



Vegetation Response to Rocky Mountain National Park's Elk and Vegetation Management Plan

Analysis of 2008–2018 Data

Natural Resource Report NPS/ROMO/NRR—2023/2537





ON THIS PAGE

Aspen stand in Rocky Mountain National Park (NPS)

ON THE COVER

Willows inside enclosure HP40 along Fall River in Horseshoe Park, Rocky Mountain National Park, Colorado (NPS)

Vegetation Response to Rocky Mountain National Park's Elk and Vegetation Management Plan

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Executive Summary

Rocky Mountain National Park (RMNP) developed the Elk and Vegetation Management Plan (EVMP) to address well-documented declines in the ecological condition of aspen stands and riparian communities from high levels of elk herbivory. The EVMP aims to reduce the impacts of elk on vegetation and restore the natural range of variability in the elk population and affected plant communities, including preventing the loss of aspen clones within high elk-use areas, restoring montane riparian willow cover and height within suitable habitat, and reducing levels of elk grazing on herbaceous vegetation. The EVMP described a range of management actions including reductions in the size of the elk herd and the installation of fencing to reduce herbivory levels and facilitate recovery in degraded communities.

The EVMP established a monitoring protocol in focal communities to assess progress towards these vegetation goals and facilitate adaptive management. RMNP has collected data quantifying biomass offtake in upland herbaceous and riparian communities, willow height and cover, and aspen stand structure and regeneration periodically since implementation of the plan began in 2008. This report summarizes the results of analyses of EVMP data collected from 2008–2018, the last year comprehensive measurements were made.

The EVMP was developed with a focus on the primary elk winter range in the upper montane zone on the east side of RMNP. The plan distinguishes core elk winter range, where elk concentrate during winter, and noncore winter range areas that typically have less elk use. Data were also collected in the Kawuneeche Valley in the headwaters of the Colorado River on the west side of the park, an area supporting extensive wet meadows and areas once dominated by willow. Data were also collected to investigate the effects of wildland fires that burned through the winter range in 2012 to determine the potential for using fire as a management tool to achieve EVMP goals.

The overwintering elk population in the park has declined, from its peak of 1,500 animals in 2001, and over the course of EVMP implementation, from 614 animals in 2009 to 124 animals in 2019. Declines in the overwintering elk population may be best explained by increased cow elk harvest outside of the park, and, most notably, by a change in seasonal migration patterns and habitat use that have elk moving to lower elevation wintering areas following the fall rut.

In sites in aspen communities, stand structure was changed little or declined across sampling periods in unfenced plots with continued patterns of little regeneration and recruitment and steady progression toward stands dominated by large-diameter trees. However, there was a progression towards taller sapling heights inside fenced plots and recruitment of small diameter tree-sized stems. Fencing had large and positive effects on aspen stand structure, with different patterns observed in fenced and unfenced core winter range and noncore winter range. Increased recruitment was observed across the winter range but occurred mainly inside fenced plots. Aspen stem counts varied between time periods and in relation to wildfire, with fenced and burned plots on the core winter range having higher stem counts by 2018 than unfenced and unburned plots.

Willow height and cover increased over time in sampled sites, but positive trends were generally restricted to sites in fenced areas. Willow height also increased on noncore (all unfenced) winter range sites. Willow in unfenced core winter range sites had only minor increase in height from baseline (2008 for most sites) to 2018, but willow in fenced plots had greater height increases over the same time period. Noncore winter range willow sites had modest height increases over the 10-yr period. Mean willow cover increased nearly 5-fold compared to baseline conditions within the core winter range fenced areas and roughly 1.5-fold in noncore winter range. Willow cover was greater in unfenced than fenced plots at baseline, but the pattern was reversed in 2013 and 2018. The highest cover occurred in 2018 in fenced core winter range plots (mean = 70.8%) and unfenced noncore winter range plots (mean = 68.6%). Mean cover increased from 14.6% at baseline to 25.3% in 2013 and 70.8% in 2018 in fenced core winter range plots. Mean willow cover changed little in unfenced core winter range plots between baseline and 2018, although the range of cover values increased over time, and willow cover increases were modest in the noncore winter range. Fencing reduced or eliminated browsing from plots located inside fences, but offtake varied widely among unfenced plots. Patterns of willow browse intensity differed management subgroups (e.g., core and noncore winter range), and generally showed a downward trend between baseline and 2018 measurements.

Herbaceous offtake in upland communities was measured in the first sample period (baseline–2013) to assess levels of grazing on herbaceous vegetation, however it was determined that the associated EVMP objectives had been achieved so measurements were discontinued after 2013. Continued monitoring of upland shrubs indicated no shift from herbaceous dominated to shrub dominated communities after 2013. Noncore upland plots had higher shrub cover than core winter range plots across all time periods, but most differences between year and core/noncore had low probability of effect. Cover for individual species varied over time and winter range plots.

Moose presence has increased in winter range aspen and willow sites over the past decade, while beaver presence at our monitoring sites has decreased.

Results indicate that RMNP is making progress toward the vegetation objectives set out in the EVMP, however positive trends were most pronounced in plots protected from ungulate herbivory through fencing. Aspen recruitment was greatest in fenced plots. Likewise, trajectories of willow height and cover were positive in fenced winter range plots. Results demonstrate that fencing is an effective means of improving condition in aspen and willow habitats. Changes outside the fences were slower and less pronounced than inside the fences, however, the positive (if small) increases in willow height and cover and aspen regeneration as well as decreases in upland herbaceous offtake, indicate that decreased wintering elk populations are also contributing to improvement of habitat conditions on the elk winter range. In the Kawuneeche Valley, which has not traditionally been heavily used by overwintering elk but does experience summer elk and moose use, poor and declining habitat condition were recorded in unfenced willow and aspen sites.

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List of Acronyms

BL:	Baseline
CI:	Credible interval
DBH:	Diameter at breast height
ES:	Effect size
EVMP:	Elk and Vegetation Management Plan
GLM:	General linear model
MCMC:	Markov Chain Monte Carlo
NPS:	National Park Service
PD:	Probability of direction
RMNP:	Rocky Mountain National Park
ROMN:	Rocky Mountain Inventory and Monitoring Network
SD:	Standard deviation
Yr:	Year

Introduction

Rocky Mountain National Park (RMNP) supports diverse ecosystems providing habitat for numerous wildlife species including several large ungulates. Elk (*Cervus elaphus*) are particularly abundant, strongly shaping ecosystem patterns and processes in the habitats they utilize (Baker et al. 1997; Binkley et al. 2003; Kaye et al. 2005). Native to the RMNP region, elk populations have varied dramatically since Euro-American settlement in the 19th century (Singer and Zeigenfuss 2002; Zeigenfuss et al. 2011; Johnson et al. 2016). Elk were extirpated from the area prior to RMNP formation but reintroduced in 1913 and 1914 and by 1944, ~1,000 animals were estimated within the park boundaries (Packard 1947a). Populations were managed through active culling through the late-1960s when a “natural regulation” paradigm was adopted National Park Service (NPS)-wide. Lacking the presence of keystone predators or active culling, elk populations increased dramatically in RMNP and the neighboring Estes Valley in the following decades (Figure 1), leading to increasing conflicts between elk and humans both inside the park and in surrounding communities, and affecting the condition of habitats used by elk (Singer et al. 1998; NPS 2007).

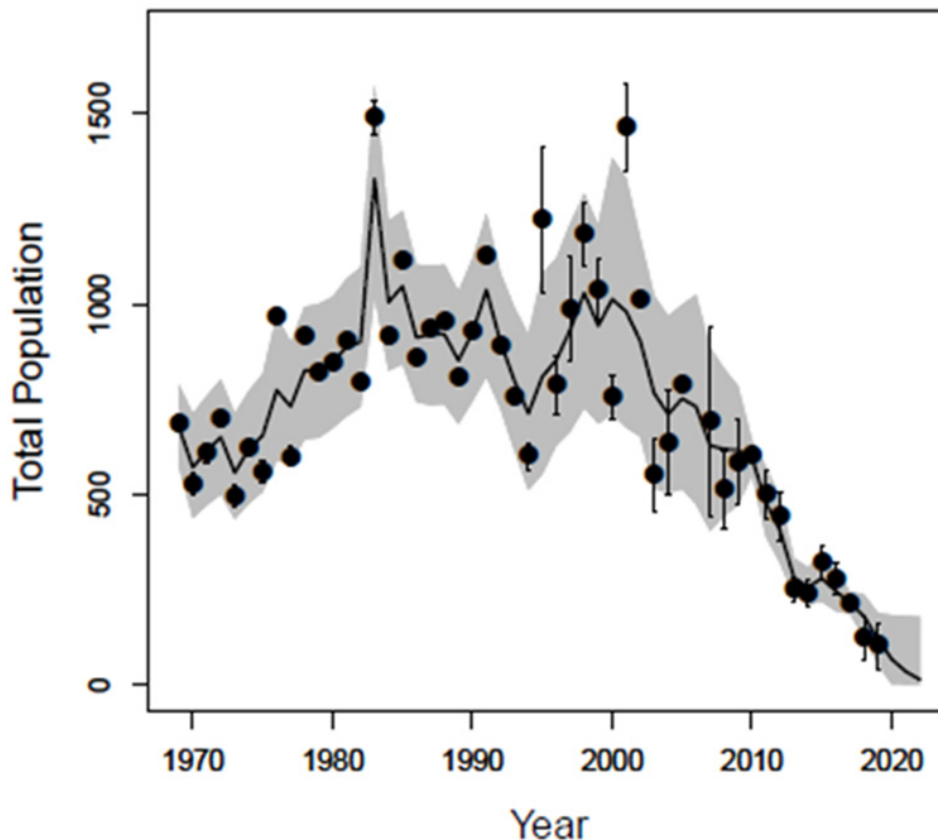


Figure 1. Annual population estimates (filled circles) and medians of the marginal posterior distribution of elk population size (solid line) on winter range in Rocky Mountain National Park during 1969–2019 and forecasts for 2020, 2021, and 2022. Forecasts assume 0 culling and use projections of harvest from 2019 through 2021. Shaded areas define 95% highest posterior density intervals on the true population size. Vertical lines are approximate 95% confidence envelopes on the mean of the adjusted census estimates (Hobbs and Abouelezz 2020).

Elk winter range areas are located at lower elevations on the east side of RMNP and the Estes Valley and surrounding foothills. The winter range is used in the spring, summer, and fall by elk migrating between summer range in addition to use by elk populations that overwinter in RMNP (Figure 2; Bear 1989; Larkins 1997; Lubow et al. 2002). The primary areas used by elk during winter are referenced as either core winter range (areas of high use) or noncore winter range (areas with less intense use). Elk diets vary seasonally, shifting from primarily herbaceous species in summer to include more woody plants, primarily willows (*Salix* spp.) and aspen (*Populus tremuloides*), in winter (Hobbs et al. 1981; Monello et al. 2006). As elk numbers increased after adoption of the natural regulation policy, growing evidence indicated a decline in the condition of aspen and riparian communities due to excessive levels of herbivory (Baker et al. 1997; Singer and Zeigenfuss 2002).

Elk herbivory strongly suppresses the growth of aspen suckers, reducing or eliminating their recruitment into the canopy and resulting in stands with few small or medium diameter trees, potentially threatening the long-term persistence of aspen clones in areas of the elk winter range (Olmsted 1979; Baker et al. 1997; Binkley 2008; Zeigenfuss et al. 2008). A long-term lack of regeneration can alter community structure, reduce diversity and negatively impact ecosystem resilience (Seager et al. 2013).

While structure of willow canopies naturally differs between willow species, with plant age, elevation, (beaver mediated) hydrologic gradients and along a human disturbance gradient (Bilyeu et al. 2008; Marshall et al. 2014; Schweiger et al. 2016), ungulate herbivory remains a key determinant of structure in many RMNP riparian communities (Peinetti et al. 2001; Singer and Zeigenfuss 2002; Zeigenfuss et al. 2002; NPS 2007; Zeigenfuss and Johnson 2015). Herbivore access to and consumption of willow branches can influence seed production and dispersal potentially limiting new willow establishment and threatening the long-term persistence of willows (Maschinski 2001; Gage and Cooper 2005; Cooper et al. 2006; Marshall et al. 2013).

Studies in RMNP demonstrated how excessive herbivory was contributing to dramatic changes in the structure of willow communities that dominate wet meadow and riparian areas (Singer et al. 1998; Singer and Zeigenfuss 2002; Baker et al. 2005, 2012). The canopy structure of riparian willows critically influences habitat suitability for beaver, birds, understory vegetation condition and other many other taxa and indirectly impact aquatic biota through effects on stream water temperature (Allen et al. 1983; Schulz and Leininger 1991; Shaw and Bible 1996; Stanley and Knopf 2002; Zeigenfuss et al. 2002; Baker et al. 2005; Moore et al. 2005; Baril et al. 2011; Schweiger et al. 2016). Elk were shown to suppress the height and areal cover of willows, species that provide essential forage and dam building materials for beaver, an ecosystem engineer that has historically shaped riparian areas in RMNP (Packard 1947b; Jones et al. 1994; Peinetti et al. 2002; Baker et al. 2005). Declines in beaver populations in RMNP were initially tied to trapping but in recent decades have been linked to a lack of tall willow needed by beaver for food and dam building (Monello et al. 2006; Schweiger et al. 2019), and this lack of tall willow has been tied both directly and indirectly to elk herbivory (Baker et al. 2005, 2012; Kaczynski 2013). Additionally, loss of beaver in these ecosystems can lead to a downward spiral for willow as habitats created by beaver dams are ideal for willow seedling establishment and growth.

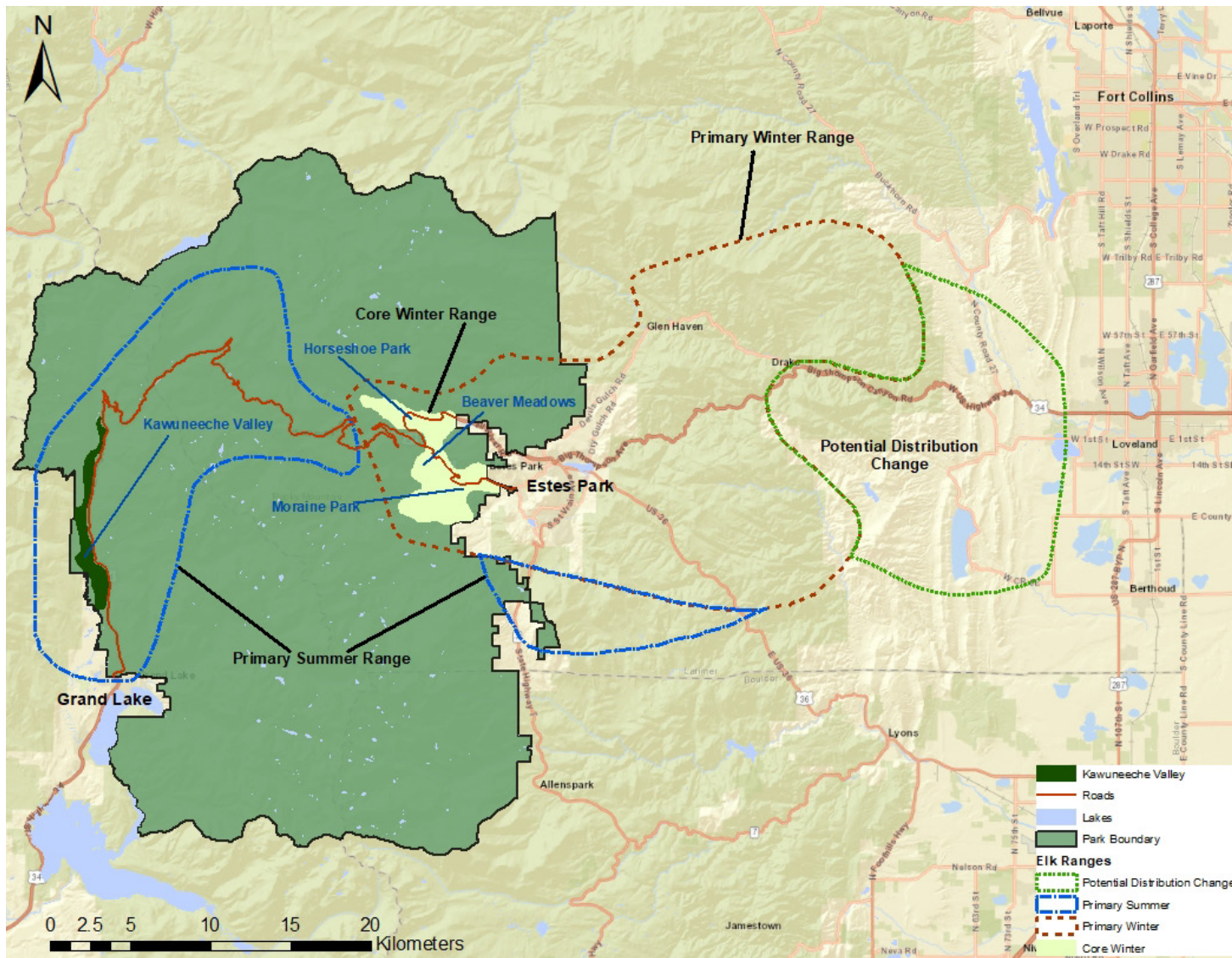


Figure 2. Approximate locations of primary winter range, area of potential distribution change where elk are now wintering in greater numbers, and primary summer range in Rocky Mountain National Park, Colorado (NPS 2007). The elk core winter range sits within the primary elk winter range and was the area with the highest concentration of elk when the Elk Vegetation Management Plan was written (NPS 2007).

Motivated by evidence documenting the decline of key aspen and riparian shrub communities (Olmsted 1979; Baker et al. 1997; Singer et al. 1998; Suzuki et al. 1999; Singer and Zeigenfuss 2002; Kaye et al. 2003), RMNP undertook development of a range of management alternatives in the early 2000s. After public and scientific comment, RMNP chose a suite of actions codified as the Elk and Vegetation Management Plan (EVMP; NPS 2007, 2008). The EVMP is intended to guide the management of elk and vegetation over a 20 year period, reduce the impacts of elk on vegetation, and bring target ecosystems into the natural range of variability for elk and vegetation in aspen, montane riparian willow, and upland communities (NPS 2007).

The EVMP relies on the principles of adaptive management (Holling 1978; Lancia et al. 1996; Walters 1996; Bormann et al. 1999). Adaptive management is defined by Bormann and others (1999) as “an approach to managing complex natural systems that builds on learning—based on common sense, experience, experimenting, and monitoring—by adjusting practices based on what was learned”. The EVMP includes management options such as habitat fencing, non-lethal redistribution and lethal culling of elk herds, and active vegetation restoration. The plan calls for the gradual lethal reduction of elk to achieve a wintering elk population in the park of 600–800 animals (NPS 2008). Fencing of up to 160 acres of aspen and 440 acres of suitable willow habitat in high elk-use areas of the primary summer and winter ranges was also included as part of this plan. The primary areas used by elk during winter (Figure 2) are referenced as either core winter range (areas of high use) or noncore winter range (areas with less intense use). RMNP constructed elk exclosures around patches of aspen and willow habitat in core elk winter range to exclude large herbivores and facilitate the recovery of vegetation beginning in 2008. An exclosure was also installed in the Kawuneeche Valley (Figure 2) elk summer range on the west side of the park in 2011, adding to areas already fenced as part of earlier research and restoration efforts on the east side of the park.

As part of the EVMP, RMNP and the U.S. Geological Survey (USGS) developed a vegetation monitoring plan focused on the goals of the EVMP (Zeigenfuss et al. 2011). The plan included measurement protocols and a statistical sampling design and monitoring schedule. Monitoring plots were established in aspen, willow, and upland communities in core and noncore parts of the elk winter range with initial (baseline) data collected from 2006–2009. In addition to these baseline data, subsets of plots have been sampled annually for vegetation offtake by ungulates and extensive sampling of aspen and willow shrub structure and composition across all plots has been conducted on a five-year (yr) basis. The first five-yr round of sampling was completed in 2013, and the second in 2018.

In the mid 2000’s overwinter elk populations inside the park approached 1,500 animals (Figure 1), declining to a median estimate of 614 animals in 2009 (Hobbs and Hoeting 2009). Lethal removal of 131 adult elk (almost all cows) from the park population occurred during the 2009/2010 and 2010/2011 winters, however, the number of elk overwintering in RMNP has continued to decline long after cessation of culling possibly as the result of habitat, land use, and management changes outside of RMNP. During the winter of 2018/2019, following the collection of the vegetation data analyzed in this report, the overwinter elk population estimate inside RMNP declined to a median estimate of 124 animals (Hobbs and Abouelezz 2020).

The animals that utilize RMNP elk winter range are part of a larger regional elk herd referred to as the E9 St. Vrain herd by Colorado Parks and Wildlife. Increased cow hunter harvest occurring outside of RMNP's boundary also contributed to lower regional elk population estimates, which may have contributed to a reduction in the number of elk overwintering in RMNP. Elk overwinter distribution appears to have shifted to lower elevations with animals occupying short grass prairie, agricultural, and suburban landcover types outside RMNP between November and March. However, elk are observed in the park in greater numbers during the spring calving and fall breeding seasons, occupying the winter range for many weeks outside the winter season. The effect of culling on patterns of elk distribution was presumed to be minimal, but the elk exclosures reduced the area available to elk possibly leading to an increase in willow offtake in unfenced core winter range plots and potentially noncore winter range as well (Zeigenfuss et al. 2011; Zeigenfuss and Johnson 2015).

Developing a greater understanding of elk space use and population estimates during nonwinter months may help to inform next steps in elk management as a part of the EVMP but is beyond the scope of this analysis. This report does not include specific analyses and interpretation of the relationship between elk population size and vegetation response. Rather it focuses on the primary objectives of documenting change (or lack thereof) in aspen, willow, and upland grassland habitats on the primary elk winter range of RMNP during the first 10 years since implementation of the EVMP based on the specific indicators and thresholds described in EVMP documentation (NPS 2007, 2008), and secondary objectives that address aspen and willow conditions in response to a fire that occurred on the winter range in 2012, as well as conditions in these habitats on the Kawuneeche Valley summer range.

Primary Objectives

The EVMP was focused on the influence of elk on vegetation structure and condition for three vegetation communities (NPS 2007; Zeigenfuss et al. 2011) on the primary elk winter range of Rocky Mountain National Park (Figure 2). Monitoring in the EVMP program is focused on assessing several community attributes (Zeigenfuss et al. 2011) including:

- Aspen stem density and stem size distribution assessed every 5 years through a full resampling of plots.
- Willow height and cover resampled every 5 years across plots.
- Vegetation offtake by herbivores in riparian shrub (with an emphasis on willow, *Salix* spp.) and upland herbaceous communities assessed through annual subsampling of plots.

