant species, ultimately making the habitat more competitive for drought-tolerant species (Rood et al., 2008). Green ash (*Fraxinus pennsylvanica*) is especially vulnerable to decreased water availability, as UMRBNM is already at the arid edge of its range (Lesica, 2009). Many riparian plants also depend on occasional floods to reduce the abundance of shade tolerant, mid and late successional tree species such as elm and maple that would likely take over their habitat in the absence of disturbances (Johnson, 1998).

Cottonwood trees (*Populus* sect. *Aigeiros*), the dominant species in the riparian habitat of UMRBNM, are dependent on regular pulses of high streamflow. Cottonwood seeds are released to coincide with spring floods. These floods not only disperse the seeds, but they also scour river banks and flood plains, creating the ideal habitat for the seeds, which require abundant moisture and minimal competition to germinate (Rood & Mahoney, 1990). Scott et al. (1997) found that most cottonwood trees along the Missouri River in UMRBNM were established within two years of events in which flows exceeded 1400 m³/s at the Fort Benton gauge (the equivalent of 1850 m³/s at the Virgelle gauge; Bovee & Scott, 2002), a flow high enough to place seeds above the reach of more regular seasonal flows and ice scours. These high flows occurred every 9.3 years on average between 1881 and 1992 (Scott et al., 1997), but it is unclear if this is still the case more than 30



While high spring streamflow will still likely be present under climate change, they are likely to occur earlier in the year due to earlier snowmelt, potentially desynchronizing them from cottonwood seed release. Though warmer winter and spring temperatures will probably make cottonwood seed release happen earlier, the influence of day-length might still cause release to occur after peak flows, impairing seedling establishment (Rood et al., 2008).



Figure 4. Arrow Creek, an undammed tributary of the Missouri river in UMRBNM, supports juvenile as well as adult cottonwood trees

Damming upstream of the monument has also disrupted streamflow (Bovee & Scott, 2002). As a result, young cottonwood stands in the monument are largely restricted to tributaries that have not yet been dammed (Fig. 4). However, increased annual precipitation, faster snowmelt, and an expected increase in extreme precipitation events throughout the year (Knapp et al., 2023), may still increase the frequency of flooding and high runoff events in UMRBNM (Whitlock et al., 2017). Whether greater streamflow volatility will make up for streamflow restrictions, which would only increase with longer, drier summers (Montana DNRC, 2015), requires further assessment.





Figure 5. The aftermath of an ice jam along the Missouri River.

Regular freezing events combined with a greater likelihood of warmer winter temperatures and possibility of rain-on-snow events would affect the frequency and intensity of ice jams in the future (Burrell et al., 2023).



Ice jams are another recurring event affecting riparian habitat in UMRBNM. Jams occur when river ice thaws rapidly and the resulting ice flow builds up, obstructing streamflow and causing water levels to rise. Ice jams have inundated flood plains along the Missouri river in a mix of water and ice, scarring mature trees, removing saplings, and scouring river banks much like spring floods. Though winter temperatures are expected to rise on the monument, even under RCP 8.5, mean winter temperatures will be just over freezing by the end of the century (Table 1), meaning the Missouri River will still likely freeze regularly for the foreseeable future.

Climate change is also expected to facilitate the spread of invasive species in riparian habitats on the monument. Three invasive plants in particular, Russian olive (*Elaeagnus angustifolia*), tamarisk (*Tamarix* spp.), and Siberian elm (*Ulmus pumila*), have been found on or near the monument, are drought tolerant, and benefit from higher annual temperatures (Perry et al., 2018).

While tamarisk and Siberian elm may simply outcompete or displace native riparian plants (Zouhar, 2003; USDA, 2014), Russian olive can change the successional dynamics of a habitat through its ability to

reproduce under shade and create self-replicating stands (Katz and Shafroth, 2003). Streams invaded by Russian olive are also less nitrogen limited, and the primary limiting nutrient may even shift to phosphorus (Mineau et al., 2011), potentially changing which plant species are most competitive in the habitat.