The primary objective of this report is to assess progress toward the EVMP goals over the first 10 years of monitoring. The EVMP outlined the desired future condition for each of the target vegetation types and identified indicator variables associated with these desired conditions which would be monitored over the 20-yr life of the plan (NPS 2007). Thresholds were identified for each indicator variable in the EVMP to inform adaptive management of landscapes and ungulate populations (Table 1). The desired future condition of aspen on the primary elk winter range is an increase in aspen regeneration such that at least 45% of stands develop a regeneration cohort within each decade. Successful recruitment of young aspen will be reflected in a higher diversity of age classes so that the distribution of stem diameter at breast height (dbh) reflects many (approximately

75%) small-diameter stems, some (approximately 20%) medium-diameter stems, and few (approximately 5%) large-diameter stems. The desired long-term future condition of riparian montane willow is up to 70% willow cover within suitable willow habitat and an increase in the amount of willow that reaches a height beyond the reach of elk browsing (2–2.5 m height). The EVMP acknowledged that this level of recovery could not likely be accomplished within the 20-yr timeframe of the plan, therefore the plan calls for an annual progressive increase in willow cover on the elk winter range to at least 10 percent greater than baseline conditions and average willow height increasing to at least 20 percent greater than baseline conditions over the 20-yr life of the plan (Table 1).

Table 1. Vegetation types and their desired future conditions, indicator variables, and thresholds specified in the Elk and Vegetation Management Plan (NPS 2007; Zeigenfuss et al. 2011; Zeigenfuss and Johnson 2015).

Category	Desired future condition ¹	Indicator	Thresholds
Aspen	At least 45% of aspen plots across the winter range regenerating. Distribution of stem dbh reflects many (~75%) small-diameter stems, some (~20%) medium-diameter stems, and few (~5%) large-diameter stems	Stem density by height and diameter class	Progressive increase in aspen regeneration above the baseline level of 13% (presence of stems < 2 cm dbh reaching 150–250 cm tall). Progressive shift in the distribution of stem sizes toward the desired future condition
Riparian Montane Willow	–	Willow consumption	No net increase in annual willow offtake across the winter range above the baseline level of 35%.
	At least 31% willow cover within suitable willow habitat across the winter range.	Cover	Progressive increase in willow cover across the winter range above the baseline level of 21%.
	Average willow height of at least 1.1 m	Structure	Progressive increase in willow height across the winter range above the baseline level of 0.9 m
Upland Herbaceous	Reduction of the level of elk grazing on herbaceous vegetation and maintenance of a diversity of grazing levels across the landscape	Herbaceous consumption	No net increase in winter upland herbaceous offtake across the winter range above baseline levels of 61% ² , with ≤ 25% of sites with offtake > 70% and ≤ 10% of sites with offtake > 85%

¹Desired future condition of willow are specific to the 20-yr timeframe of the EVMP.

²Baseline level of offtake in the EVMP Environmental Impact Statement (NPS 2007) was 47%, however, the method for calculating baseline offtake was revised in 2013 resulting in a baseline value of 61% (Zeigenfuss and Johnson 2015).

Secondary Objectives

Although not the primary focus of the EVMP, aspen and willow monitoring plots were also added in the Kawuneeche Valley. The Kawuneeche Valley (sometimes referred to as the Colorado River Valley) on the west side of the park is mainly elk summer range and was not included in the original EVMP monitoring plan. Compared to core winter range sites like Moraine Park, this area supported a

higher density of riparian willows until the early 2000s. However, there is evidence of declining community condition likely influenced by herbivory from an expanding moose population, hydrologic impacts from the Grand Ditch, increased vulnerability to infection of willows by *Cytospora chrysosperma* (Woods 2000; Kaczynski and Cooper 2013; Kaczynski et al. 2014; Kaczynski and Cooper 2015a), and persistent impacts from historical and extant human use (Schweiger et al. 2016).

In recent years, moose have been increasing in abundance throughout the park, including on the elk winter range, which may have potentially significant consequences associated with increased willow herbivory. Areas included in the monitoring plan were also affected by the Fern Lake Fire that burned parts of Moraine Park in late 2012. Although not a focus of the original EVMP, this fire provided an opportunity to study willow and aspen post-fire recovery under browsed and unbrowsed conditions. In addition to clarifying changes in condition over time due to elk, the EVMP can be used for assessing trends due to these other factors. The results are presented in relation to the goals outlined in the EVMP and past analyses of herbivores and ecological condition.

EVMP 5-year Monitoring Results and Expert Workshops

The first 5-yr vegetation monitoring sample was collected in 2013, and the data were analyzed to look at changes since EVMP implementation (Zeigenfuss and Johnson 2015). By 2013, the overwinter elk population size had dropped below 300 and overall, there was steady progress toward the vegetation objectives set out by the EVMP. Increases in the number of small-diameter, tree-sized (stems greater than 2.5 meter height) aspen stems were observed but only inside fences that excluded ungulates. In unfenced areas, stand structure was stagnant, with many medium- and large-diameter (older) stems and no replacement of small-diameter stems. Aspen saplings (stems less than or equal to 2.5 meter height) were recruiting on 29 percent of sampled sites, but this was mainly due to growth inside fences. Upland herbaceous offtake dropped below baseline levels (61%) on both core and noncore winter range and less than 10% of the upland grasslands were experiencing intense herbivory. Winter willow offtake remained steady from 2009 to 2014, showing no substantial increases nor consistent declines, and was below the baseline level of 35 percent in 2013 and 2014. Mean willow height and cover increased over the first 5 years within fenced habitat on the core winter range and on noncore winter range. Habitat fencing was concluded to have been the most effective means of improving aspen and willow habitat conditions.

After the first 5-yr sampling and data analysis was completed in 2013, two workshops (one addressing elk population objectives and the other vegetation objectives) were convened in 2014 with the purpose of evaluating the findings of the monitoring program thus far and recommending adjustments going forward. University and government scientists with expertise in the target vegetation and ungulate communities were enlisted to contribute to the workshop. During these workshops, recommendations were made regarding future management actions and monitoring, including changes to the offtake protocols and the need to develop an understanding of the use of the elk winter range by moose as their populations continue to disperse from the west side of the park.

The expert panel recommended RMNP managers consider taking additional management actions to further facilitate vegetation recovery and support quicker removal of fences. EVMP criteria for

willow recovery in the 20-yr timeframe of the plan was intentionally low as recovery was anticipated to be slow and having met those modest objectives within the first 5 years of EVMP implementation, RMNP managers should work toward the long-term goals of 70% willow cover in suitable willow habitat and willow heights tall enough to be out of reach of elk browsing. The panel also noted that while aspen stems of 2 cm dbh and 2.5 m tall may be considered successful recruitment by EVMP criteria, stems of this size may not be large enough to survive elk browsing after fences are removed and they recommended that aspen stems be 4 cm dbh and 4 m tall prior to considering fence removal (T. Johnson, written notes).

An implicit goal of management in the EVMP is the return of beaver to riparian areas as beaver are an important component of fully functioning, sustainable ecosystems. Data from EVMP monitoring through 2013 demonstrated the positive effect of fencing on the height and cover of riparian willows and suggest increasing suitability for beaver. Re-establishing beaver would require even more woody biomass, therefore the 2014 expert panel noted that there remained significant potential to increase willow cover through the establishment of new plants through planting of willow and other riparian species (alder [*Alnus incana*], birch [*Betula occidentalis*], cottonwood [*Populus angustifolia*]). Additionally, increasing water tables could help increase willow height and the use of simulated beaver structures (aka beaver dam analogs) to achieve this was encouraged. Establishment of new willows by seed requires disturbed soils which could be facilitated by management actions such as scarifying or burning selected areas inside fences. Similarly, the 2014 panel suggested the extent of existing aspen stands could be broadened using trenching or light burns just enough to darken the soil to stimulate suckering in new areas. The continued monitoring of burned aspen and willow plots was recommended to determine relationships between herbivory, fire, and plant recovery.

Additional management actions could be taken to facilitate the increase in willow and aspen biomass (e.g., actions to increase willow height and cover inside fences; increase the footprint of aspen inside fences using methods such as trenching and low intensity burns; establish aspen stands in strategic locations near riparian areas). Management actions to increase other woody species such as alder, birch and cottonwood could also be important to supporting beaver recovery. The panel concluded that continued growth of existing willows was expected inside fences (at least in some places), and maybe to some extent outside fences, however, willow height was likely limited in some places by water availability.

Continued aspen growth, sapling regeneration and recruitment of tree sized stems is expected inside fences, and to some extent in non-core areas. Additional fences would possibly be needed to achieve the aspen objective for percentage of recruiting stand. Vegetation recovery outside fences depends on the amount and season of elk use, which is not well understood, and the expert panel identified the need to develop a better understanding of elk use of the winter range during summer and spring/fall migration to provide information to determine the feasibility and likelihood of vegetation restoration outside fenced areas. The 5-yr monitoring results indicated management actions, such as fencing, planting, and elk redistribution, would be needed to stop the continued decline of willow and aspen conditions in the Kawuneeche Valley as well.

Methods

Study Area

The EVMP was developed with a focus on the primary elk winter range encompassing three major valleys in the upper montane zone on the east side of RMNP (Marr 1961; NPS 2007). The plan distinguishes core elk winter range, where elk concentrate during winter (Moraine Park, Beaver Meadows, Horseshoe Park), and noncore winter range areas that typically have less elk use; Figure 2). Data have also been collected in the Kawuneeche Valley in the headwaters of the Colorado River, an area supporting extensive wet meadows and areas that were formerly willow-dominated riparian shrublands, to provide information on this critical and changing area of the park.

Common species in riparian shrub communities include *Betula occidentalis* (Western river birch), *Alnus incana* subsp. *tenuifolia* (thinleaf alder), and several willows including *Salix monticola* (Bebb), *S. geyeriana* (Andersson), *S. planifolia* (Pursh), *S. bebbiana* (Sargent), *S. drummondiana* (Barratt), and *S. lucida* subsp. *caudata* (Nutt.) E. Murr. Upland and wetland meadows are dominated by native and introduced grasses, sedges, and herbaceous dicot species, with additional cover provided by other shrubs. Aspen communities are characterized by a canopy of trembling aspen with aspen suckers, grasses, herbaceous dicots, and shrubs dominant in the understory. Upland grasslands and shrublands make up 43% of the winter range (Monello et al. 2006). Dry grasslands are made up of a variety of grasses (*Agropyron* spp.) and sedges (*Carex* spp.), timber oatgrass (*Danthonia intermedia*), Parry's oatgrass (*Danthonia parryi*), and needle-and-thread grass (*Hesperostipa comata*). Upland shrub areas consist of antelope bitterbrush (*Purshia tridentata*), big sagebrush (*Artemisia tridentata*), rabbitbrush (*Chrysothamnus viscidiflorus*), blue grama (*Bouteloua gracilis*), and mountain muhly (*Muhlenbergia montana*).

Each valley in the park has a distinct ecological history, influencing current vegetation composition and structure. Factors such as historical land use and herbivore dynamics, geomorphic processes and hydrologic regime vary among EVMP study areas with many areas seeing significant changes since Euro-American settlement. For example, in Moraine Park (a core winter range site), floodplains along the Big Thompson River were historically dominated by willows with abundant beaver (Packard 1947b). Agricultural and recreational development prior to the formation of RMNP coupled with high levels of elk browsing have severely degraded willow communities, and beaver are now mostly absent (Peinetti et al. 2002; Singer and Zeigenfuss 2002). Willows are commonly in the "short-old" morphotype due to ungulate browsing (Peinetti et al. 2002; Baker et al. 2005), with taller willows limited to the western-most portion of the valley and within exclosures built to eliminate ungulate herbivory.

In addition to elk, mule deer (*Odocoileus hemionus*) also use areas in the park year-round, mainly upland shrub communities. Moose (*Alces alces*) are another significant herbivore affecting RMNP aspen and riparian communities. Introduced northwest of RMNP in the late-1970s, their numbers have increased dramatically since they first appeared in the Kawuneeche Valley in 1980. In recent years, moose have been observed increasingly on the east side of RMNP (Zeigenfuss and Abouelezz 2018).

Sample Design

The aims of the EVMP included improvement of aspen and willow conditions inside fences, while ensuring at a minimum that conditions do not decline outside fences. The sample design attempted to monitor the effects of the full range of management actions being implemented by using a stratified random sampling design that stratified each focal vegetation community into elk winter-range zones of fenced core winter-range, unfenced core winter-range, and noncore winter-range, thus allowing separate monitoring of vegetation responses to management actions for each type of range (Zeigenfuss et al. 2011). Vegetation communities of interest were selected from the park vegetation map (Salas et al. 2005) using ArcMap (v.10.8), clipped to the winter-range boundary, and then further subsetted to represent core and noncore winter range and potential fenced areas. Random points were generated in each vegetation type and elk winter-range zone and monitoring sites were selected from these points. Details of sample design can be found in the report that presents the monitoring plan and describes baseline data collection (Zeigenfuss et al. 2011).

The distribution of monitoring plots varies by vegetation type with original targets of 35 plots each in the core and noncore winter range for willow, 20 each for aspen, and 25 each in upland vegetation with an additional 20 plots each in fenced core winter range aspen and willow (Zeigenfuss et al. 2011). Original targets fell slightly short of these goals and the design was further complicated by effects of fire and floods (Table 2; Figures 3 and 4). Additional plots were established in 2013 along with some plot replacements necessitated by the construction of new exclosures (Zeigenfuss et al. 2011; Zeigenfuss and Johnson 2015). Excluding plots removed from the data set due to issues like the 2013 summer flood, there were 246 plots in total at the time of this analysis—123 willow plots (49% of total EVMP plots), 75 aspen plots (31%) and 48 upland grassland/shrubland plots (20%)—distributed among fenced and unfenced plots in the core winter range, noncore winter range, and Kawuneeche Valley (Table 2).

Table 2. Count of Elk and Vegetation Management Plan (EVMP) plots by vegetation community type and range type in Rocky Mountain National Park. Parentheses denote number of plots that were burned in 2012 fire.

Range type	Total number of EVMP sample plots (and count of burned plots) by vegetation type		
	Aspen	Upland	Willow
Core winter range–unfenced	32 (4)	25	46 (13)
Core winter range–fenced	14 (3)	0	34 (15)
Noncore winter range	21 (1)	23	34
Kawuneeche Valley	8	0	9*

*Four of these plots are located inside a single fenced exclosure

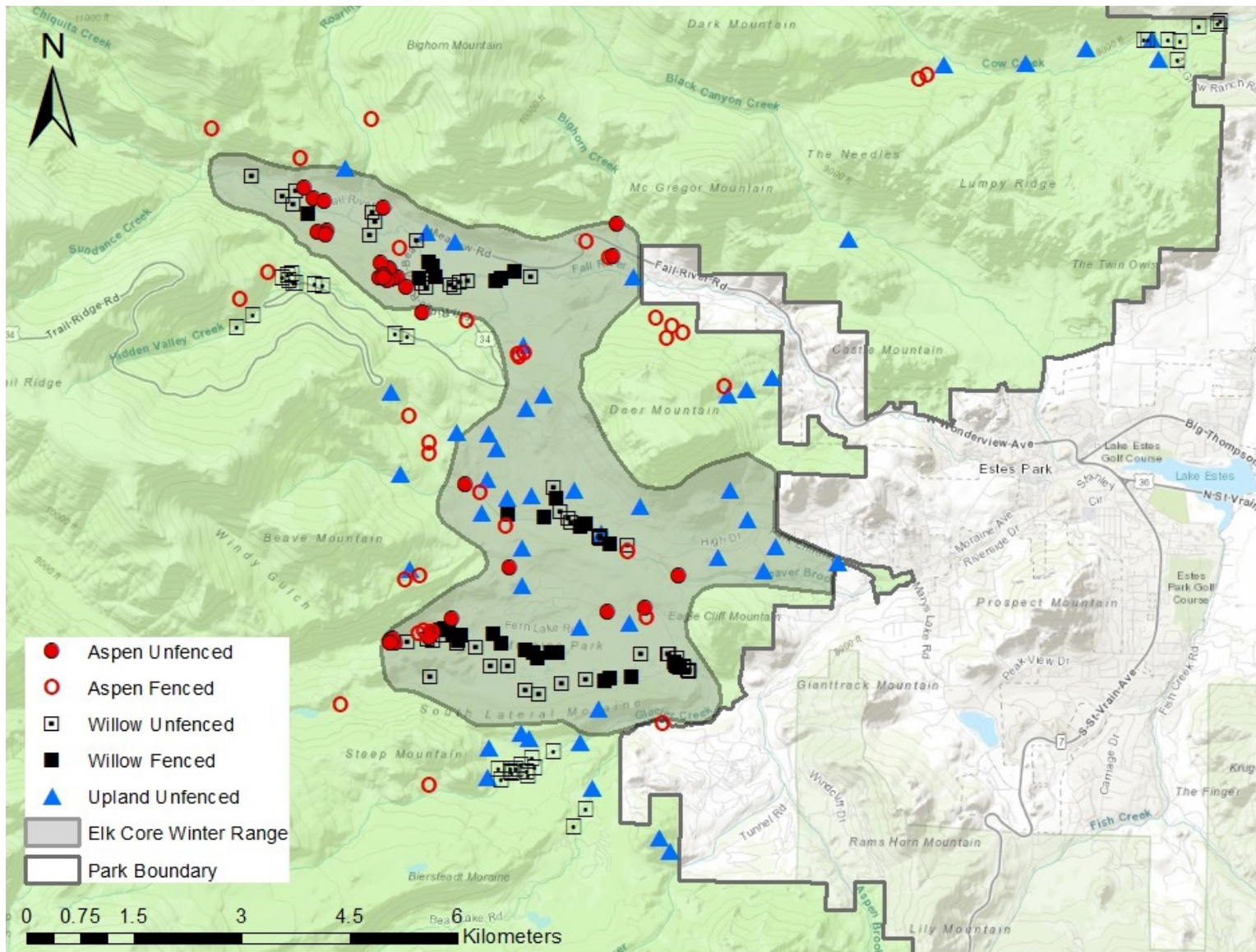


Figure 3. Location of Elk and Vegetation Management Plan plots by vegetation type and fencing status used in plot stratification on the Rocky Mountain National Park elk winter range at time of 2018 sampling.

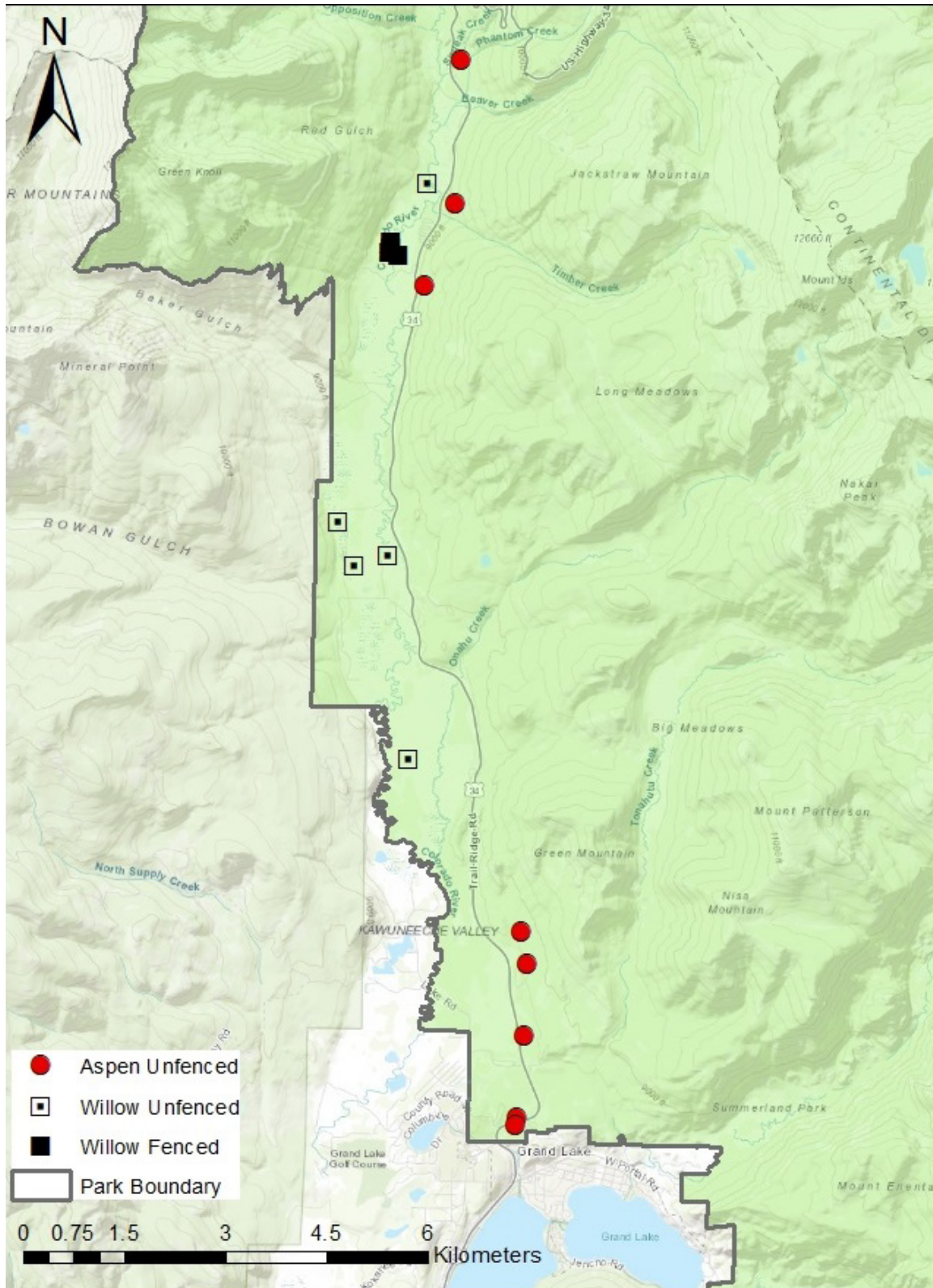


Figure 4. Location of Elk and Vegetation Management Plan plots by vegetation type and fencing status in the Kawuneeche Valley, Rocky Mountain National Park, at time of 2018 sampling. All fenced plots in the Kawuneeche Valley are located in willow habitat.

A wildfire burned through Moraine Park in December 2012 causing variable levels of plant mortality (Kaczynski and Cooper 2015b) and affecting 31 sampling plots, 17 of which were replaced due to the severity of burn effects. Burned plots occurred in both fenced and unfenced areas, offering an opportunity to evaluate wildfire effects on aspen and willow communities with and without browsing, so these original plots were retained for sampling. Fire can stimulate abundant suckering from aspen root systems (Brown and DeByle 1987). These suckering events may potentially lead to rapid recruitment of young aspen into the canopy within fences, but this result will likely not be seen outside the fences. To determine aspen responses to both fire and herbivory, additional burned sites were added to the monitoring program in 2013. Annual monitoring of the burn/unburn subset of plots for 5 years (2014–2018) was recommended by Zeigenfuss and Johnson (2015) to provide information on the short-term effects of fire on aspen regeneration and recruitment in the presence and absence of elk herbivory, however, this recommendation was not acted upon until 2015. These sites will be sampled every 5 years after 2018 for the duration of the EVMP to provide information on long-term effects of wildfire and ungulate herbivory. No willow plots in the noncore winter range were burned and only one aspen plot in the noncore winter range was burned (Table 2), so sampling and inference for fire effects are restricted to the core winter range.

The final EIS and Record of Decision (NPS 2007, 2008) included the Kawuneeche Valley in the scope of the EVMP but did not include separate indicators and thresholds for this area. RMNP managers recognized that it was unrealistic to apply the winter range thresholds to the Kawuneeche Valley because 1) the main use of this area by elk occurred in the summer; 2) the confounding factor of moose browsing which was occurring in the Kawuneeche Valley but not the winter range at that time; and 3) the condition of willow and aspen in the Kawuneeche Valley was much poorer than the winter range. All these factors combined would make it difficult, if not impossible to achieve the EVMP vegetation objectives in the Kawuneeche Valley within 20 years without more intensive management. A separate, robust sampling design would be necessary to address all these issues for Kawuneeche Valley aspen and willow and given funding limitations at the time the EVMP was initiated, RMNP managers determined that only limited monitoring would be conducted in the Kawuneeche Valley (Linda Zeigenfuss, Therese Johnson, personal communications). In this report, Kawuneeche Valley data are presented separately from winter range data and do not contribute to the assessments of progress toward EVMP goals.

Habitat restoration exclosures, built to exclude elk, are a key element in the EVMP. Comparisons of vegetation characteristics (e.g. height, canopy area) in fenced and unfenced areas is of particular interest (Zeigenfuss et al. 2011; Zeigenfuss and Johnson 2015). Elk exclosures were only erected with the goal of improving habitat condition inside the elk exclosures. Only willow and aspen habitats were fenced on the core winter range and some willow habitat was fenced in the Kawuneeche Valley. No upland habitat was fenced (Figures 3 and 4).

Baseline sampling occurred from 2006 to 2009 on the winter range (2011 through 2014 for Kawuneeche Valley willow), with revisits occurring through 2018. Sampling of all plots was done in 2013 and 2018, with subsets of plots measured in intervening years. To capture the effects of weather and ungulate population size on annual consumption rates, offtake was measured annually on a

subsample of plots using a serially-alternating sampling design with a 20-yr monitoring period (2009–2028) divided into five panels (Urquhart and Kincaid 1999; Zeigenfuss et al. 2011). Because shrub height and cover and aspen recruitment change more slowly, they were measured on a 5-yr rotation (baseline, 2013, 2018).

Vegetation Data Collection

Field data were collected following the protocol of the EVMP monitoring plan (Zeigenfuss et al. 2011). Methods are only briefly summarized here; for full details on sampling design and field protocols, see the original monitoring plan (Zeigenfuss et al. 2011). Baseline data collection occurred during site establishment (2006–2008). As fences were built, additional sites were added so baseline for these sites ranged from 2009–2012. Baseline years are identified in the individual vegetation sections. The EVMP provides a timetable for sampling as well as specifying field methods for the different variables measured in each community analyzed:

- Willow
 - Percent offtake from 0 to 100%
 - Willow cover (%) and height
- Aspen
 - Stem density and height
- Upland
 - Percent offtake from 0 to 100%
 - Shrub cover from 0 to 100% and height

Aspen

Regeneration and recruitment

To assess aspen regeneration and recruitment, all live and dead aspen stems greater than 2.5 m in height (trees) were tallied inside 25 m² square plots (Figure 5). Stems were placed into three diameter classes: small-diameter (0–10 cm diameter at breast height, dbh), medium-diameter (10–20 cm dbh), and large-diameter (greater than 20 cm dbh). Small diameter stems were further broken down into newly-recruited stems of 0–2 cm. Aspen stems less than or equal to 2.5 m in height (saplings) were tallied by height class (0–50 cm, 51–100 cm, 101–150 cm, 151–200 cm, 201–250 cm). To evaluate regeneration, raw stem counts were directly analyzed and modeled. In addition, stem counts were converted to stem density (stems/acre) by dividing by plot area and scaling up to an acre.

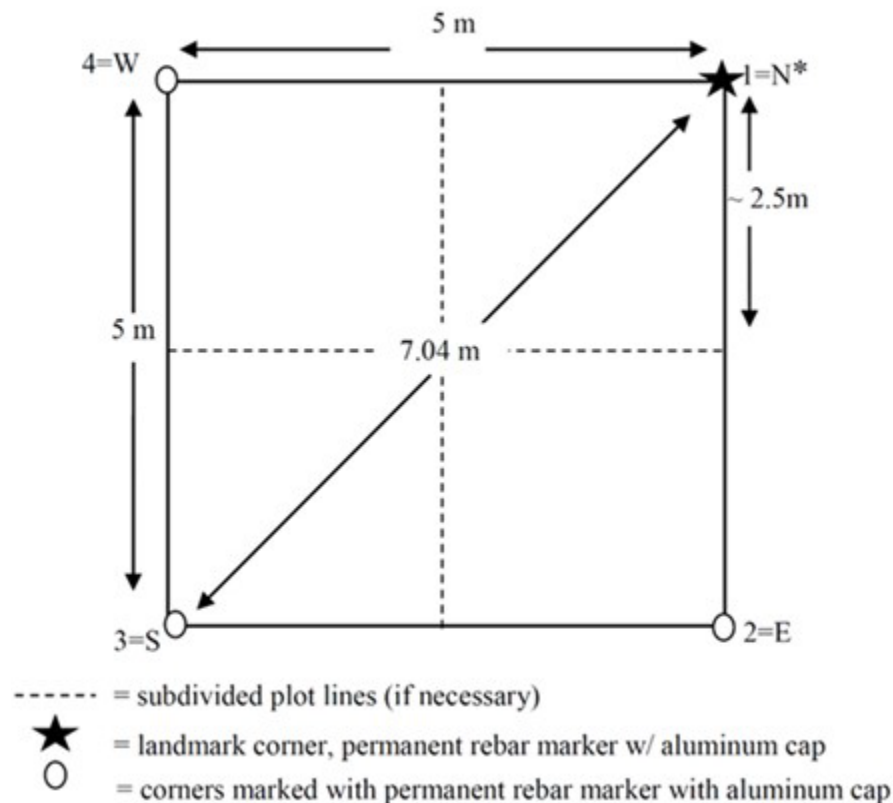


Figure 5. Diagram of aspen plot. More details are found in the monitoring plan (Zeigenfuss et al. 2011).

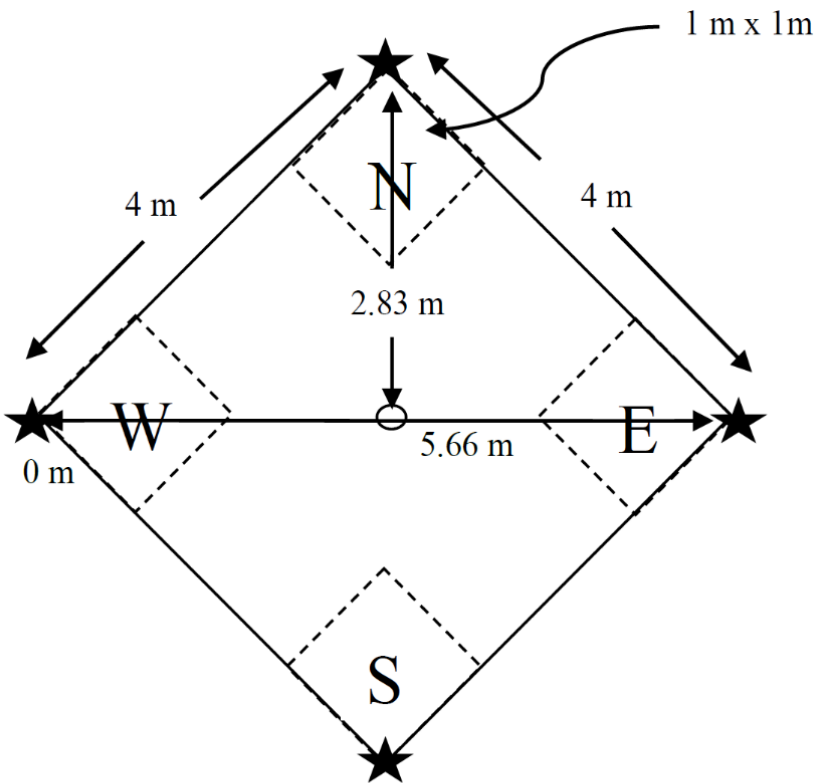
Willow

The majority of the original willow plots were established in 2008 with over half of those plots measured for offtake in the spring of 2009 (Zeigenfuss et al. 2011). Photos were collected and measurements were made of shrub cover and height, species composition, and winter offtake at each plot to form the baseline data. Elk exclosures were constructed in three locations on the core winter range in 2008. The remaining original set of willow plots were established within these fenced areas and had little or no offtake during the winter and were initially measured in 2009–2010. Plots were added over the following years in response to loss of sites due to floods and fire, and to provide a subset of paired sites in burned and unburned areas affected by fire. Sites were added in the Kawuneeche Valley starting in 2011 including sites inside a willow exclosure established in 2012. See Zeigenfuss et al. (2011) and Zeigenfuss and Johnson (2015) for details.

Height and cover

Two methods were used for assessing willow cover and height—macroplot and line intercept. All willow species within a 16 m² square plot (macroplot) were identified to species and measured for cover and height (Figure 6). Macroplot willow canopy cover was derived by calculating the canopy area of each individual willow that intersected the macroplot (canopy diameter measures were converted to area measures using the elliptical area formula), then multiplying by the proportion of the willow plant that fell within the macroplot as detailed in the EVMP field protocol (Zeigenfuss et al. 2011). Individual canopy areas were then totaled for the plot and divided by the macroplot area (16 m²) and multiplied by 100 to derive percent willow cover estimate for the plot. Macroplot willow

heights were determined by averaging height measures of all willows that fell within the plot. All height measurements included both the maximum height (tallest stem on the plant) and the average height (height of the majority of an individual shrub's canopy) for each measured plant. Willow cover and height were measured separately by species along a 5.7 m-long transect bisecting macroplots from east to west following the methods described previously for upland shrub cover. Line intercept willow canopy cover was calculated by totaling the entire length (in meters) of the transect line that was intercepted by willow for the plot and then dividing by 5.7 m (the length of the transect line) and multiplying by 100 to get a percent cover value. Line intercept mean heights (maximum and average canopy height) for each plot were derived by averaging across all willow on the transect line at a plot. Although, the information gathered using both methods is similar, the macroplot method emphasized in this report is more accurate and intensive than the line intercept method (Zeigenfuss and Johnson 2015). Both methods are built into the protocol to allow flexibility for less intensive data collection in the event that there is a lack of supporting funds over the 20-yr timeframe of the EVMP. We did analyze the line-intercept data and results are available in supplemental materials.



- ★ = permanent rebar marker w/ aluminum cap
- = plot center, temporary marker only

Figure 6. Schematic diagram of plots used to measure variables in willow plots. More details are found in the monitoring plan (Zeigenfuss et al. 2011).

Browse measures

Willow offtake, defined as the percentage of current year's annual willow shoot growth removed by browsing ungulates (Zeigenfuss et al. 2011), was measured on a subsample of willow plots beginning in 2009 and continuing through 2014. Details of data collection methods are described by Zeigenfuss and others (2011) and Zeigenfuss and Johnson (2015). Following the advice of an expert panel that reviewed the program in 2014, measurements of offtake were revised in 2015. Currently, measurements are taken using a browse intensity (BI) index which is simpler and faster to measure. This index, also known as percent leader use, measures the percent of twigs browsed on a stem, based on a subsample of twigs and stems selected following methods described by Bilyeu and others (2008) (Equation 1). The browse intensity index measurements were gathered and used as part of the previous measure of offtake and have been collected since the inception of this monitoring work, thus allowing for a continuous dataset which utilized this method. Most measurements of willow browse intensity (BI) were made in mid-April through May, although weather pushed sampling as late as mid-July in some years.

$$BI = \left(\frac{b}{b + u} \right) * 100 \tag{1}$$

where, b , the number of browsed shoots on the stem; and u , the number of unbrowsed shoots.

Upland

Shrub cover and height

Upland line intercept transect data were collected in core and noncore areas. Shrub species, cover, and height were measured for each shrub whose canopy intercepted a permanently marked 30-m transect line through the plot (Figure 7). Shrub cover was calculated by totaling the entire length (in meters) of the transect line that was intercepted by shrubs for the plot and then dividing by 30 m (the length of the transect line) and multiplying by 100 to get a percent cover value. These measurements were taken from all upland plots at baseline, 2013, and 2018. Herbaceous offtake was collected during the first sampling period (baseline through 2013) but was discontinued because reduction of elk population and redistribution of herbaceous offtake in line with EVMP objectives were achieved during that time period.

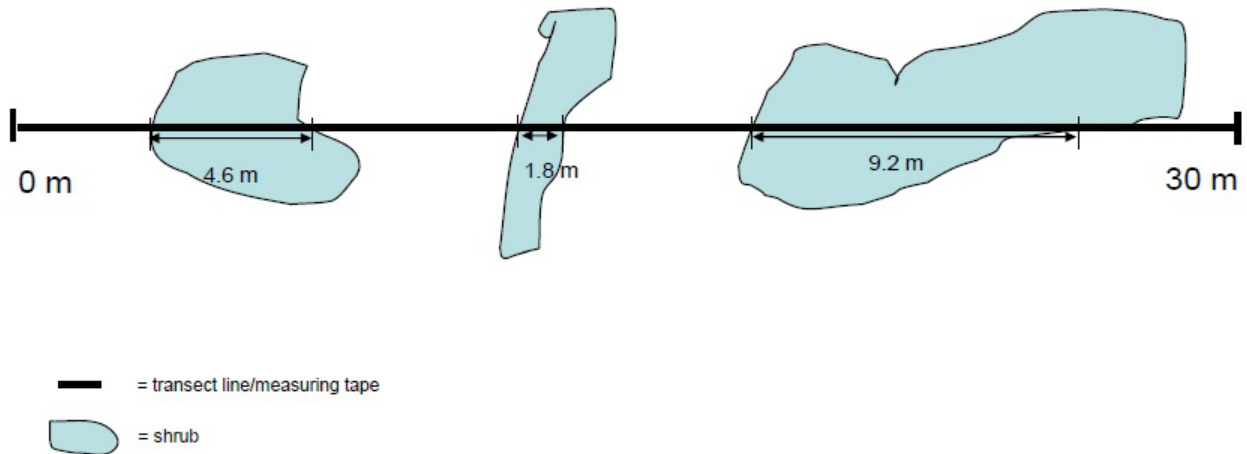


Figure 7. Schematic diagram of line-intercept method to measure upland shrub cover. More details are found in the monitoring plan (Zeigenfuss et al. 2011).

Winter-Range-Wide Estimates of Aspen and Willow

We used weighted averages based on the amount of aspen and willow area available in the core and noncore winter range to report winter-range-wide estimates of cover, heights, and aspen regeneration. The weighted averages were adjusted in 2013 to account for mean values and area of the winter range that was fenced and burned as well. These estimates are used to evaluate progress toward EVMP goals for each sample year.

Moose and Beaver Presence

Due to revisions made to the monitoring protocol over time, sampling consistency for moose and beaver presence was variable across years. Elk presence, per se, was not recorded because a separate effort surveyed elk winter populations each year and reported elk population size (Figure 1; Hobbs and Abouelezz 2020).

Moose presence was not recorded in any vegetation plot types during the baseline sampling period and was recorded only in aspen plots in 2013. An expert panel that was convened in 2014 to review EVMP results through 2013 identified the apparent increase in moose dispersal from the west side of the park to the east side as a critical factor potentially influencing vegetation changes on the elk winter range. Moose presence was recorded in all three vegetation plot types in 2018. Indicators such as tracks, scat, browse, and moose sightings within the boundary of each plot were considered evidence of presence.

Beaver presence was recorded in willow plots during baseline, 2013, and 2018, as well as in aspen plots in 2013. It was not recorded in aspen plots during baseline or 2018, or in upland plots for any sampling years. Indicators such as food caches, cuttings, dams, lodges, tracks, scat, and beaver sightings within the boundary of each plot were considered evidence of presence.

Data Analyses

Data were transferred from paper datasheets to MS Excel worksheets following baseline and 2013 data collection. In 2018, some data were electronically recorded on tablets in the field and then

downloaded to Excel. Baseline data and 2013 data were then checked between data sheets and electronic spreadsheets for accuracy of data entry. Raw data provided by RMNP as MS Excel workbooks were combined and cleaned using R statistical software (R version 4.0.3) to facilitate exploratory data analyses, quality assurance, and statistical modeling. Limited quality assurance was conducted on the raw data, including checking electronic data against paper data sheets and exploratory data analysis to identify anomalous data points. Specific issues encountered and addressed included inconsistently named/typed factors; missing values; and data values outside of expected range or showing unusual patterns. Derived data sets were produced and exported for use in statistical analyses.

Statistical analyses were aimed at answering three main questions. First, was there evidence of trends (continuous change in a single direction between years sampled) in key variables over the entire winter range since the initiation of monitoring? Secondly, how do variables of interest differ among management subgroups (e.g., core winter range vs. noncore winter range, fenced vs. unfenced plots, burned vs. unburned plots)? Lastly, is there evidence of similar trends among management subgroups over time?

Burned plots were included in the analyses when comparing aspen and willow variables on the winter range, because the objective of the EVMP monitoring is to assess condition on aspen and willow communities on the entire winter range. Removal of the burned plots from this analysis has the potential to inflate willow cover values (as areas where willow were burned would no longer be considered representative of the winter range) and decrease aspen regeneration and recruitment values (due to the positive effect that fire has on aspen suckering) resulting in an assessment of conditions on only unburned portions of the winter range. To address the effects of fire, separate models were fit with year (baseline, 2013, 2018), fencing and burning as main factors only and with their interactions.

Bayesian methods were used for statistical inference. Bayesian inference is not based on statistical significance (i.e., with effects tested against “zero”), but offers a probabilistic view of model parameters and their uncertainty (Gelman and Stern 2006; Gelman et al. 2014; Hobbs and Hooten 2015). Under the Bayesian paradigm, it is valid to assign a probability to a model parameter, conditional on the data and model (Bolstad and Curran 2016; Morey et al. 2016).

To provide inferences on differences between year sampled, range type, and fencing/burning subgroups, we fit Bayesian Generalized Linear Models to core winter range, noncore winter range, and Kawuneeche Valley subsets of the data using the R statistical software (R version 4.0.3). Random effects models with plots treated as random factors were developed using the ‘stan_glmmer’ function in the rstanrm package (Brilleman 2018; Goodrich et al. 2020; Stan Development Team 2022). In addition, models were fit to particular subgroups of data. For aspen data, these included short and tall saplings; for willow and upland data, different groups of species (e.g., all willows, all shrubs, etc.) were separately analyzed.

Bayesian estimation was performed via the Hamiltonian Markov Chain Monte Carlo (MCMC) algorithm, a method for numerically sampling from a probability distribution, adding independent

priors specific to the data type being modeled on regression coefficients. Four chains with 8,000 iterations and a burn-in of 1,000 iterations were run and convergence evaluated using graphical methods and MCMC diagnostic statistics. Burn-in refers to discarding samples at the initial start of the MCMC chain and is intended to give the algorithm time to reach its equilibrium distribution. Weakly informed priors as recommended by Lemoine (2019) were used for different response types (e.g., willow height, aspen suckering density).

Continuous response variables (e.g., willow height) were fit with a gamma distribution, which restricts possible responses to positive real numbers. Aspen stem counts were modeled as Poisson processes, while proportion data (e.g., cover) were modeled using a beta distribution (Ferrari and Cribari-Neto 2004), which constrains values from 0 to 1.

Graphical posterior predictive checks created using the ‘bayesplot’ R package (Gabry et al. 2019; Gabry and Mahr 2019) were used to evaluate model fit. Posterior predictive checking helps evaluate model fit by comparing data generated by the model to the observed data. Pairwise comparisons were made between competing models using expected log pointwise predictive density in the ‘loo’ R package (Vehtari et al. 2017; Yao et al. 2018; Vehtari et al. 2019). Density estimations from posterior distributions for model parameters from MCMC draws were used to calculate centrality and uncertainty in posterior estimates.

Functions in the BayestestR package in R were used to describe the posterior distributions of parameters from Bayesian models. We report measures of centrality, uncertainty, effect existence, effect significance, and effect size. The median of the posterior distribution was used as the index of centrality; in probabilistic terms, this means that there is 50% probability that the effect is higher or lower. We used the 95% Highest Density Interval (HDI) credible interval to evaluate uncertainty in parameter estimates. The Probability of Direction (PD), also known as the Maximum Probability of Effect, was used to evaluate effect existence and direction. This index of effect existence varies between 50% and 100% and can be interpreted as the probability that a parameter (described by its posterior distribution) is strictly positive or negative (whichever is the most probable). The PD is correlated with the frequentist p-value, with a two-sided p-value of respectively 0.1, 0.05, and 0.01 approximated by a PD of 95% (0.95), 97.5% (0.975), and 99.5% (0.995) (Makowski, Ben-Shachar, Chen, and Lüdtke 2019; Makowski, Ben-shachar, and Lüdtke 2019).

Assessments of effect significance were based on a threshold of 0.05 of the standard deviation (SD) of the outcome variable and a threshold of $0.3 \times \text{SD}$ of an outcome variable was used to assess whether effect size was large. The effect size (ES) index gives an idea about the strength of an effect however, studies have shown that a large effect size can still be suggestive of low statistical power (Funder and Ozer 2019), particularly when coupled with low probability of direction (PD). Note, both thresholds are inherently subjective (as is the $\alpha=0.05$ typically used in the frequentist null hypothesis testing). However, for our purposes, an ES greater than 0.3 indicates a large effect, while less than 0.3 would indicate a small effect, and less than 0.05 would be a tiny effect. Differences between factors (e.g., year, fenced/unfenced) were compared using functions in the ‘emmeans’ R package (Lenth 2021). More details on statistical models are provided in the appendix.