Climate change may make ash trees more susceptible to pathogens as well as invasive species. The fungus *Perenniporia fraxinophila* is common in the woodlands of the northern Great Plains, and adversely affects ash trees by weakening their heartwood, making their branches and trunks more likely to be toppled by wind or ice events (Lesica et al., 2003). It is expected that drought stress would make ash trees more vulnerable to the fungus (Lesica et al., 2003). Even more threatening to ash trees is the emerald ash borer (EAB; *Agrilus planipennis*), an invasive beetle that attacks and kills entire populations of ash trees (Aubin et al., 2015). EAB has not yet reached UMRBNM, but it is expected to do so eventually (Liang & Fei, 2014).



IV. INDIVIDUAL OBJECTS



Photo: Tobias Klenze

(A) GREATER SHORT-HORNED LIZARD

(Phrynosoma hernandesi)

The greater short-horned lizard generally inhabits shrublands and open coniferous forest, and, in Montana, it is most often found in areas with a large amount of bare ground and loose soil (Maxell et al., 2009). The size and shape of the lizard makes moving through denser vegetation difficult, implying that invasive plant species that generate in high densities, such as cheatgrass, may make habitats less suitable to them (Environment Canada, 2014). Greater short-horned lizards also require patchy

vegetation cover for the intermittent shade it provides, allowing them to thermoregulate as necessary (Environment Canada, 2014).

Given their ability to manage high temperatures, climate warming may benefit this species by reducing their time in hibernation, giving them more time to forage and reproduce (Maxell et al., 2009; Environment Canada, 2014).

However, overwintering lizards in Alberta have been found to require a certain amount of snow cover for adequate insulation, which suggests

that a potential decrease in snowfall may cause greater mortality in populations in UMRBNM (Environment Canada, 2014).

It is important to note that there is a lack of baseline data on greater short-horned lizards in Montana (Maxell et al., 2009). Further research may reveal additional risk factors.

Figure 6 (right). A greater short-horned lizard found in UMRBNM.



(B) GREATER SAGE GROUSE

(Centrocerus urophasianus)

Greater sage-grouse are obligate residents of the sagebrush ecosystem, requiring large areas of contiguous, heterogenous sagebrush to meet all of their habitat requirements. Sage-grouse broods prefer open stands of sagebrush in the summer (Montana sage-grouse Work Group, 2005), but in winter they move to denser stands tall enough to appear above the snow, which provide a critical source of food and shelter (Connelly et al., 2011).

Sage-grouse population declines are usually linked to juvenile mortality, as adult survival is considered high (Montana sage-grouse Work Group, 2005). Both nesting and rearing require a combination of shrub and grass cover for concealment from predators. When chicks hatch in late spring they also depend on abundant forbs and insects for forage (Remington et al., 2020). During dry summers, as palatability of forage declines, sage-grouse move to moist areas that still support succulent vegetation. Sage-grouse may remain dispersed during wet summers (Montana Sage Grouse Work Group, 2005).