Results and Discussion

Aspen

Stand Structure

Saplings (0 to 2.5 m height)

Counts of aspen saplings binned by height class varied between years and in relation to fencing treatments, with different patterns across core and noncore range winter range. This is visually apparent in repeated site images (Figures 8 and 9). Sapling recruitment (minimum of 162 aspen stems/acre that were 151–250 cm in height) across the entire winter range increased from 13% at baseline to 39% in 2018 (Table 3). Increases in recruitment were observed across the core and noncore winter range and both inside and outside of elk exclosures. However, the largest increases were observed within fenced plots (Table 3). There was recruitment into greater height classes over time observed in some but not all plots. There was no difference in aspen sapling density in the tallest stem height class (151–250 cm) between core and noncore plots at baseline (Table 4). However, by 2013 and continuing into 2018, the density of the tallest saplings had increased inside elk exclosures in the core winter range. There was a moderate increase in short saplings on unfenced core winter range aspen plots from baseline conditions and later time periods, whereas the density of short saplings decreased over time on fenced core winter range aspen plots possibly due to crowding that occurred as small saplings grew into taller height classes (Table 4).



Figure 8. Comparison of vegetation composition over time at unfenced plot AC22 in west Horseshoe Park. Little to no aspen regeneration at baseline (A, fall 2009), five-yr EVMP sample (B, summer 2013), or ten-yr EVMP sample (C, summer 2018).

Only two fenced plots had a small fraction of saplings in the 151–250 cm height classes at baseline, but substantial recruitment into these size classes was evident in 2013 and 2018 (Figure 10). In contrast, unfenced plots had far less recruitment (Figure 10). The density of tall aspen saplings (151–250 cm in height) on the core winter range increased between baseline and 2013 (Table 4; Figure 10; Zeigenfuss and Johnson 2015).

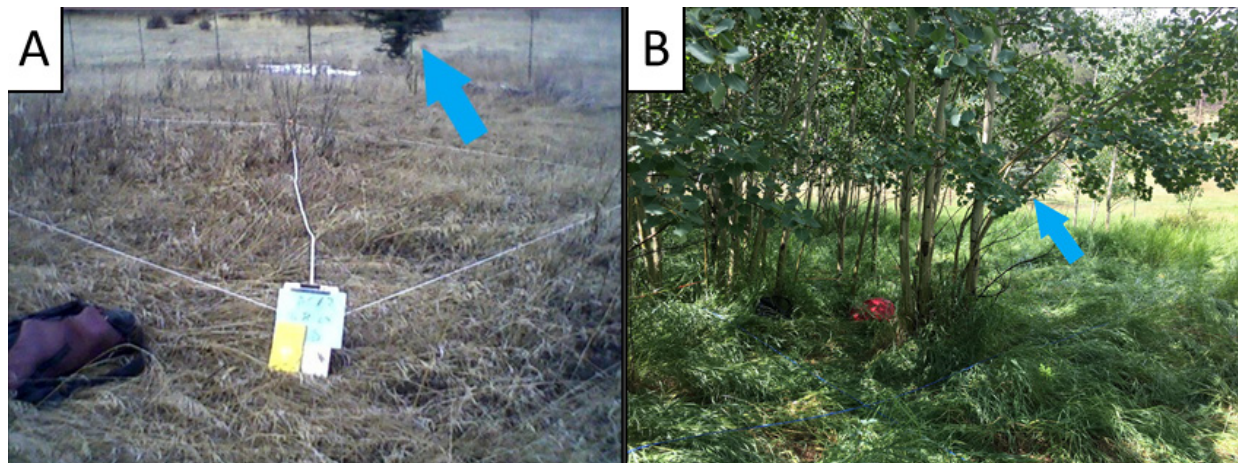


Figure 9. Comparison of vegetation composition over time at plot AC62 inside aspen exclosure HP20. Blue arrow indicates landmark conifer tree growing to the north of the plot outside the fence (behind dense aspen growth in photo B). Baseline composition two months after fence construction (A, fall 2009); Aspen regeneration after nine years of exclusion from ungulate browse (B, summer 2018).

Table 3. Percentage of plots in core unfenced, core fenced, and noncore elk winter range that meet the criteria for desired future condition for regeneration (saplings 151–250 cm height at density of at least 400 per hectare [162/acre]). Entire winter range estimate is weighted to account for the proportion of total winter range aspen area within each range type. Baseline sampling occurred from 2006 to 2008 on the winter range.

Range type	Sample date		
	Baseline	2013	2018
Entire winter range	13%	29%	39%
Core winter range total	7%	22%	31%
Core, fenced	–	64%	93%
Core, unfenced	–	12%	19%
Noncore winter range total	15%	28%	43%

Table 4. Mean (\pm standard error) aspen sapling density (stems/acre) estimates and credible intervals for short saplings (plants with height ≤ 150 cm) and tall saplings (plants with height from 151–250 cm) at core and noncore winter range aspen monitoring plots in Rocky Mountain National Park, Colorado. Includes plots burned by the Fern Lake Fire in 2012.

Range type	Short (stems ≤ 150 cm height)			Tall (stems 151–250 cm height)		
	Baseline (mean \pm SE)	2013 (mean \pm SE)	2018 (mean \pm SE)	Baseline (mean \pm SE)	2013 (mean \pm SE)	2018 (mean \pm SE)
Core range, unfenced	3,326 \pm 537	8,314 \pm 2,682	5,190 \pm 933	5 \pm 5	29 \pm 13	572 \pm 380
Core range, fenced	6,799 \pm 2,147	4,775 \pm 1,154	1,457 \pm 301	27 \pm 18	1,827 \pm 690	1,492 \pm 443
Noncore range	1,934 \pm 447	2,328 \pm 434	3,407 \pm 523	138 \pm 106	161 \pm 83	308 \pm 100

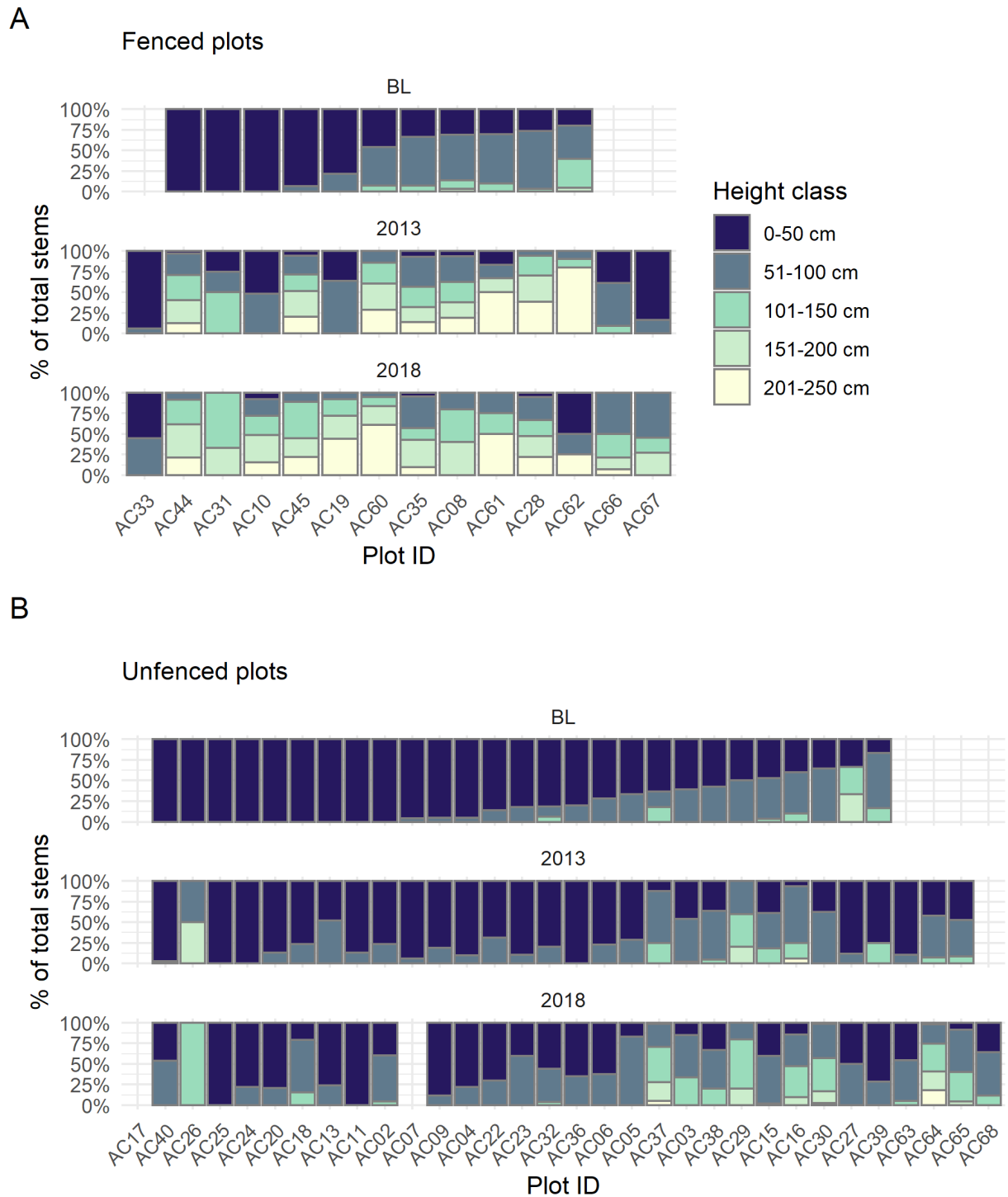


Figure 10. Percent of total aspen sapling stems in different height classes for (A) fenced and (B) unfenced core elk winter range in Rocky Mountain National Park. BL= baseline data (collected 2006–2008).

The density of stems of either short or tall saplings since baseline sampling increased, but only slightly, in noncore winter range plots (Figure 11, Table 4). Noncore winter range plots had the

highest densities of tall saplings at baseline, but fenced core winter range plots had greater density of tall saplings than noncore winter range plots in 2013 and 2018 (Table 4). The density of short saplings (≤ 150 cm in height) changed little between baseline conditions, 2013, and 2018 for noncore winter range plots.

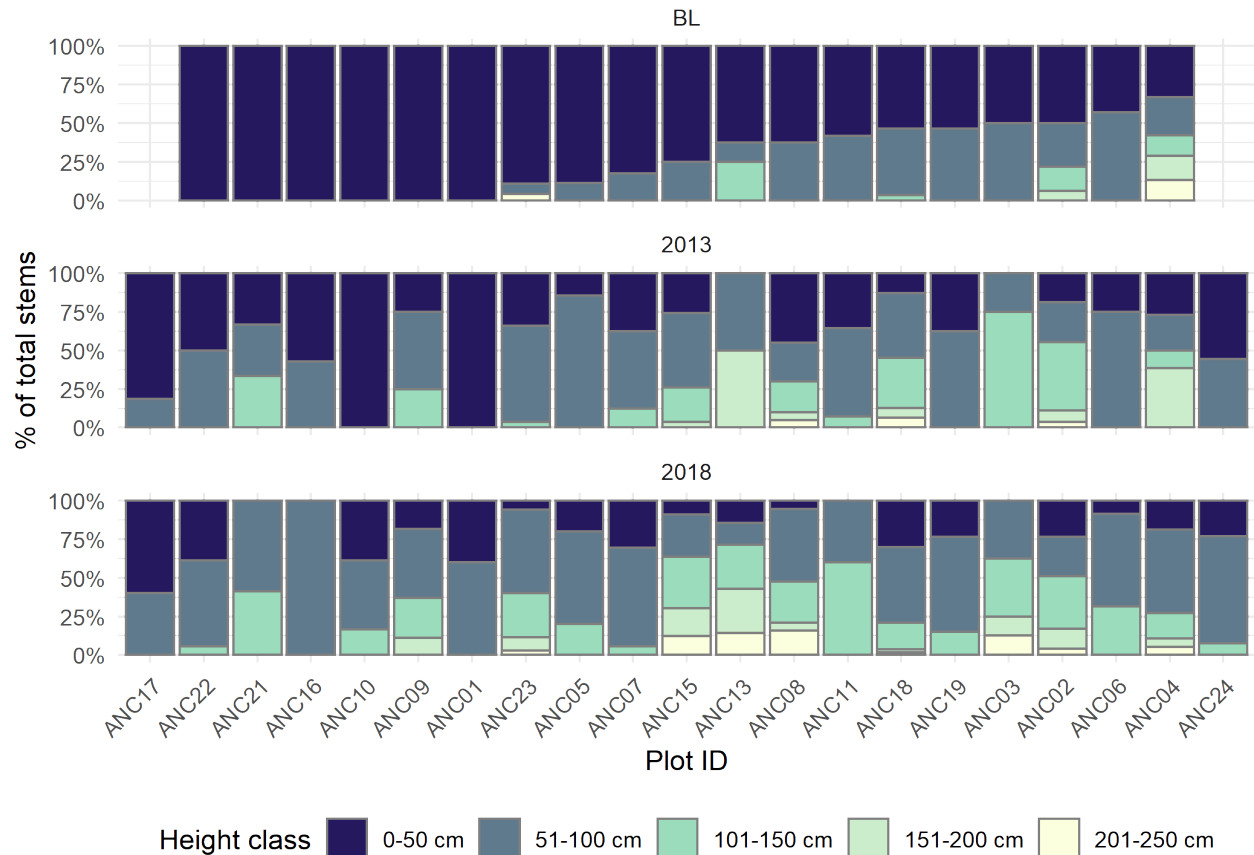


Figure 11. Percent of total aspen sapling stems in different height classes for noncore elk winter range in Rocky Mountain National Park. All noncore aspen are unfenced. BL= baseline data (collected 2006–2008).

Models for trees, short saplings (≤ 150 cm tall), and tall saplings (151–250 cm) that included main effects (fencing and year) and their interactions performed better than those that included only main effects. The effect of time-since-baseline was conditioned on whether areas were fenced, but as detailed below, the specific patterns differed for trees, short, and tall sapling classes and among sites. Eleven of 14 (78%) fenced core winter range plots had saplings 151–250 cm in 2018 and 7 of those plots had high densities of tall saplings indicating high regeneration and likelihood of recruitment to tree-sized stems. In contrast, only 5 of 32 (16%) unfenced core plots had tall saplings 151–250 cm by 2018 and only 4 of them had high sapling densities indicating potential for recruitment to tree classes. Nine of 21 (43%) noncore winter range plots had some saplings in the 151–250 cm height classes by 2018

Figure 12 shows the modeled mean stem counts in a 25m² plot for both short and tall aspen saplings in the combined core and noncore elk winter range. The changes over time were evaluated on the Probability of Direction (PD) which indicates how likely a change in one direction occurred and whether the change was small or large. Median posterior estimates and credible intervals for regression coefficients indicated a strong negative effect of year, with the largest effects seen in fenced plots. For short saplings in fenced plots (Figure 12A), the modeled mean count declined from a high of 12 stems/25m² plot at baseline dropping to half that number by 2013 (PD > 0.99) and falling to 2.2 stems/25m² plot in 2018 (PD > 0.99). This effect was large for fenced plots both years (ES = 1.0). For tall saplings in fenced plots (Figure 12B), a positive effect of fencing was observed with modeled stem counts increasing from 0 at baseline to over 2 stems/plot (a stem density of 324 stems/acre) in 2013 (PD > 0.99) and 2018 (PD > 0.99). The effect was larger in 2013 (ES = 1.0) than 2018 (ES = 0.62). The wide credible intervals indicate high uncertainty in predictions.

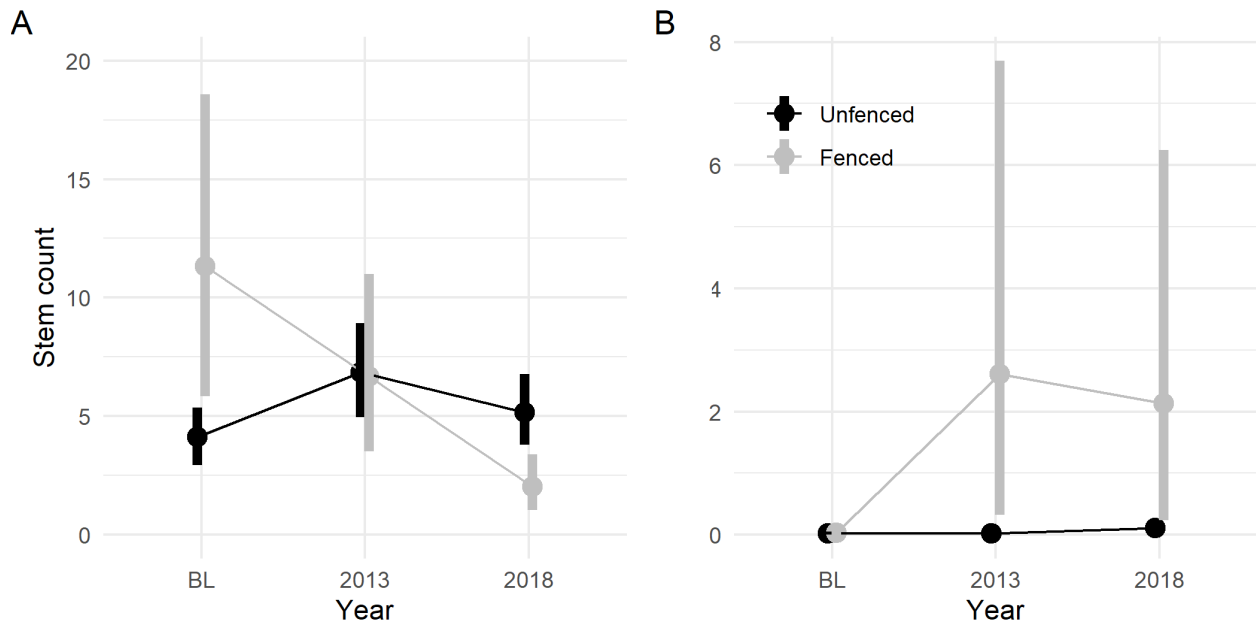


Figure 12. Modeled mean stem counts (points) and credible intervals (bars) for fenced and unfenced (A) short (≤ 150 cm height) and (B) tall (151–250 cm height) aspen saplings on combined (core and noncore) elk winter range in Rocky Mountain National Park. BL= baseline data (collected 2006–2008).

We observed similar temporal trends when sapling stem counts were modeled separately for core winter range (Figure 13) to those modeled using combined core and noncore winter range plots (Figure 12). Estimated short sapling stem counts in fenced plots declined from a high of 12 stems/plot at baseline to 3 stems/plot in 2018 (PD > 0.99 for both 2013 and 2018) indicating a large (ES = 1.0), negative effect of fencing, likely due to crowding as stems moved into the taller sapling class in fenced plots. Unfenced short sapling counts increased slightly from 5 stems at baseline to ~8 stems in 2013, declining slightly to just over 5 stems in 2018, although overlap in credible intervals between these time periods indicate weak evidence of trends (Figure 13A). The effect of fencing on tall (151–250 cm) sapling stem counts was positive for both 2013 (PD > 0.99) and 2018 (PD > 0.99) but the effect size was lower in 2018 (ES = 0.62) compared to 2013 (ES = 1.0). Modeled mean stem

counts/plot were near 0 for all time periods in unfenced plots but increased from baseline for 2013 and 2018 in fenced plots, although again with wide and overlapping credible intervals (Figure 13B).

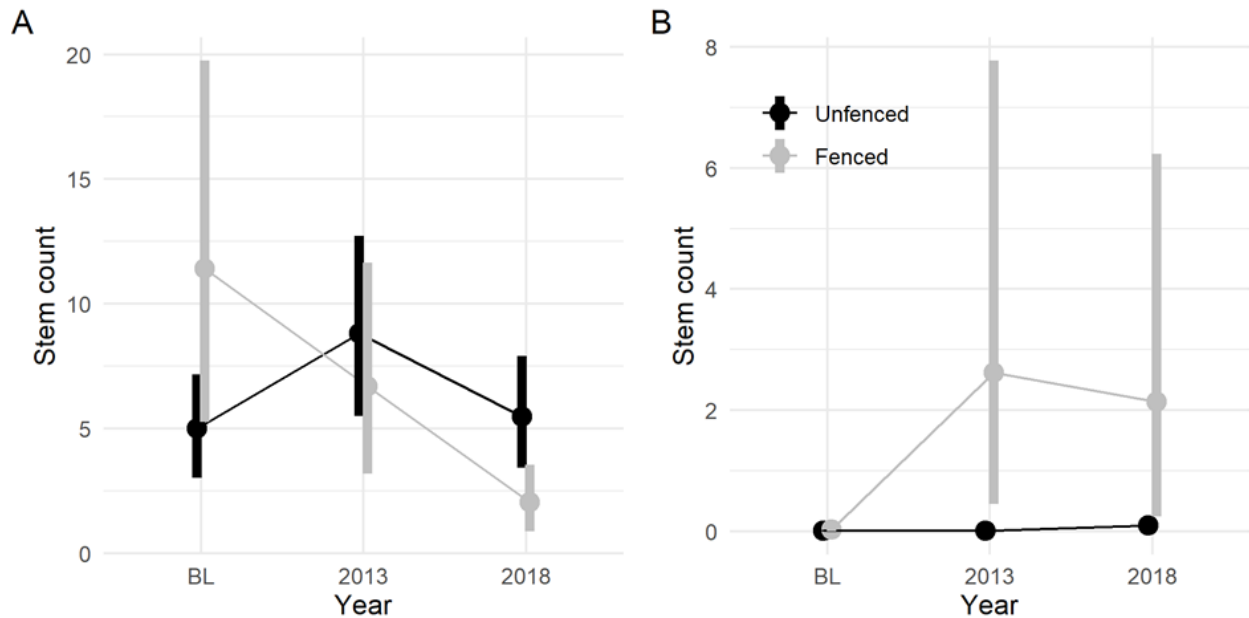


Figure 13. Modeled mean stem counts (points) and credible intervals (bars) for fenced and unfenced (A) short (≤ 150 cm height) and (B) tall (151–250 cm height) aspen saplings on core elk winter range in Rocky Mountain National Park. BL= baseline data (collected 2006–2008).

When we analyzed noncore winter range aspen plots alone, there was high probability of positive increases in small sapling stem counts in 2013 (PD = 0.99) and 2018 (PD > 0.99). The effect size was small in 2013 (ES = 0.14), but large in 2018 (ES = 1.0). Modeled mean stem counts for short saplings in noncore elk winter range plots (all unfenced) increased from 3 stems at baseline to 5.5 stems in 2018 (Figure 14A). The number of tall saplings in noncore plots did not change by 2013 (PD = 0.73, ES = 0.39), but had increased by 2018 (PD > 0.99, ES = 1.0). However, mean stem counts were low (<1 stem/plot) in all years (Figure 14B).

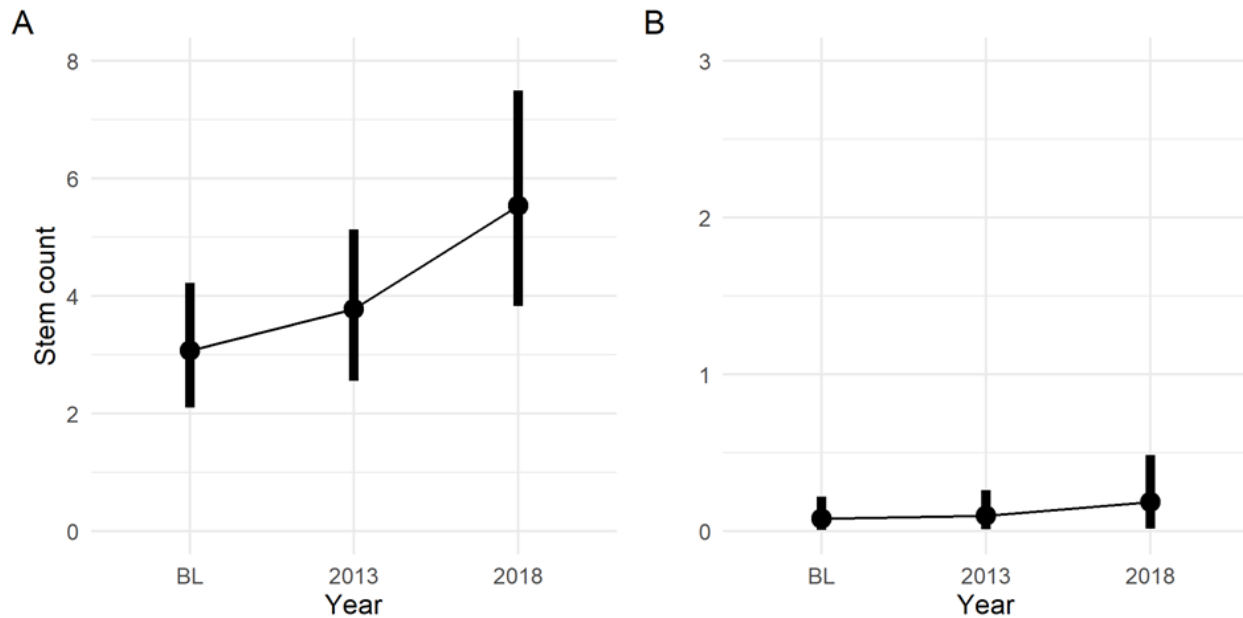


Figure 14. Modeled mean stem counts (points) and credible intervals (bars) for (A) short (≤ 150 cm height) and (B) tall (151–250 cm height) aspen saplings on noncore elk winter range in Rocky Mountain National Park. All noncore plots were unfenced. BL= baseline data (collected 2006–2008).

Trees (> 2.5 m; 0 to 20 cm DBH)

Aspen stands in unfenced core winter range areas maintained a declining size structure across time. Stems transitioned into larger size classes with no recruitment of smaller or medium-diameter stems at many plots. The percentage of trees in small DBH classes (< 10 cm) increased little over time in unfenced core winter range plots (14% of plots at baseline to 15.6% of plots in 2018), but there was a progressive increase in small-diameter stems in the fenced core plots from 17% of plots at baseline to 86% of plots by 2018 (Figure 15). Three fenced plots that had no tree-sized stems at baseline had stems in the 2–10 cm DBH range by 2018, whereas none of the unfenced plots where tree-sized stems were absent at baseline had recruited stems by 2018 (Figure 15). Two plots where all remaining tree-sized stems were dead following the 2012 wildfire experienced recruitment of small diameter stems by 2018 (sites AC37 and AC64 in Figure 15).

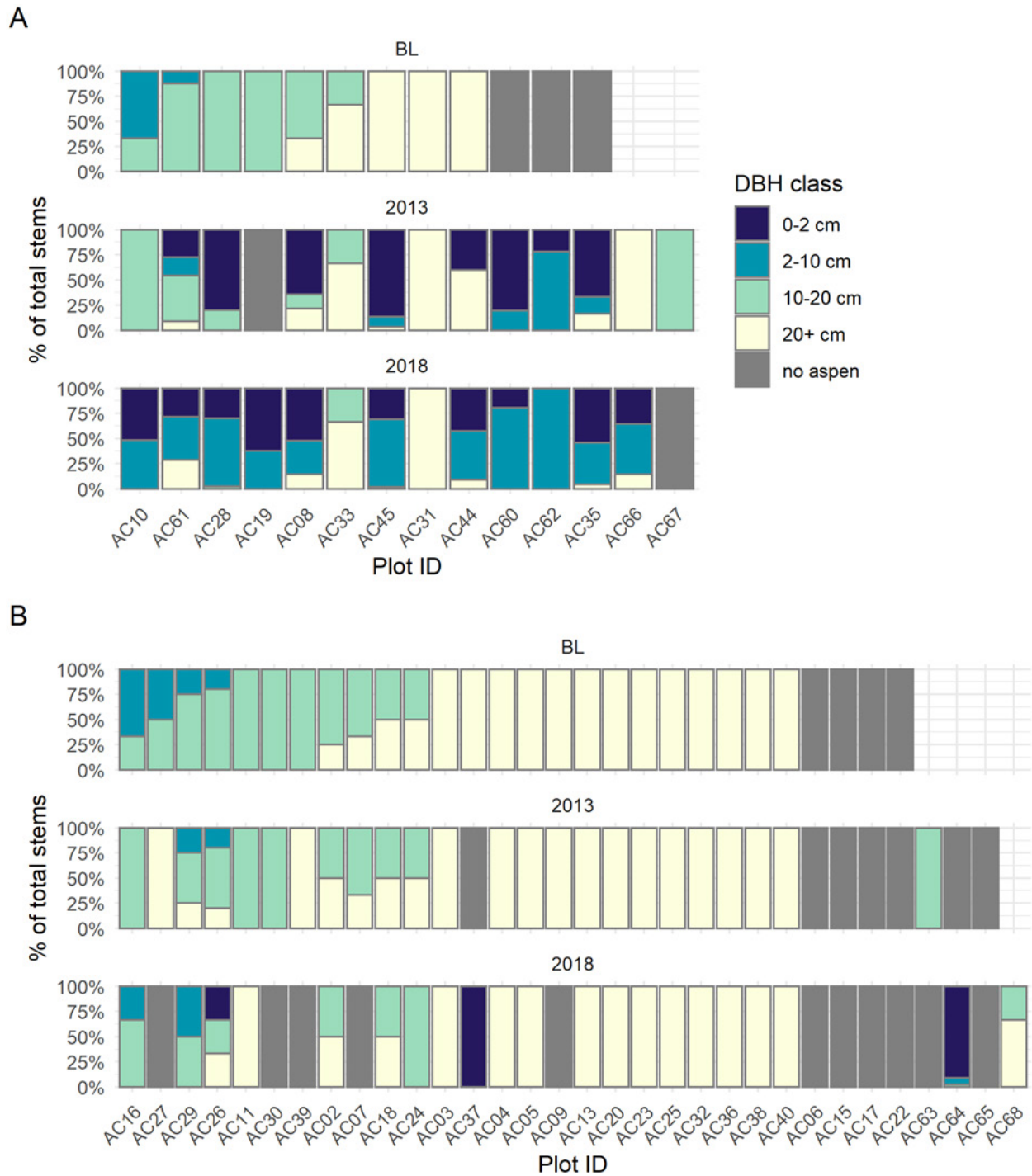


Figure 15. Stand structure of (A) fenced and (B) unfenced aspen trees on the core elk winter range based on diameter at breast height (DBH) size classes. BL= baseline data (collected 2006–2008). Sites AC37 and AC64 were burned in the 2012 Fern Lake Fire.

Stand structure changed very little on noncore winter range plots. Small-diameter stems increased in noncore winter range over time from 15% of plots with stems <10 cm DBH at baseline to 33% of

plots by 2018 (Figure 16). However, over half of the noncore plots had only trees of 20 cm DBH or greater or no tree-sized stems at all by 2018 (Figure 16).

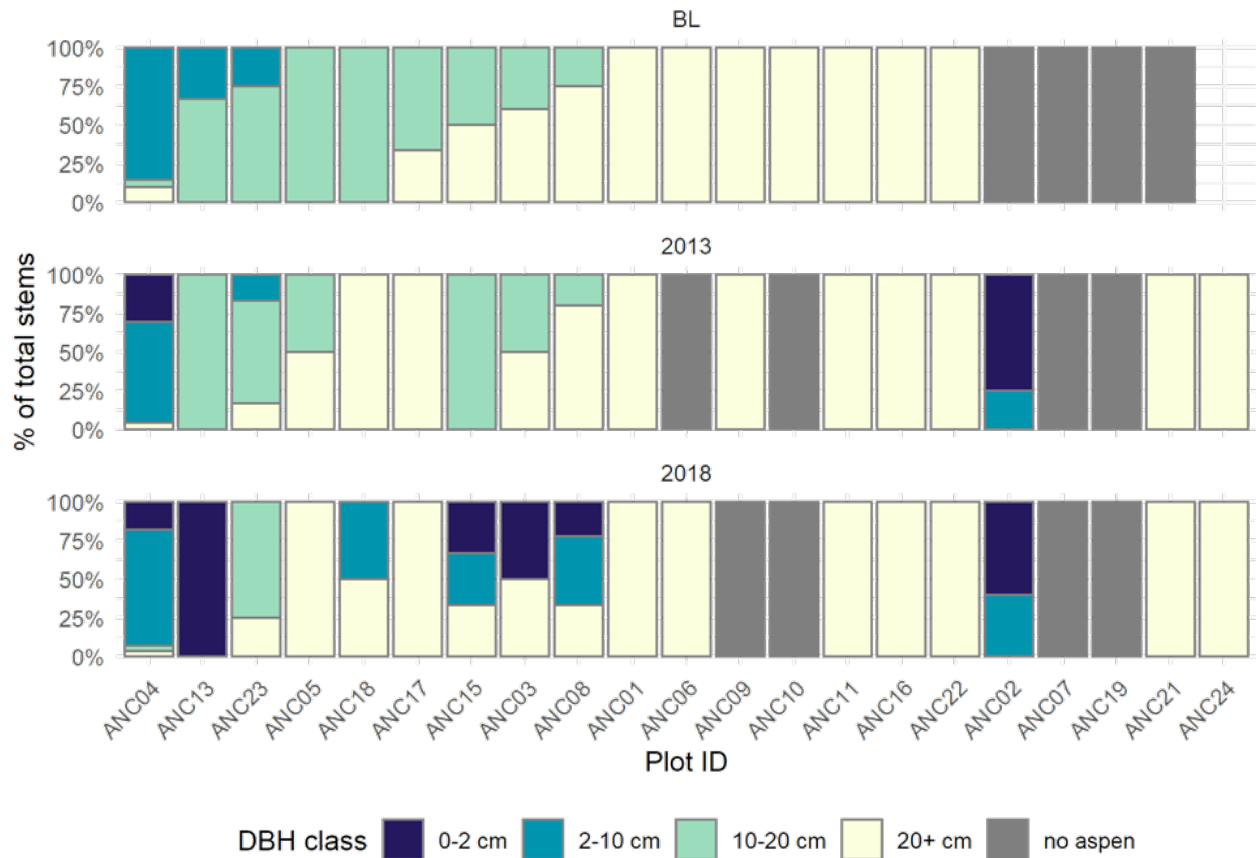


Figure 16. Stand structure of aspen trees on the noncore elk winter range based on diameter at breast height (DBH) size classes. BL= baseline data (collected 2006–2008). All noncore aspen are unfenced.

The EVMP identified a progressive increase in aspen stand structure above the baseline level as a management objective for aspen. The percentage of plots in the desired future condition of 75% small diameter (<10 cm) aspen trees rose from 0% in the core winter range, increasing to 18% of plots in 2018 (Table 5). The majority of this increase was inside the elk exclosures where 79% of plots had small diameter aspen trees by 2018 (Table 5).

Table 5. Percentage of elk winter range with plots meeting the desired future condition of at least 75% small diameter (<10 cm dbh) trees. It should be noted that only one plot had distribution of medium (10–20 cm dbh) and large (>20 cm dbh) diameter trees similar to the desired condition (20% medium diameter and 5% large diameter). Baseline sampling occurred from 2006 to 2009 on the winter range.

Location	Sample date		
	Baseline	2013	2018
Core, fenced	0%	36%	79%
Core, unfenced	0%	0%	6%
Noncore winter range total	5%	10%	14%

Effects of Wildfire

We include analysis on the effects of wildfire in aspen to determine whether fire might be used as an effective tool in reaching EVMP aspen recruitment goals. The effect of wildfire on aspen differed among plots, causing mortality of saplings stems in some plots but having a positive effect on sapling regeneration evident in 2013 (Figure 17). Overall, burning and fencing had a large positive effect on short aspen sapling stem counts in both 2013 (PD = 0.98, ES = 0.94) and 2018 (PD > 0.99, ES = 1.0). Modeled stem counts for short saplings in burned plots increased from baseline to 2013, before declining in 2018 for both fenced and unfenced plots (Figure 18A). Overall modeled stem counts/plot were lower in unburned plots, but patterns differed depending on whether plots were fenced or unfenced; fenced, unburned plots had slightly declining stem counts over time and unfenced, unburned plots showed a slight increasing trend. These patterns were different for tall saplings. Models of fencing and burning for tall saplings had low probability of either positive or negative effect of fencing and burning in either 2013 (PD = 0.79) or 2018 (PD = 0.94). Modeled mean stem counts/plot for tall aspen saplings (Figure 18B) were near 0 in both unfenced and burned plots at all time periods, while counts slightly increased in fenced plots regardless of burning. This was potentially due death/removal of stems by burning in 2013, then necessary time for sapling to reestablish and grow into taller size classes. Measurements in 2023 may indicate a trend that was not evident by 2018.

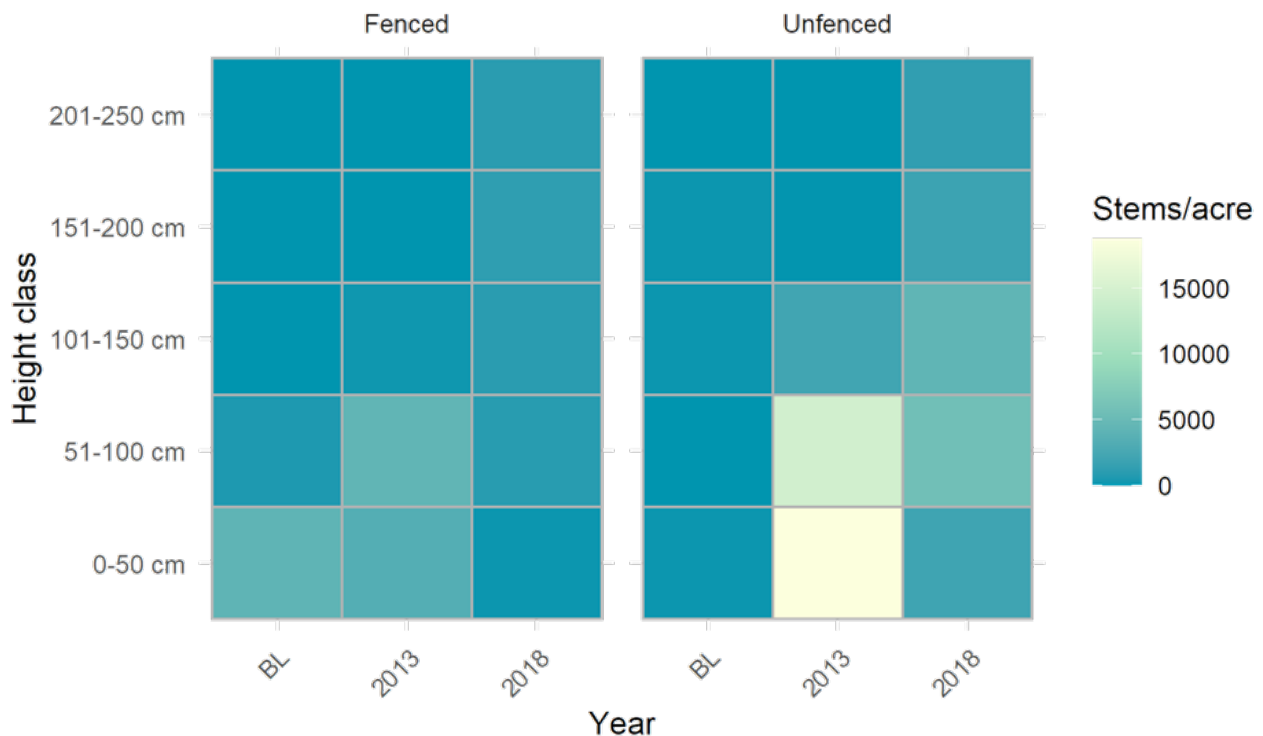


Figure 17. Mean live aspen sapling density (stems/acre) in burned plots in different height classes in fenced and unfenced core elk winter range plots of Rocky Mountain National Park. BL = Baseline data (collected 2006–2008).

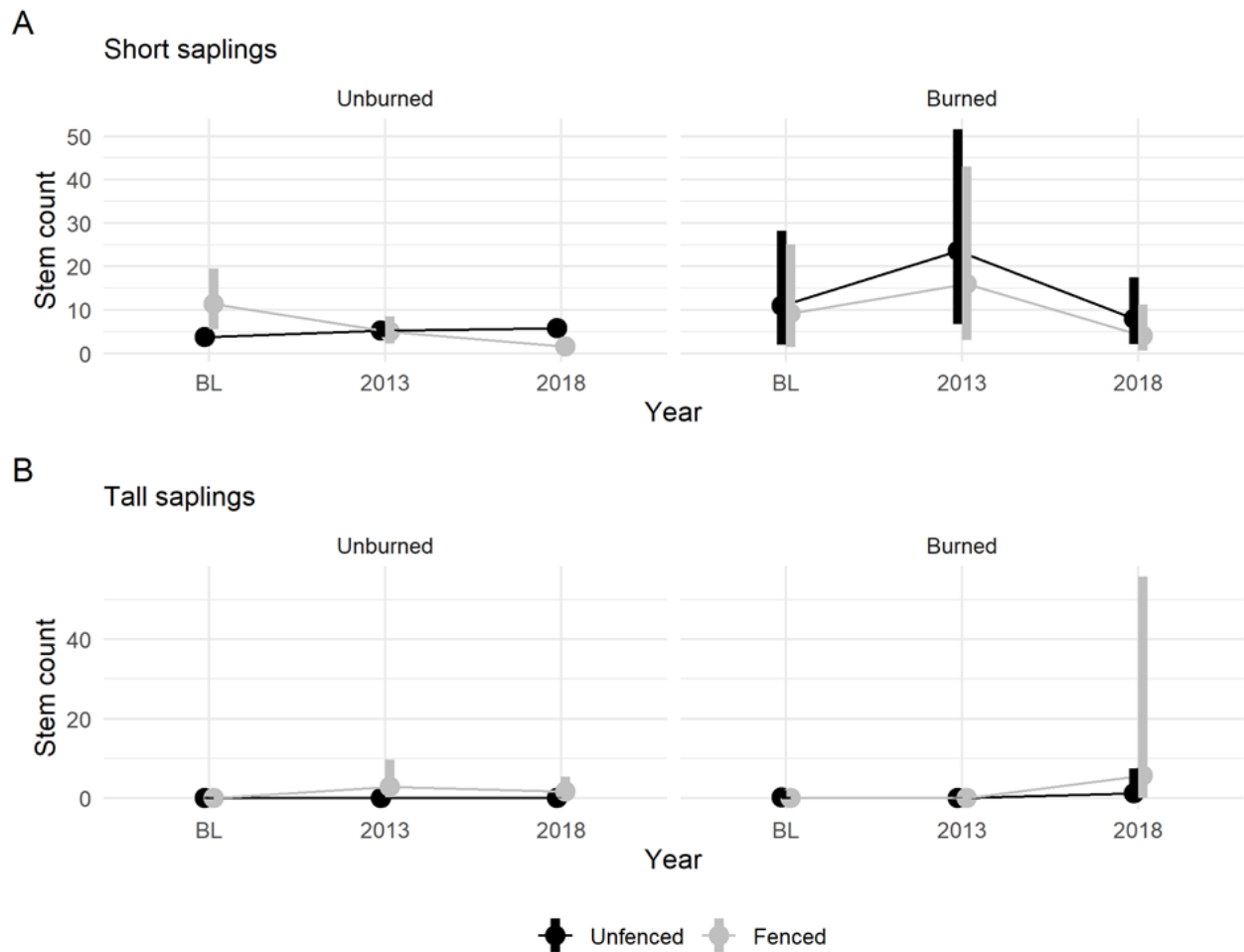


Figure 18. Modeled mean aspen stem counts (points) and credible intervals (bars) for burning and fencing treatments on combined (core and noncore) elk winter range in Rocky Mountain National Park for (A) short (≤ 150 cm height) and (B) tall (151–250 cm height) saplings. Sites were burned in 2012. BL= baseline data (collected 2006–2008).

The mean count of dead aspen trees in plots burned in 2012 showed little change between years (Figure 19), however, live tree tallies in smaller diameter classes showed a positive trend since baseline (Figure 19). This pattern was most prominent in fenced plots. This increase in small diameter trees (0–2 cm DBH, > 250 cm height) when tall saplings (151–250 cm) did not show much increase may reflect that some stems regenerated quickly enough to recruit into the tree classes between 2013 and 2018.



Figure 19. Mean tree count by DBH (diameter at breast height) class of live and dead aspen trees on fenced and unfenced burned sites on elk winter range aspen sites in Rocky Mountain National Park over time (baseline, 2013, and 2015 through 2018). Sites were burned in December 2012. BL = baseline data (collected 2006–2008).

Kawuneeche Valley

Stand structure in Kawuneeche Valley aspen plots was stagnant across the study period (Figure 20–22) and characterized by large-diameter trees and little regeneration or recruitment of aspen saplings. Statistical comparisons indicated little support for a positive or negative effect of year for total stem counts (Figure 22). No plot in the Kawuneeche Valley had any saplings taller than 100 cm (Figure 20). Half of the Kawuneeche Valley plots had an increase in the number of saplings in the 0 to 50 cm class, with mean stem counts/plot unchanged between baseline and 2013 but increasing in 2018. The percentage of live aspen stems in different DBH classes differed among individual Kawuneeche Valley plots (Figure 21). All stems in 3 plots were in the 20+ cm DBH class from baseline to 2018 and all plots but one had an increasing percentage of stems in this largest size class (Figure 21). Only a single plot had any small diameter stems at baseline, however, by 2018, no small diameter trees were found in any of the Kawuneeche Valley plots (Figure 21).