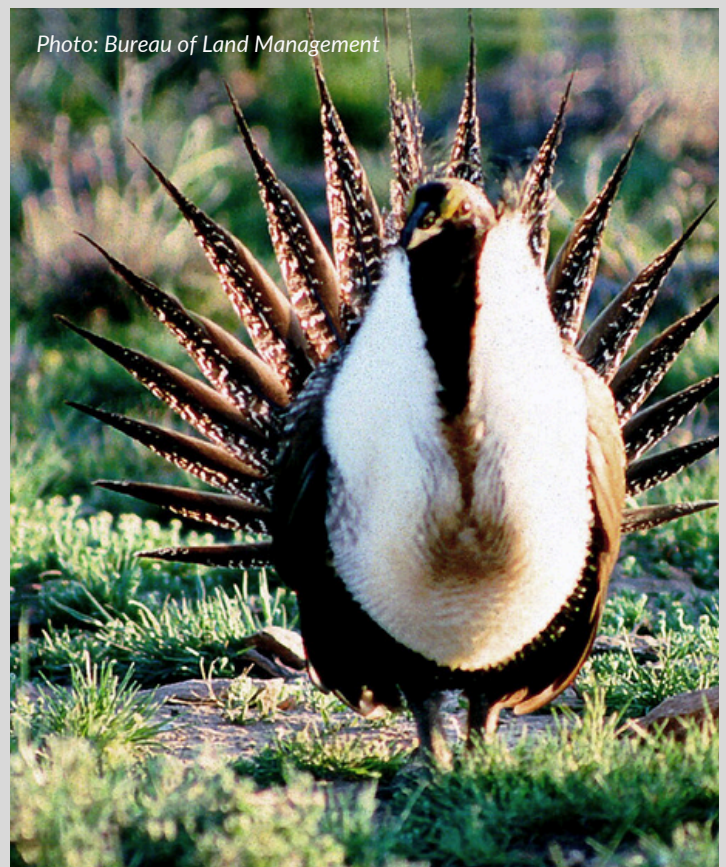




Photo: Oregon Department of Fish and Wildlife

The threats climate change imposes on sage-grouse are primarily linked to sagebrush habitat. Drought immediately after hatching (late spring and early summer) increases the already-high juvenile mortality rate by depriving chicks of food sources and predator concealment (Montana Sage Grouse Work Group, 2005).

Invasive grasses such as cheatgrass also displace and remove native grasses sage-grouse depend on for forage (Brummer et al., 2016). Wildfire can eradicate sagebrush habitat in the burned area for over 50 years, depriving sage-grouse of the habitat they depend on. Climate change stressors may also reduce the height of sagebrush (see section 3.b, paragraph 2), depriving sage-grouse of a vital winter resource.

Because climate change is not projected to remove sagebrush habitat from the northern Great Plains (Chambers et al., 2023), sage-grouse are expected to persist in the region (Palmquist et al., 2021). However, stressors brought on by climate change may still detrimentally affect populations in UMRBNM, especially through juvenile mortality. These stressors may also cause non-migratory populations to become migratory by reducing the amount of contiguous quality habitat for the species, forcing them to move between different wintering, nesting, and brooding habitats (Montana Sage Grouse Work Group, 2005).



Photo: Ken Frank

(C) PRONGHORN

(Antilocapra americana)

The pronghorn's long-distance vision and speed makes it well adapted to the gentle, rolling terrain of sagebrush steppe habitat (McNew et al., 2023). The amount and quality of forage on the steppe is especially important to pronghorn populations, both because they are uniquely dependent on consistent energy intake as opposed to fat reserves (Clancey et al., 2012), and because they invest more energy into reproduction than any other ungulate (Robbins & Robbins, 1979). Dunn and Byers (2008) found that a drought in the National Bison Range in western Montana caused a decrease in



overwinter survival and spring fecundity for pronghorn, but not any other resident ungulate species. Future droughts may be expected to have a similar, unique effect on pronghorn in UMRBNM by depriving them of the quality forage they require. There is also the risk that cheatgrass expansion will have a similar effect by outcompeting and displacing the native vegetation pronghorn rely on (Jones, et al., 2023).

Pronghorn depend on the vegetation of the sagebrush steppe for cover as well as for forage. Thick shrub density provides cover for newborn pronghorn to hide from predators (Jacques et al., 2015), and adult pronghorn rely on particularly tall sagebrush during the winter to provide cover from cold winds, and to produce areas of lower snow depth where it is easier for them to forage (Bruns, 1977; Ryder and Irwin, 1987). A potential increase in sagebrush density (See Section 3.B, paragraph 2) may provide newborns with more protection from predators, but a shift toward smaller individual sagebrush (Palmquist et al., 2021) could limit availability of the tall sagebrush that are relied on in winter.

One possible benefit of climate change to pronghorn is a reduction in winter snow depth as a result of warmer winter temperatures (Table 1). Pronghorn's relatively small hoof area hinders their ability to traverse snow, and makes it an especially costly energy expenditure (Telfer and Kelsall 1984). Snow depth also impacts pronghorn's ability to access forage (Malpeli, 2022), further reducing their energy reserves. Lower snow depths, and warmer winters in general, would likely counteract the negative effects of more persistent drought on pronghorn populations, though it is unclear as to what extent.