Modeled short (≤ 150 cm) aspen sapling stem counts in Kawuneeche Valley plots showed a weak increase over time, from a low of 3 stems at baseline to just over 4 stems in 2018 (PD > 0.99, ES = 0.71), and large overlapping credible intervals indicate weak support for differences between years (Figure 22A). In contrast, modeled counts for tall saplings were 0 for all time periods (Figure 22B).

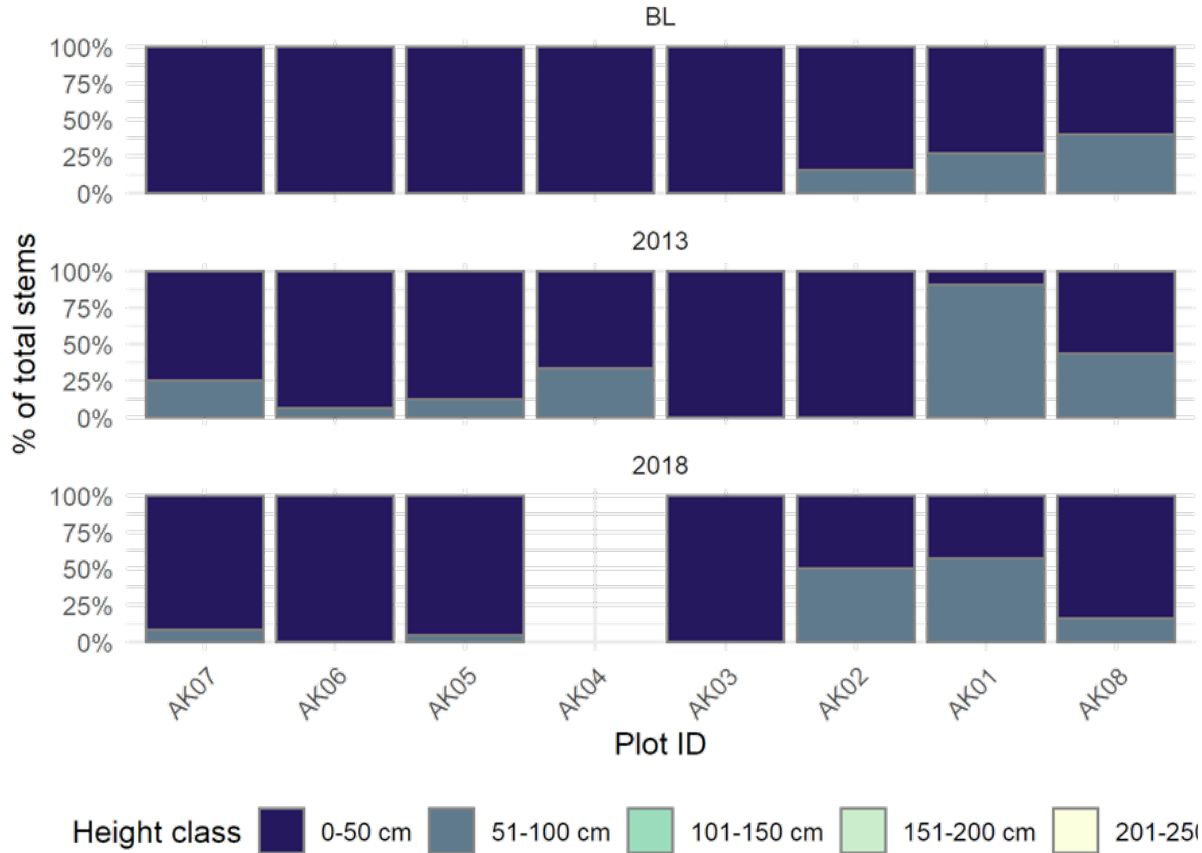


Figure 20. Percent of total aspen sapling stems in different height classes in the Kawuneeche Valley of Rocky Mountain National Park. All Kawuneeche Valley aspen are unfenced. BL = baseline data (collected 2006–2008).



Figure 21. Stand structure of aspen trees in individual Kawuneeche Valley aspen plots based on diameter at breast height (DBH) size classes. All Kawuneeche Valley aspen are unfenced. BL= baseline data (collected 2006–2008).

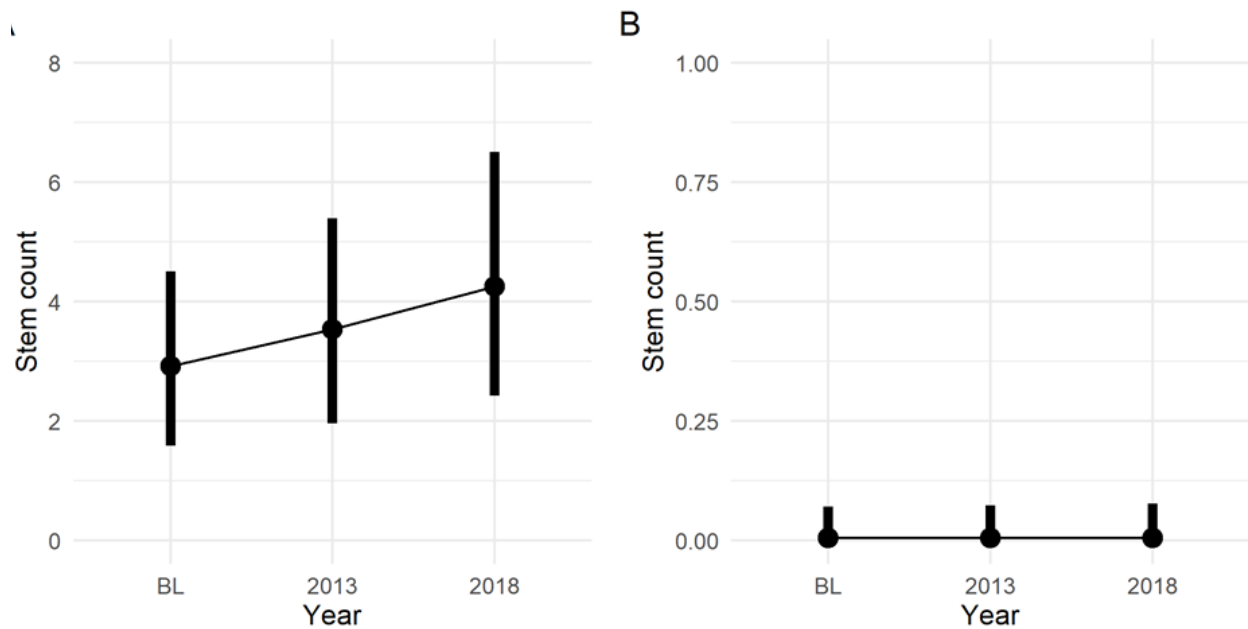


Figure 22. Modeled mean stem count (points) and credible intervals (bars) for (A) short (≤ 150 cm height) and (B) tall (151–250 cm height) aspen saplings in the Kawuneeche Valley of Rocky Mountain National Park. BL= baseline data (collected 2006–2008).

Progress Toward EVMP Aspen Goals

The EVMP management objectives for aspen communities focused on regeneration and stem recruitment, assessed using measurements of stem count (and subsequently, stem density), diameter, and height (NPS 2007; Zeigenfuss and Johnson 2015). A key management goal for aspen was a progressive increase in regeneration and a shift in stem size distribution for tree-sized stems (Table 1). At least 45 percent of stands should develop a regeneration cohort within each decade, evidenced by a representation of stems in different age and size classes. In 2018, 39% of aspen stands on the winter range showed regeneration (Table 3) and there has been an increase in stands with >75% small diameter stems from 4% at baseline to 15% in 2018 (Table 5). These analyses demonstrate a shift toward the desired future condition primarily reflecting recruitment of younger and smaller-diameter trees inside elk exclosures and limited but poorer progress outside of elk exclosures on both core and noncore ranges indicating a stagnant or declining stand structure. Exclosures continue to have a large effect on aspen recruitment in the core elk winter range consistent with past studies on the effects of fencing in RMNP and elsewhere (Kay and Bartos 2000; Kay 2001; Kaye et al. 2005; Rogers et al. 2015).

While aspen forests occur in a wide variety of settings across North America (Rogers et al. 2014), their distribution in RMNP is limited (Kaye et al. 2003; Binkley 2008; Zeigenfuss et al. 2008). Aspen can persist for centuries through asexual reproduction, but older and larger diameter stems eventually die, and without adequate recruitment, the long-term persistence of clones can be threatened (Weisberg and Coughenour 2003; Kulakowski et al. 2004). Many EVMP plots had stems transition into larger size classes with no recruitment of smaller or medium-diameter stems, which suggests a lack of regeneration. Absent a change in trajectory, this may threaten the persistence of aspen stands as older stems eventually die. While there was progress toward the aspen vegetation goals, it is primarily in fenced areas on the winter range. EVMP data indicate ungulate herbivory is still an important constraint on recruitment in many unfenced areas. The strong effect of fencing in aspen models highlight the value of this management tool.

Evaluating fire effects on aspen and riparian vegetation was not part of the EVMP objectives, but the 2012 Fern Lake fire that burned through portions of Moraine Park affected multiple plots. These plots reflect conditions in core parts of the elk winter range so were retained in these analyses but were also analyzed separately to help understand the potential responses of aspen and willow communities to wildfire. While fire can cause stem mortality, it can also stimulate production of suckers (Bartos and Mueggler 1981; Mueggler 1989; Bartos et al. 1994). EVMP data from burned plots showed an increase in the number of dead aspen stems following the Fern Lake fire. However, by 2018, burned plots protected from browsing had higher stem counts in a variety of tree diameter and sapling height classes. Burned aspen plots exhibited higher rates of suckering, but suckers are an attractive food source for large ungulates and recruitment of stems in areas where ungulate densities remain high is likely to be restricted to exclosures (Romme et al. 1995; Durham and Marlow 2010). The results support the potential use of wildland and prescribed fire for inducing asexual reproduction (Shepperd et al. 2001; Kilpatrick et al. 2002; Krasnow et al. 2012), and if such aspen regeneration is protected, they can serve as cohorts for future recruitment (Baker et al. 1997; White et al. 1998).

Willow

Willow Height

Modeled mean willow height increased across the winter range for all willow species combined (including *S. bebbiana*, *S. brachycarpa*, *S. drummondiana*, *S. geeyeriana*, *S. eriocephala*, *S. monticola*, *S. petiolaris*, *S. planifolia*), but increases were greater in fenced plots (Figures 23–26, Table 6). Modeled mean willow height increased 35.8 cm between baseline and 2018 when data were combined across core and noncore winter range plots, with high probabilities of a positive effect of fencing in both 2013 (PD > 0.99) and 2018 (PD > 0.99), though the effect was smaller in 2013 (ES = 0.74) than 2018 (ES = 1.0; Figure 27). The pattern was similar whether examining maximum individual plant height or average individual plant canopy height, therefore we focused the remainder of our reporting on average canopy height. Modeled mean willow height was lower at baseline in fenced plots (60 cm) than unfenced ones (100 cm) but increased to over 140 cm in 2018. In contrast, unfenced mean willow heights increased from 100 cm to only 120 cm over the same time period.



Figure 23. Comparison of vegetation composition over time inside willow enclosure HP50 at plot WC16. Willow composition at baseline (A, fall 2008), five years after fence construction (B, summer 2013), and willow regeneration after ten years of exclusion from ungulate browsing (C, summer 2018). Note person holding placard is barely visible standing in willow in photo C whereas overall willow height was about knee-high in photo B. Baseline photos during summer not available.

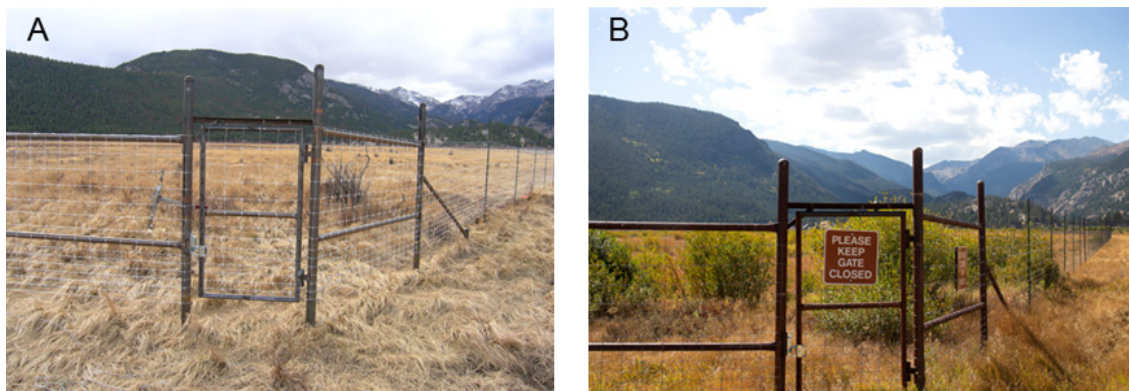


Figure 24. Comparison of vegetation composition over time inside willow enclosure MP50. Baseline condition immediately after fence construction (A, fall 2008) and willow regeneration after ten years of exclusion from ungulate browsing (B, fall 2018).



Figure 25. Comparison of vegetation composition inside willow exclosure MP50 at plot WC28 over time. Blue arrow indicates landmark mountain to the north of the plot. Willow composition at baseline (A, fall 2008), five years after fence construction in 2008 (B, summer 2013), and willow regeneration after ten years of exclusion from ungulate browsing (C, summer 2018).

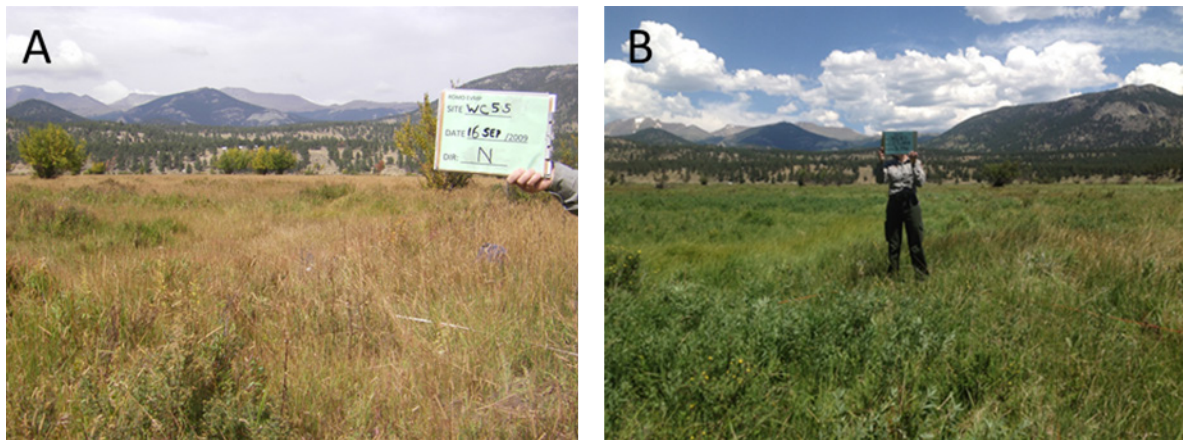


Figure 26. Comparison of vegetation composition over time at unfenced plot WC55 in Moraine Park. Little to no willow regeneration at baseline (A, fall 2009) and at ten-yr EVMP sample (B, summer 2018). Note that vegetation growth at the bottom left of the photo is not willow.

Table 6. Modeled mean willow canopy height (cm) and credible intervals from baseline (2006–2009) to 2018 on core and noncore elk winter range in Rocky Mountain National Park, CO. Analysis includes plots that were burned by the Fan Lake Fire in 2012.

Range type	Baseline mean (CI)	2013 mean (CI)	2018 mean (CI)
Core winter range, unfenced	84.7 (69.9,100.6)	85.2 (70.7,100.9)	94.1 (77.9,111)
Core winter range, fenced	56.9 (44.9,69)	91.2 (73,111)	144.8 (115.4,175.3)
Noncore winter range	129.6 (102.6,157.3)	161.4 (127.6,196.5)	171.2 (135.2,207.3)

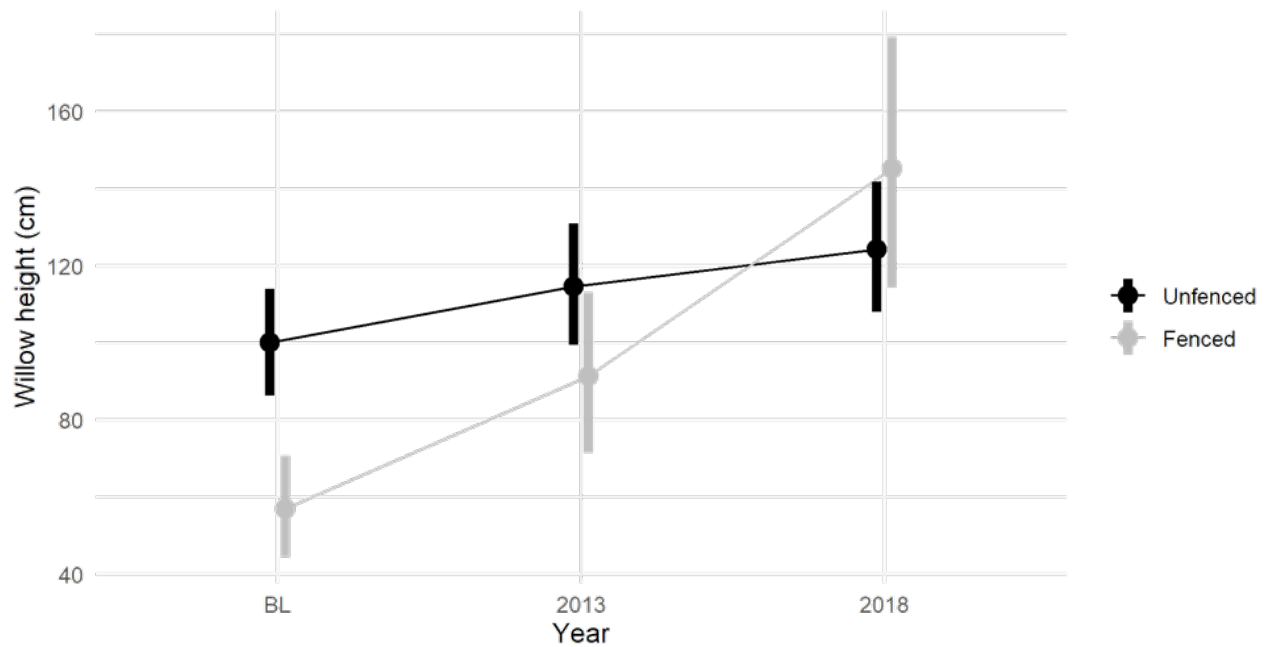


Figure 27. Modeled mean canopy height in centimeters (points) and credible intervals (bars) for all willow species on the combined elk winter range (core and noncore) of Rocky Mountain National Park. BL = baseline data (collected 2006–2008).

Modeled estimates of willow height for core winter range plots alone showed similar patterns as those from the combined core and noncore plots (Figure 28), strongly increasing through time from ~50 cm at baseline to over 140 cm in 2018 for fenced plots (2013: PD > 0.99; 2018: PD > 0.99) and this effect was large (ES = 1.0). In unfenced plots, estimates showed a modest increase from baseline height, but widely overlapping credible intervals provide weak evidence of any differences across time. At noncore elk winter range plots (all unfenced), modeled willow height increased through time (2013: PD > 0.99; 2018: PD > 0.99 ; Figure 29), but the effect size was small (2013 ES = 0.02 and 2018 ES = 0.25). At baseline, estimated height was 130 cm, increasing to 161 and 171 cm in 2013 and 2018, respectively. Credible intervals were wider than for core winter range plots, likely due to smaller sample sizes in the noncore range and also had high overlap.

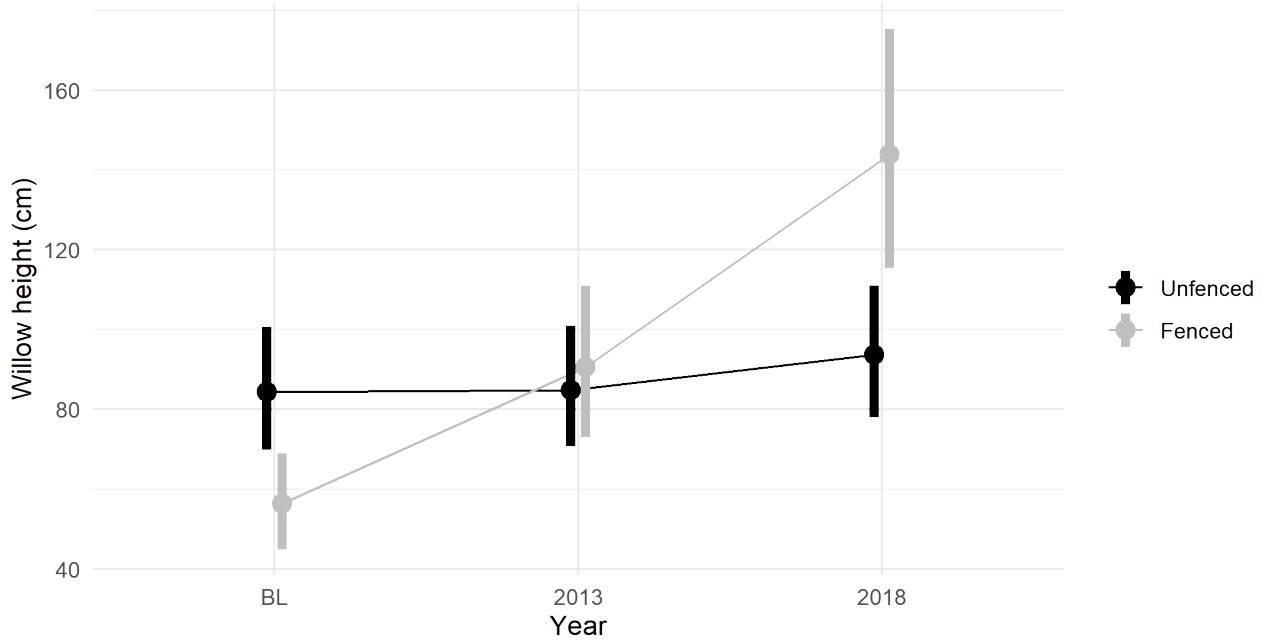


Figure 28. Modeled mean canopy height in centimeters (points) and credible intervals (bars) for fenced and unfenced willow on the core winter range of Rocky Mountain National Park. BL= baseline data (collected 2006–2008).

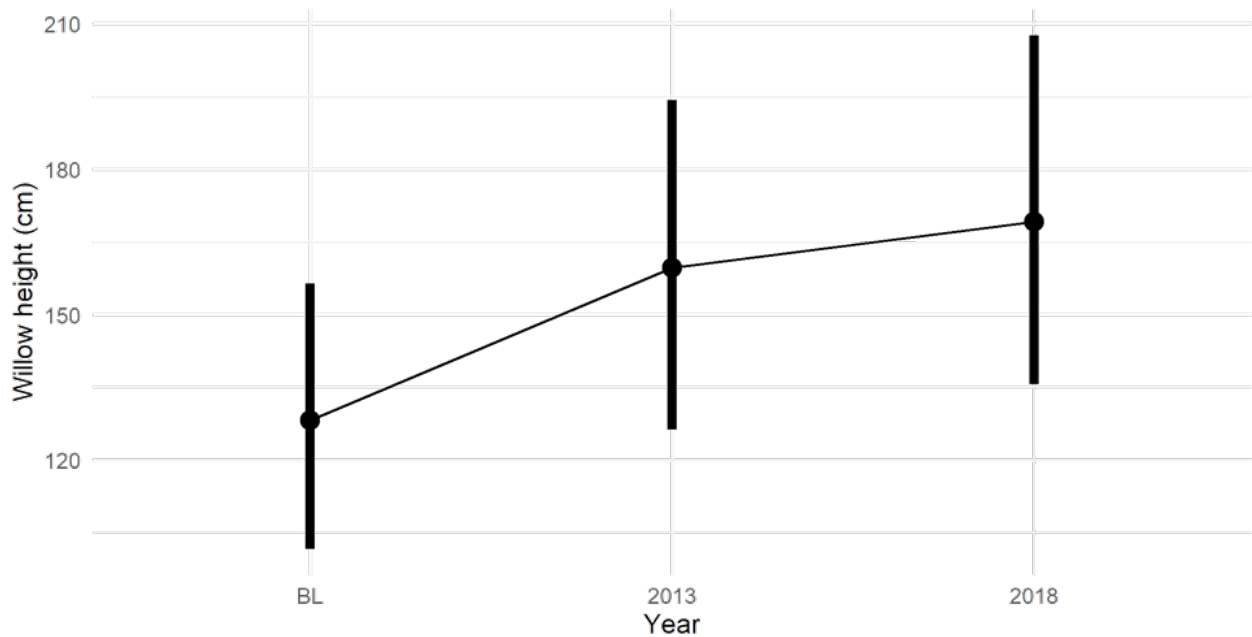


Figure 29. Modeled mean canopy height in centimeters (points) and credible intervals (bars) of willow on the noncore elk winter range of Rocky Mountain National Park. All noncore willow plots are unfenced. BL= baseline data (collected 2006–2008).

Height trends inside and outside of fenced areas were similar whether line intercept or macroplot data were evaluated (line intercept data are provided in supplemental materials). Modeled mean willow height (all willow species combined) increased 88 cm in fenced core winter range plots (Table 6). In

contrast, height increased only 9 cm in unfenced core winter range plots over the same period. In noncore unfenced willow plots, mean height increased 41.6 cm from baseline to 2018 (Table 6).

Species composition was variable across plots in the core and noncore winter range and most species occurred in only a small fraction of plots. The three most abundant willows by frequency across plots were *Salix planifolia*, *S. monticola*, and *S. geyeriana*. Height changes varied by species, likely influenced by interspecific differences in growth form, physiology, palatability to herbivores, hydrology, historical human use, and distribution within and across the EVMP plots. However, similar overall trajectories were observed regardless of species, with height in fenced plots increasing through time and a weaker but positive trend observed in unfenced plots (Figure 30).

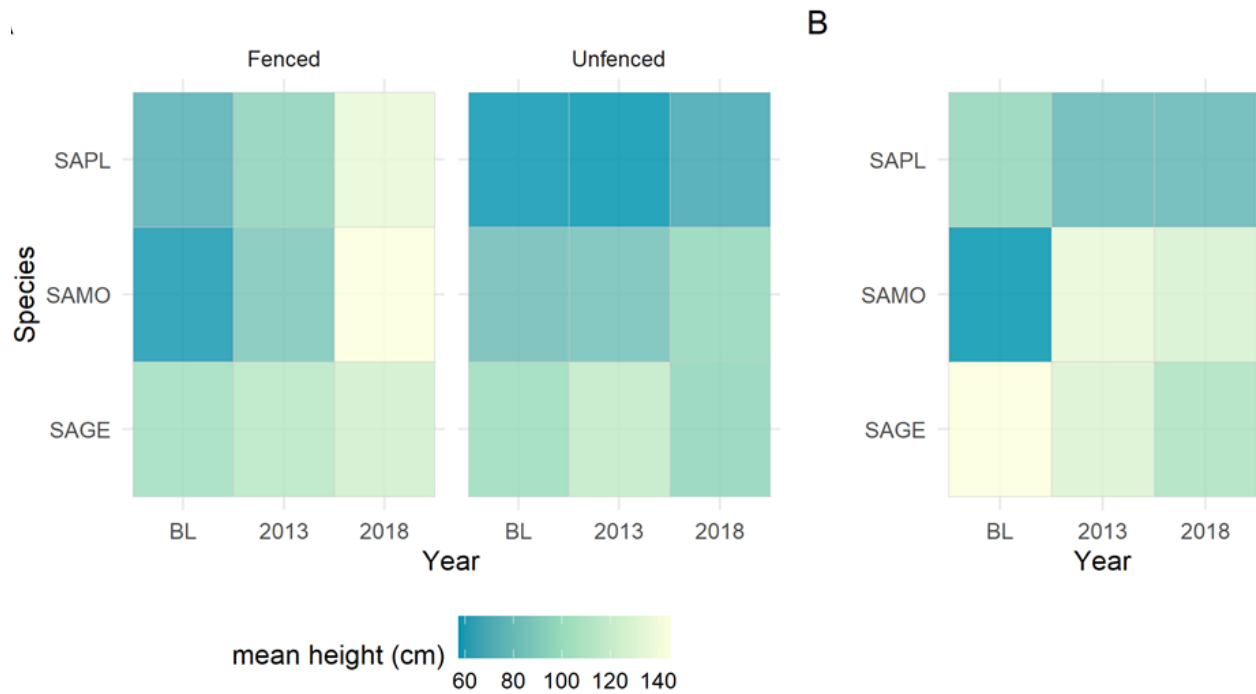


Figure 30. Mean shrub height in (A) fenced and unfenced core and (B) noncore elk winter range plots for the three most abundant willow species on Rocky Mountain National Park elk winter range: *Salix planifolia* (SAPL), *S. monticola* (SAMO), and *S. geyeriana* (SAGE). All noncore willow plots are unfenced. BL = baseline data (collected 2006–2008).

Willow Cover

Modeled mean willow cover varied across the winter range and in the core winter range was highly dependent on whether plots were fenced (Table 7). This is visually apparent in repeated site images (Figures 23–26).

Table 7. Modeled mean willow canopy cover (%) and credible intervals from baseline (2006–2009) to 2018 on core and noncore elk winter range in Rocky Mountain National Park, CO. Analysis includes plots that were burned by the Fern Lake Fire in 2012.

Range type	Baseline mean (CI)	2013 mean (CI)	2018 mean (CI)
Core winter range, unfenced	22.5 (10.7,35.7)	24 (12.7,36.1)	31.6 (18.3,46.1)
Core winter range, fenced	14.6 (5.2,25.8)	25.3 (11.9,40)	70.8 (55.8,85.3)
Noncore winter range	44.5 (28.1,62.3)	57.6 (40.9,73.8)	68.6 (54.5,82.6)

The cover of willows in macroplots was greater in unfenced than fenced plots at baseline, but the pattern was reversed in 2013 and 2018 (Table 7). The highest cover occurred in 2018 in fenced core winter range plots (mean = 70.8%) and unfenced noncore range plots (mean = 68.6%; Table 7). Mean cover nearly doubled from baseline to 2013 and again to 2018 (Table 7) in fenced core winter range plots. Willow cover was unchanged in unfenced core winter range plots between baseline and 2018 (Table 7).

Models suggest that there were increases in willow cover over time and due to fencing in the combined winter range (Figure 31). The probability of direction was positive, but low between baseline and 2013 (PD = 0.77, ES = 0.49), but by 2018 this probability of direction was high and the effect size was large (PD > 0.99, ES = 1.0). Cover increased more strongly in fenced plots, from a low of 15% at baseline to over 70% in 2018. For unfenced plots, cover estimates were higher than for fenced plots at baseline and in 2013, but the reverse was true by 2018 (Figure 31). A similar pattern was observed when using only core winter range sites (Figure 32), with much stronger probability and larger effect size of fencing in 2018 (PD > 0.99, ES = 1.0) than between baseline and 2013 (PD = 0.91, ES = 0.74).

Mean modeled willow cover on the noncore winter range also increased over time from 45% at baseline to nearly 70% in 2018, but the models indicated lower probability of an increase between baseline and 2013 (PD = 0.97, ES = 0.79) than by 2018 (PD = 0.99, ES = 1.0; Figure 33). As with willow height, willow cover response had high spatial variability with wide overlapping credible intervals. In addition, patterns differed between species (Figure 34).

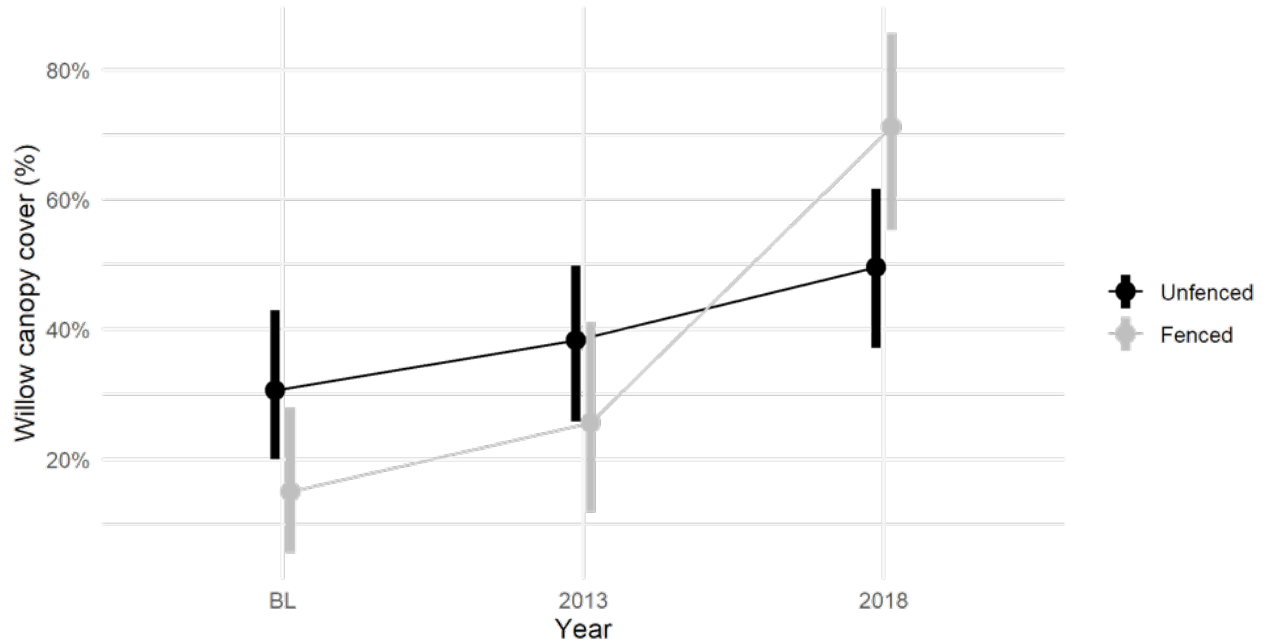


Figure 31. Modeled mean percent willow canopy cover (points) and credible intervals (bars) for all willow species on the combined elk winter range (core and noncore) of Rocky Mountain National Park. BL = baseline data (collected 2006–2008).

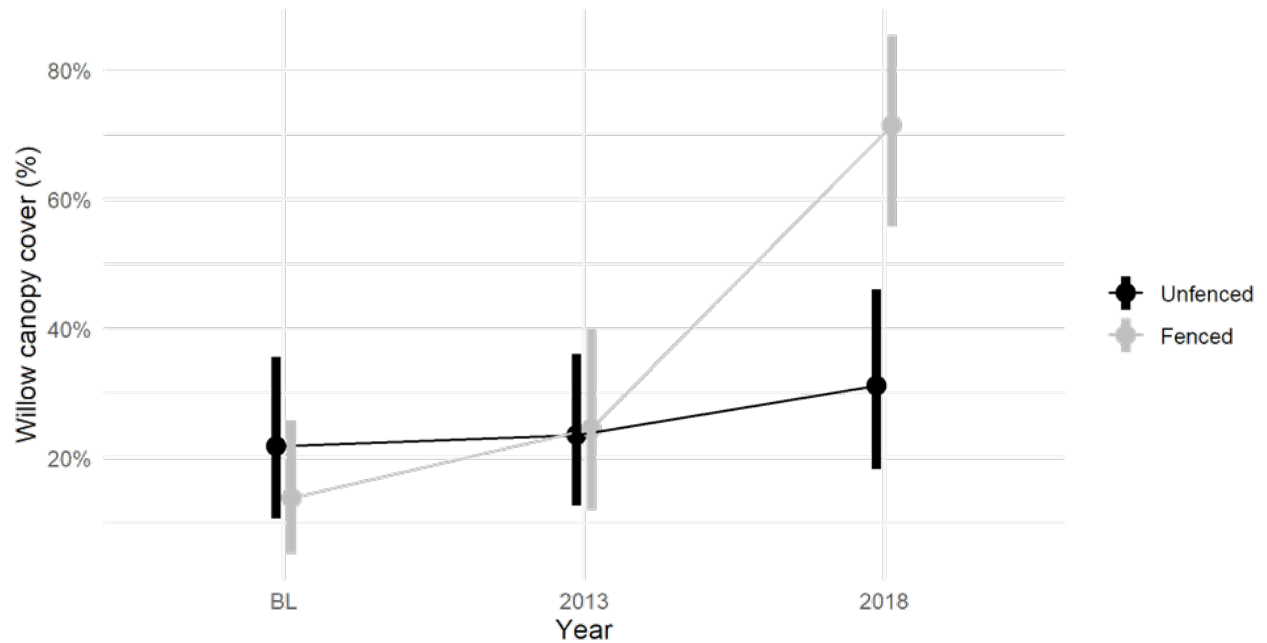


Figure 32. Modeled mean percent willow canopy cover (points) and credible intervals (bars) for fenced and unfenced willow on the core elk winter range of Rocky Mountain National Park. BL = baseline data (collected 2006–2008).

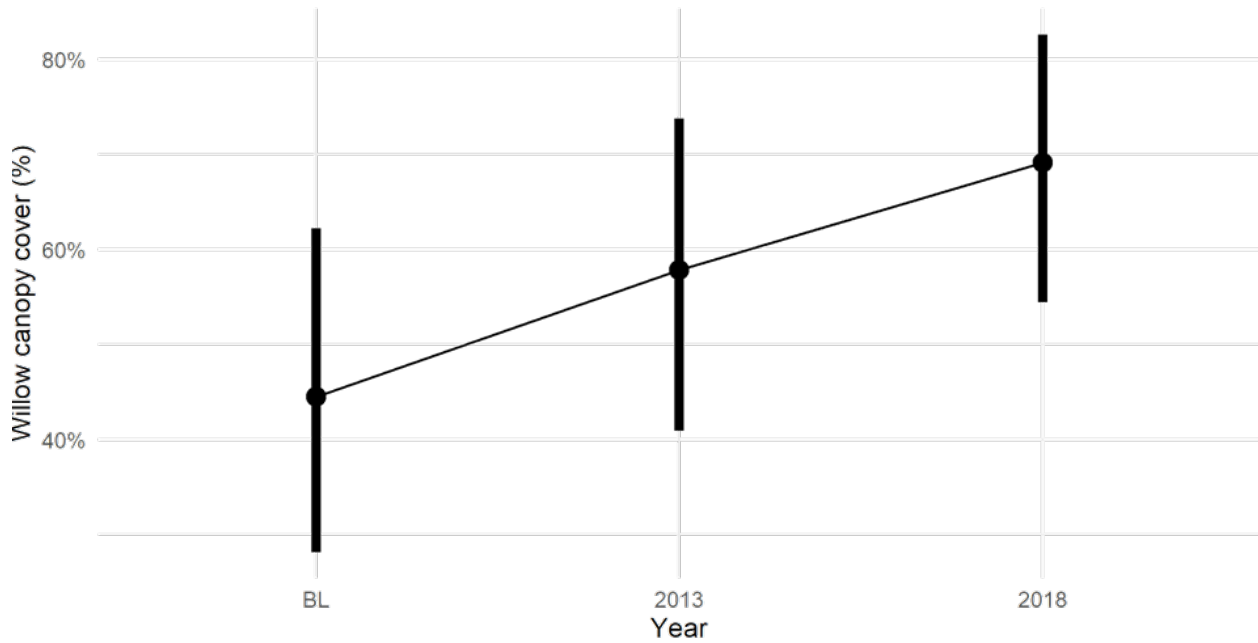


Figure 33. Modeled mean percent willow canopy cover (points) and credible intervals (bars) for all willow species on the noncore elk winter range of Rocky Mountain National Park. All noncore plots were unfenced. BL = baseline data (collected 2006–2008).

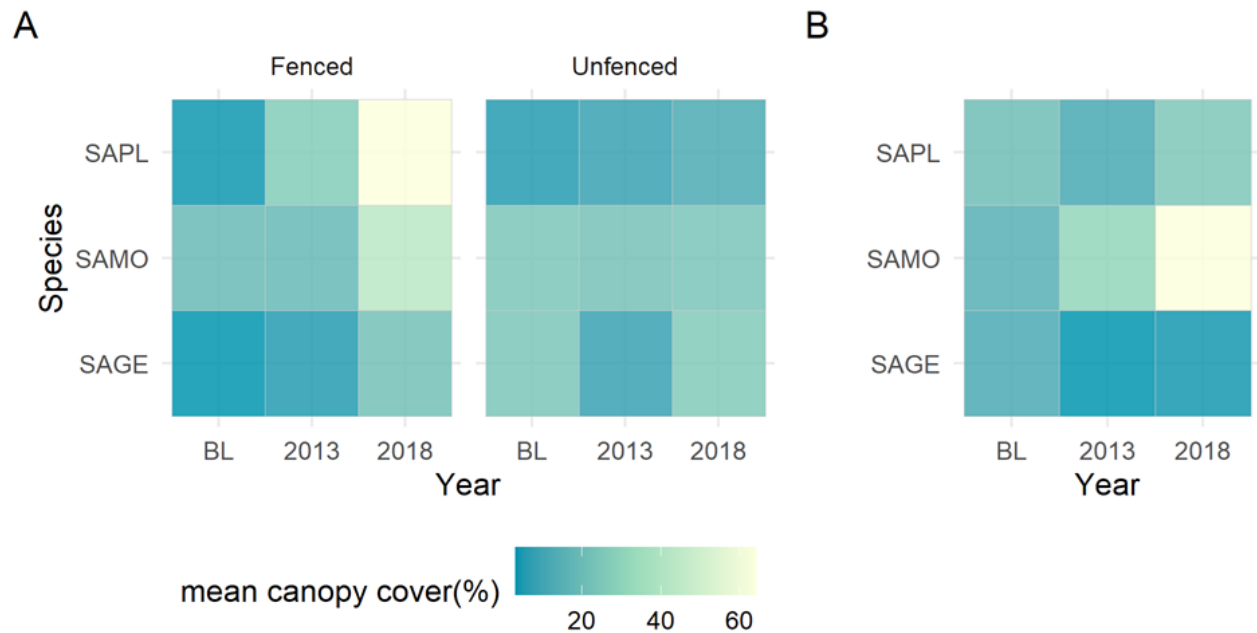


Figure 34. Mean shrub canopy cover for the three most abundant willow species across (A) fenced and unfenced core and (B) noncore elk winter range in Rocky Mountain National Park: *Salix planifolia* (SAPL), *S. monticola* (SAMO), and *S. geyeriana* (SAGE). All noncore willow were unfenced. BL = baseline data (collected 2006–2008).

Effects of Wildfire

Burned plots had lower cover across all time periods in combined core and noncore plots and fenced/unfenced plots. Mean cover was higher in 2018 compared to baseline and 2013 for both burned and unburned plots, with a positive trend across time periods. When including both burning and fencing, the highest mean willow cover (88.2%) was seen in 2018 fenced and unburned plots in the core winter range, followed by unburned plots in the noncore winter range (68.6%; Table 8).

Models of willow cover including the main effects (fencing, burning, and year) and their interactions performed better than those modeling only main effects for core winter range data. The effect of year depended on whether areas were fenced with the greatest willow cover in fenced and unburned plots in 2018 (Figure 35), though the PD values were low (2013-fenced-burned: PD = 0.66; 2018-fenced-burned: PD = 0.53). Whereas willow canopy cover decreased or remained nearly unchanged in burned plots in 2013, immediately following the fire, by 2018, cover had increased inside the fences, (Figure 35), but there was less cover than in fenced sites that did not burn.

Table 8. Modeled macroplot willow (all willow species combined) mean cover (%) and credible intervals, grouped by fencing and burned status.