(D) ELK AND MULE DEER

(Cervus canadensis and Odocoileus hemionus)

The habitat requirements and adaptations of elk and mule deer are similar enough that they share the same vulnerabilities to climate change. Their vulnerabilities are similar to pronghorn as well, in that they rely on quality forage for energy, select for vegetation cover, and are negatively impacted by deeper snow depths and colder winters (Mackie, 1970). However, their ability to use forest as well as sagebrush habitat (Wood, 2004), and their ability to draw more energy from fat reserves (Robbins & Robbins, 1979), makes them less vulnerable to climate change than pronghorn.

Both mule deer and elk regularly use ponderosa forest as thermal cover from both cold winds and deep snow in winter and warm sunlight in summer (Wood, 2004). Forest habitat also provides more accessible

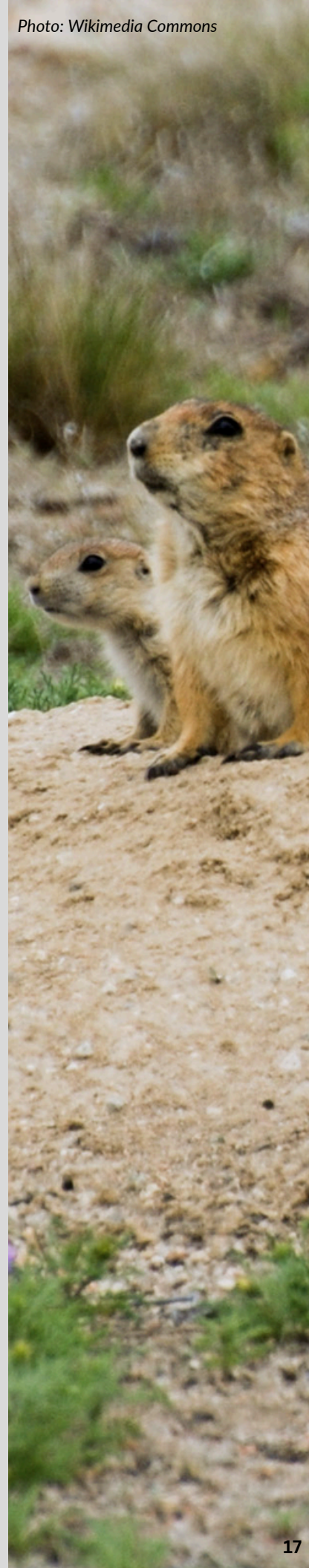
forage when ground vegetation is covered by snow (Wood, 2004). The potential for ponderosa pine to reduce growth under warmer and drier summer conditions (see Section 3.A, paragraph 4) may lead to shifts in stand structure, which could limit its effectiveness as a source for thermal cover and browse.

(E) BLACK-TAILED PRAIRIE DOG

(Cynomys ludovicianus)

Black-tailed prairie dogs inhabit large tracts of short to mid-grass habitat, and will closely crop this vegetation not just for their main source of food, but also to create habitat in which predators can be easily spotted and avoided (Van Pelt, 1999). This relationship to vegetation means prairie dog towns can change dramatically in periods of drought. In such conditions prairie dog towns become less dense and more dispersed, likely because of their improved ability to see predators over shorter vegetation, their need to cover more food resources, and their lower survival and recruitment (Bruggeman & Licht, 2020). On the other hand, warmer, wetter conditions that lengthen the growing season and improve forage quality may make towns more dense in response to more plentiful resources (Bruggeman & Licht, 2020). Greater interannual variability in rainfall might cause the size of prairie dog towns in UMRBNM to vary significantly over the course of years.

More interannual rainfall variability would likely exacerbate the spread of plague through prairie dog towns in the monument. Plague causes nearly 100% mortality in black-tailed prairie dog populations (Cully et al., 2010), and is described as the most significant threat to the species (Luce, 2002). Eads et al. (2020) found that fleas carrying plague were most abundant in Utah prairie dog towns (*Cynomys parvidens*) during dry years that followed wet years. They speculate humid conditions in prairie dog burrows provided an ideal microhabitat for flea production, and the following year's dry conditions limited vegetation production, making prairie dogs weaker and more exposed to ectoparasites. A model synthesizing black-tailed prairie dog colony extinctions from plague under different climate, soil, and colony characteristics verified this relationship, and pointed out that greater prairie dog densities during wet years also increase the likelihood of plague transmission (Barile et al., 2023). The effect of increasing temperature on plague mortality is less clear, as warmer temperatures shorten flea development time and increases their abundance (Samuel et al., 2022), but fleas seem to transmit plague more effectively in cooler temperatures (Williams et al., 2013; Barile et al., 2023).



(F) SAUGER, BLUE SUCKER, & SMALLMOUTH BUFFALO

(*Sander canadensis*, *Cycleptus elongatus*, and *Ictiobus bubalus*)

The life history and adaptations of sauger, blue sucker, and smallmouth buffalo are similar enough that they share the same vulnerabilities to climate change. All three are long-lived species that only reproduce under optimal conditions, which likely involve higher than average streamflow and water temperature being within a certain range particular to each species (Holmquist, 2024). Both of these conditions will likely become more variable under climate change (see Section 2, paragraph 5), which may limit recruitment for these species in the future. Warming water temperature may also disrupt the emergence of food resources that these species rely on (Lessard & Hayes, 2003), and benefit non-native species like smallmouth bass that are better adapted to warmer temperatures (Holmquist, 2024). Projected lower summer streamflow (Fig. 3) would be especially detrimental to sauger, which are adapted to highly turbid, low visibility environments (Ali & Anctil, 1977). Lower streamflow would decrease turbidity, making the habitat more suitable to non-native visual predators such as walleye, northern pike, and smallmouth bass (Holmquist, 2024).



Photo: USFWS Mountain-Prairie

(G) AMERICAN WHITE PELICAN

(*Pelecanus erythrorhynchos*)

The American white pelican is a migratory species, and flies to Montana each spring from its wintering grounds in the southern United States and central America (Montana Field Guide, 2024). Pelicans typically arrive in Montana in April and form large breeding colonies in wetlands in which they reproduce, lay and hatch eggs, and rear young until August,



Photo: Ingrid Taylor

when they begin migrating south (Hendricks and Johnson, 2002; Montana Field Guide, 2024). Though there are no known breeding colonies in UMRBNM, adults likely travel to the monument to forage for their young in the Missouri river and its tributaries.

The presence of pelicans in UMRBNM is determined more by the viability of the breeding colonies closest to the monument than conditions on the monument. The four known breeding colonies in Montana are at Medicine Lake, Bowdoin Lake, Arod Lakes, and Canyon Ferry Reservoir (Montana Field Guide, 2024). The most significant factor in breeding habitat viability is the depth of water that surrounds them.

Pelicans in Montana experienced significant declines in the 1960's and 1970's as a result of water level fluctuations at their breeding colonies (Montana Field Guide, 2024). Climate change in the pelican's wintering grounds may also affect breeding colony viability, as Sovada et al. (2014) has found pelicans' arrival at a breeding colony in North Dakota has advanced by 16 days over a 44-year period, exposing chicks to early spring storms that increase their mortality.



V. KNOWLEDGE GAPS

Photos (top to bottom): Bureau of Land Management (1), Bureau of Land Management (2), Matt Lavin, Neal Herbert, & NatureServe



Reservoirs upstream of UMRBNM may further restrict streamflow as a response to drier summer conditions from climate change. How significant an impact will increased water restrictions have on streamflow in the monument?



Higher atmospheric CO₂ concentrations improve plant growth and water-use efficiency. How significantly will higher CO₂ concentrations counteract the effects of drought-stress under climate change as well as influence invasive vegetation response and woody encroachment?



Climate change will likely increase the likelihood of stand-replacing fire in ponderosa pine forest, and may restrict ponderosa seedling establishment. Will this lead to a contraction in ponderosa pine habitat?



The NatureServe climate change vulnerability index provides a quick, cost-effective estimate of a species' vulnerability to climate change, but it requires detailed information on the species' natural history to generate predictions. Can the information needed to produce an estimate for the species of interest be pooled from monument staff and resources? ●

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