Range type	Burn status	Baseline mean (CI)	2013 mean (CI)	2018 mean (CI)
Core winter range, unfenced	Unburned	26.9 (12.7,41.9)	30.7 (16.2,46.3)	38.8 (22.8,55.8)
Core winter range, unfenced	Burned	22.9 (0,66)	9.3 (0.9,21.3)	15.3 (2.3,32.8)
Core winter range, fenced	Unburned	23.8 (6.8,43.5)	42.8 (20,66.7)	88.2 (77,96.9)
Core winter range, fenced	Burned	8.1 (1.2,17.8)	11.2 (2.3,22.6)	43 (19.2,66.8)
Noncore winter range	Unburned	44.5 (27.5,61.7)	57.6 (40.8,74.1)	68.6 (52.8,82.2)

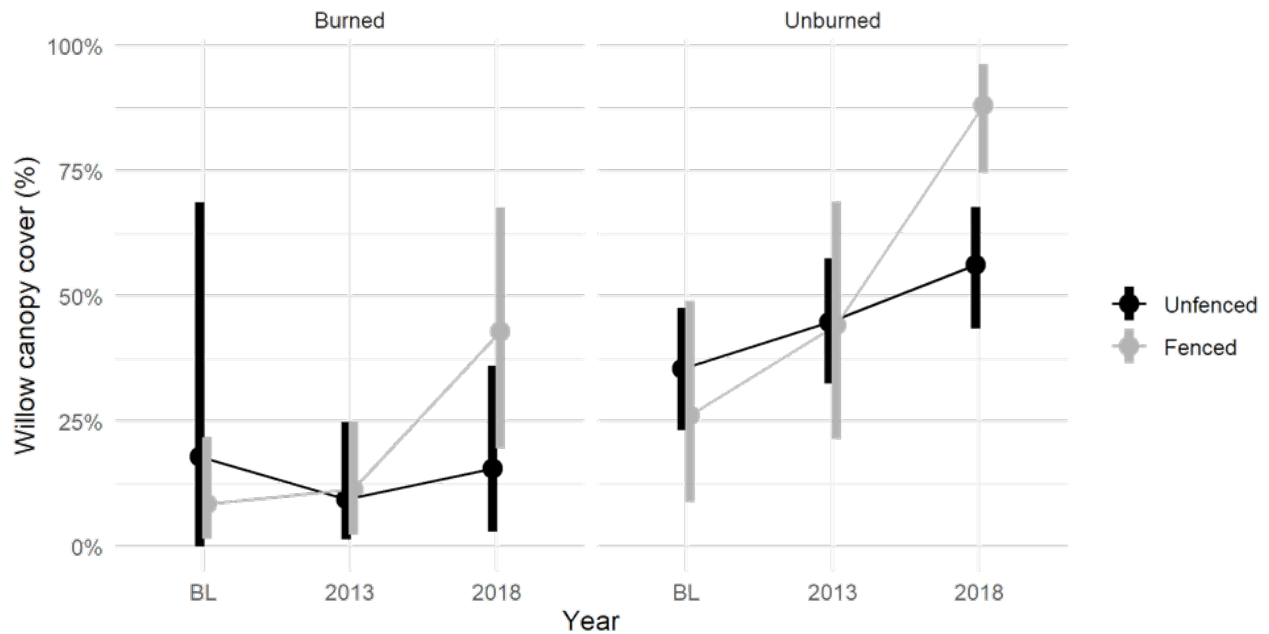


Figure 35. Modeled mean percent willow canopy cover (points) and credible intervals (bars) grouped by burned and unfenced status on the combined (core and noncore) elk winter range of Rocky Mountain National Park. BL = baseline data (collected 2006–2008).

The effect of burning on estimated willow height differed depending on whether plots were fenced or unfenced, though the effect of burning was not as great on willow height as cover (Figure 36). For combined core and noncore elk winter range plots, modeled height increased through time for fenced plots, whether plots were burned or unburned. Credible intervals overlapped in 2018 indicating no differences between burned and unburned plots inside fences or unburned plots outside fences (Figure 36). There was a negative effect of burning on willow heights in unfenced in both 2013 (PD = 0.92) and 2018 (PD > 0.99), this effect was small in 2013 (ES = 0.41) compared to 2018 (ES = 0.93; Figure 36) and was likely due to the complete removal of many willow stems in burned plots. Similar temporal patterns in relation to fencing and burning were seen when just core elk winter range plots were modeled. Very few noncore willow plots were burned, so no separate analysis of noncore willow was performed.

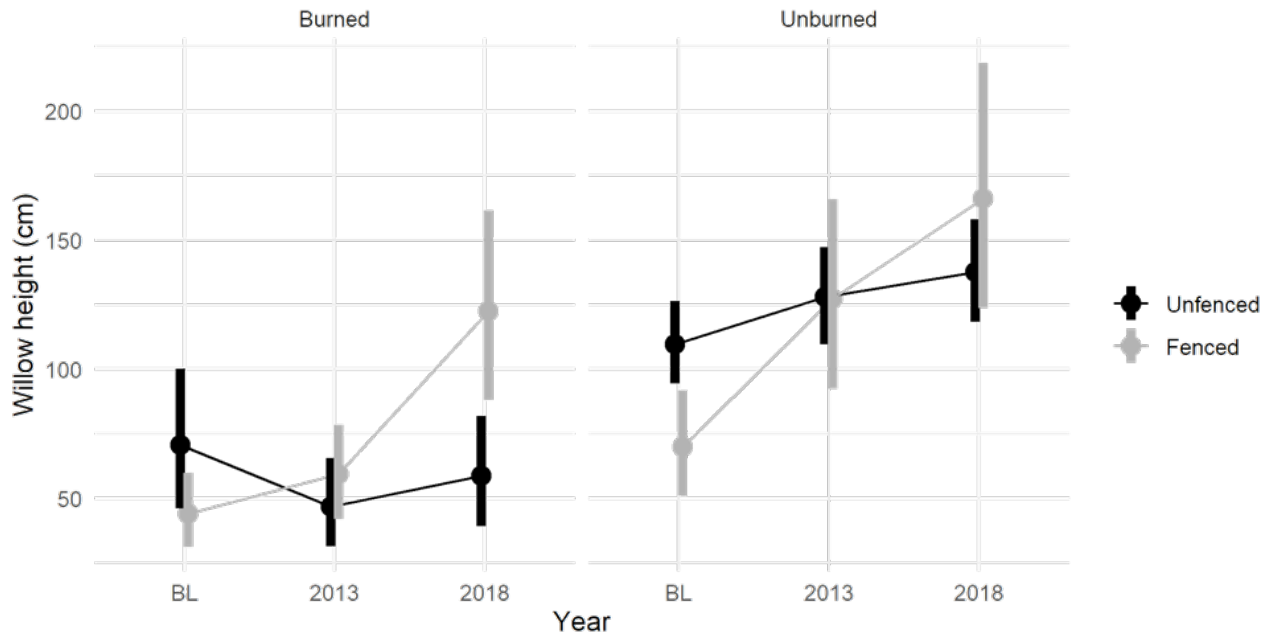


Figure 36. Modeled mean willow height in centimeters (points) and credible intervals (bars) grouped by burned and fenced status on the combined (core and noncore) elk winter range of Rocky Mountain National Park. BL = baseline data (collected 2006–2008).

Range-Wide Willow Cover and Height

We weighted mean willow cover and height for each year sampled based on the proportion of the winter range that fell into each category of range-type, fencing, or burning (Table 9). These proportions changed over time as areas were fenced and burned. Mean cover over the entire winter range did not change much between baseline and 2013 (likely due to losses from the 2012 fire), but a nearly 75% increase in willow cover had occurred by 2018 as well as 35% increase in mean height (Table 9).

Table 9. Weighted modeled mean cover and willow height for each time period and classification. Baseline sampling occurred from 2006 to 2009 on the winter range.

Range	Baseline					2013				2018			
	Weight	Mean cover (%)	Weighted mean cover (%)	Mean height (cm)	Weighted mean height (cm)	Mean cover (%)	Weighted mean cover (%)	Mean height (cm)	Weighted mean height (cm)	Mean cover (%)	Weighted mean cover (%)	Mean height (cm)	Weighted mean height (cm)
Core winter range, unfenced	0.669	22.5	15.1	84.7	56.7	24.0	16.1	85.2	57.0	31.6	21.1	94.1	63.0
Core winter range, fenced	0.202	14.6	2.9	56.9	11.5	25.3	5.1	91.2	18.4	70.8	14.3	144.8	29.2
Noncore winter range	0.129	44.5	5.7	129.6	16.7	57.6	7.4	161.4	20.8	68.6	8.8	171.2	22.1
Entire winter range	–	–	23.7	–	84.9	–	28.6	–	96.2	–	44.3	–	114.3

Kawuneeche Valley

The nine Kawuneeche Valley plots also had relatively low willow cover ($17 \pm 5\%$), mean height (50 ± 4 cm), and maximum height (98 ± 12 cm) at their baseline measure in 2011. All four fenced plots were established within a single enclosure, so they are not representative of the entire Kawuneeche Valley but only of the location of the enclosure. By 2018, mean willow cover increased to 35%, with much of the increase seen in fenced plots (35%) versus unfenced plots (26%).

Models suggested that fencing had a strong positive effect on willow heights in the Kawuneeche Valley from 2016–2018 ($PD > 0.99$ for all years; Figure 37) and this effect was strong over all years ($ES = 1.0$). However, no effect of fencing on willow cover was observed by 2018 ($PD = 0.58$) (Figure 38). As with the winter range plots, in models of willow height for Kawuneeche Valley plots, the effect of time-since-baseline on willow height was conditioned on whether areas were fenced. Contrasts between years were negative for unfenced plots and positive for fenced ones (Figure 37). Fenced plots in the Kawuneeche Valley showed a steady increase in modeled willow height between baseline (50 cm) and 2018 (80 cm; Figure 37). In contrast, modeled willow height steadily declined over time in unfenced plots, from a high at baseline of ~60 cm to 30 cm in 2018. Credible intervals were wide in fenced plots, indicating more variability in height across plots in this group.

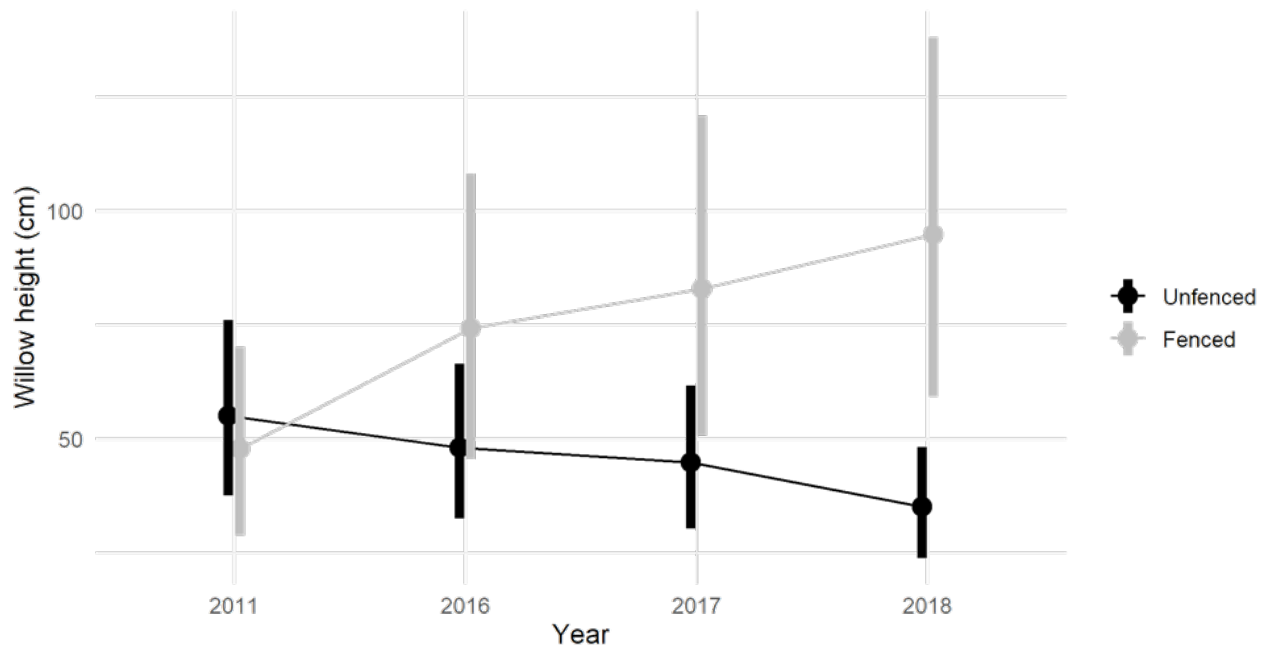


Figure 37. Modeled mean height in centimeters (points) and credible intervals (bars) for fenced and unfenced willow species in the Kawuneeche Valley of Rocky Mountain National Park. BL = baseline data (collected 2011).

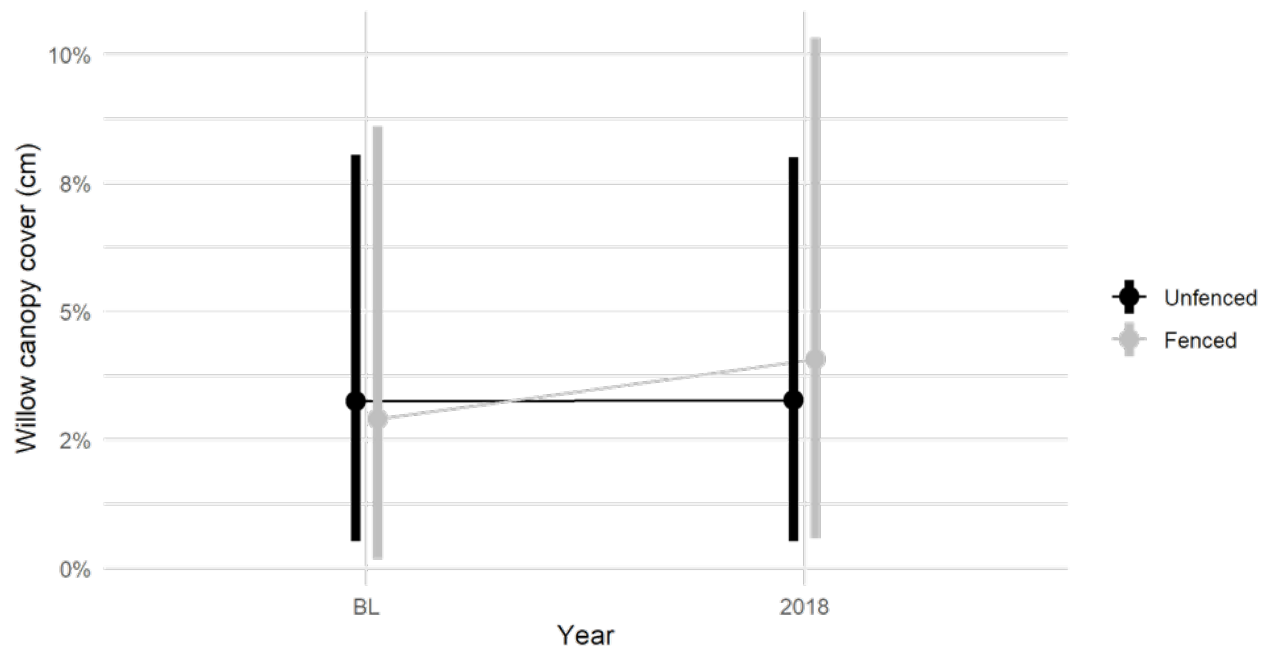


Figure 38. Modeled mean percent canopy cover (points) and credible intervals (bars) for fenced and unfenced willow species in the Kawuneeche Valley of Rocky Mountain National Park. BL = baseline data (collected 2011).

Modeled willow canopy cover in the Kawuneeche Valley was less than 5% in both fenced and unfenced plots at all time periods (Figure 38). Fenced plots showed a slight positive trend between baseline and 2018, but wide credible intervals and low PD values ($PD < 0.58$) suggest little evidence of a trend for either fenced or unfenced plots. This improvement in willow height, but not cover, may be indicative of poor hydrologic conditions that may be limiting willow canopy growth and establishment of new plants in this single location.

Willow Browse Index

We collected browse intensity data annually because the high interannual variability in offtake patterns which can be caused by weather conditions, short-term loss of habitat due to fire and floods, or in response to short-term changes in human use (e.g., road closures or construction) make it difficult to determine patterns based on a single observation every 5 years. Patterns of willow browse intensity differed among years and management subgroups (e.g., core and noncore winter range, fenced and unfenced plots). Fencing predictably reduced or eliminated browsing from those plots, but within unfenced plots, offtake varied widely among plots. The percent of stems consumed ranged from 0 to 100%, with median values less than 30%. However, all years had some stems which had browsing on all shoots in some plots.

Mean browse intensity of willow appears to have declined overall since 2010 (Figure 39). The core winter range experienced an uptick in willow browse intensity in 2015 and 2016 and a similar increase was evident in the noncore winter range in 2016. Throughout the last decade, core winter range browsing has remained higher than noncore. Browse intensity on willow in the Kawuneeche

Valley has also declined since measurements began in 2012, with an increase in 2016 and 2017, but browse intensity was at its lowest measured level in 2018 (Figure 39).

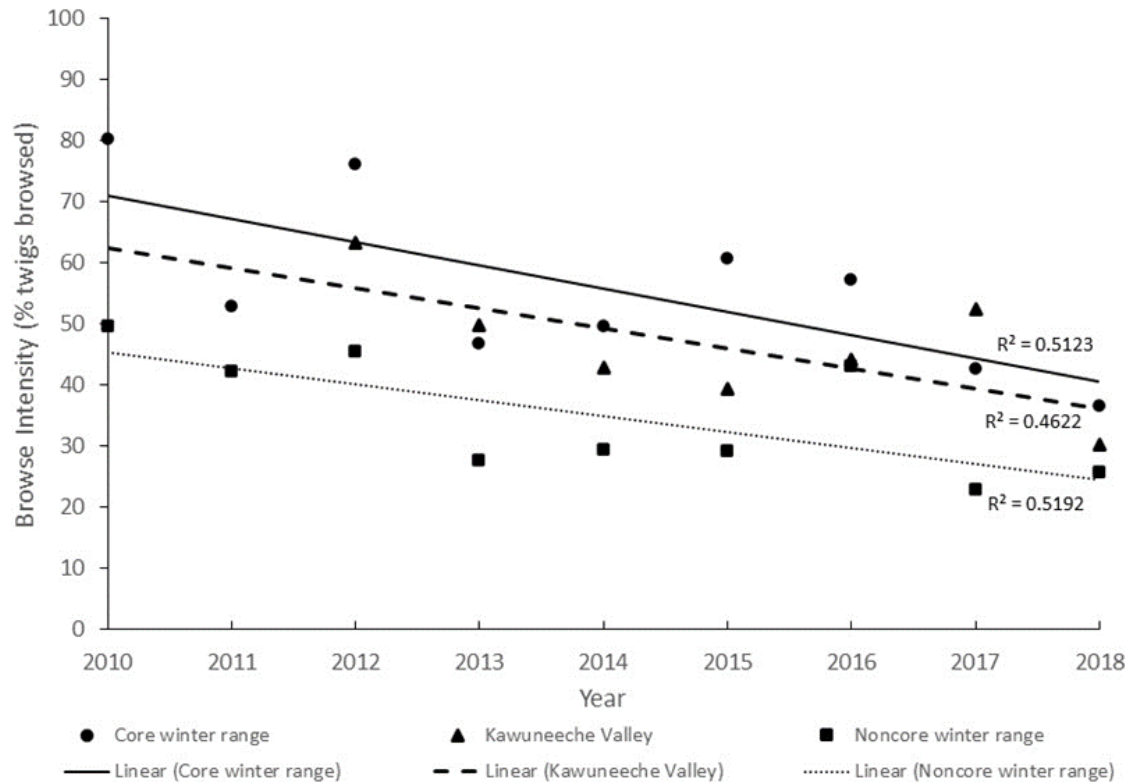


Figure 39. Willow browse intensity on the elk winter range and Kawuneeche Valley of Rocky Mountain National Park 2009–2018. R² values are for linear trend lines.

Progress Toward EVMP Willow Goals

A key EVMP management objective is a progressive increase in mean willow height across the winter range from the baseline of 0.9 m tall to at least 1.1 m tall. Desired future conditions also include an increase in willow height, with average height increasing by at least 20% relative to baseline conditions. Analyses presented here show that this threshold has or likely will soon be reached, but primarily driven by increases in areas protected from ungulate browsing by fencing. Willow height is an especially important variable because it strongly influences habitat suitability for beaver and understory vegetation condition, since taller plants produce more biomass needed for winter food caches and are more suitable than short-statured plants for dam building (Allen 1983; Baker et al. 2005; Wolf et al. 2007).

The EVMP also identified a desired future condition of up to 70% willow cover within suitable habitat. While this level of recovery was not thought to be attainable within the 20-yr timeframe of the plan, the EVMP threshold called for progress toward this desired future condition of >10% increase in willow cover over baseline conditions. At this time, the winter range in its entirety is halfway to the desired future condition with roughly 35% willow cover in areas that are considered willow habitat. However, this marks a roughly 10% increase in cover compared to baseline and is on

track to achieve the 20% cover increase desired in the 20-yr timeframe of the EVMP. The minimum desired future condition of achieving 31% willow covering within this timeframe has been achieved.

The use of fencing is a key management strategy in the EVMP. Results from this analysis demonstrate that the greatest progress towards EVMP goals for willow height and cover is within fenced areas. Past analyses also found that progress towards management goals were most pronounced in fenced areas (Zeigenfuss and Johnson 2015). After an additional five years has passed, this trend has continued. Differences in structure inside and out of exclosures is due not just to the obvious influence of herbivory. Elk exclosures have generally been established in areas deemed likely to support desired communities based on historical conditions and factors like hydrology (Zeigenfuss et al. 2002; NPS 2007; Zeigenfuss and Johnson 2015).

Willow browse intensity appears to have decreased by roughly 5–10% since baseline measurements (Figure 39) as well. Overwinter elk population size has declined steadily on the core and noncore winter range since baseline (Figure 1; Hobbs and Abouelezz 2020). Future levels of elk herbivory will be influenced by direct management actions, such as culling and active redistribution of animals, and indirect factors influencing ungulate population size and movement (e.g., weather conditions, changes in management outside of the park boundaries). Continued monitoring of vegetation and ungulate use will be essential for understanding the ecological consequences of such changes.

Although there is a shorter record of monitoring data for the Kawuneeche Valley, anecdotal data and repeat photographs indicate a substantial decline in willow condition. Factors like *Cytospora* fungi, which are more likely to infect browsed stems, possibly exacerbated by changes in climate and hydrologic regime, have been shown to contribute to dramatic diebacks in willow canopies (Kaczynski 2013; Kaczynski and Cooper 2013; Kaczynski et al. 2014).

Wild Basin, in the southeast portion of the park, was not included in the original EVMP design because it was not elk winter range. However, this area supports some of the most intact riparian willow communities remaining in RMNP and relatively high beaver density (Zeigenfuss and Abouelezz 2018). Consequently, it serves as a potential reference site for vegetation restoration or beaver habitat suitability assessment. Anecdotal observations, combined with moose presence detected during 2018 vegetation surveys, suggests that Wild Basin now has many more moose than in the past, with unknown consequences for riparian vegetation. Using similar methods to the EVMP, USGS and RMNP personnel established and sampled plots in Wild Basin in 2018. These data are fully analyzed in Zeigenfuss and Abouelezz (2018). The NPS Rocky Mountain Inventory and Monitoring Network (ROMN) also established multiple plots in Wild Basin and will monitor these over the long term (Schweiger et al. 2015) and their data will be presented in future ROMN publications. Wild Basin had much greater maximum and median willow height when compared to the 2018 EVMP data. Comparisons to EVMP data suggest that many areas are still far from reaching willow height conditions like those seen in Wild Basin. The median willow height in Wild Basin was 210 cm, much greater than willows in the core winter range plots (122 cm), noncore winter range (179.8 cm), and Kawuneeche Valley plots (73.5 cm); Figure 40). Long-term data are needed from these areas to contextualize data from EVMP monitoring sites and to allow adaptive management of riparian resources as the population and behavioral ecology of ungulates change. Increased sampling

efforts in these areas may be especially valuable. Continued monitoring in this area of RMNP is critical both because of its use as a reference for other riparian areas and its potential vulnerability to changes from a growing moose population and climate change.

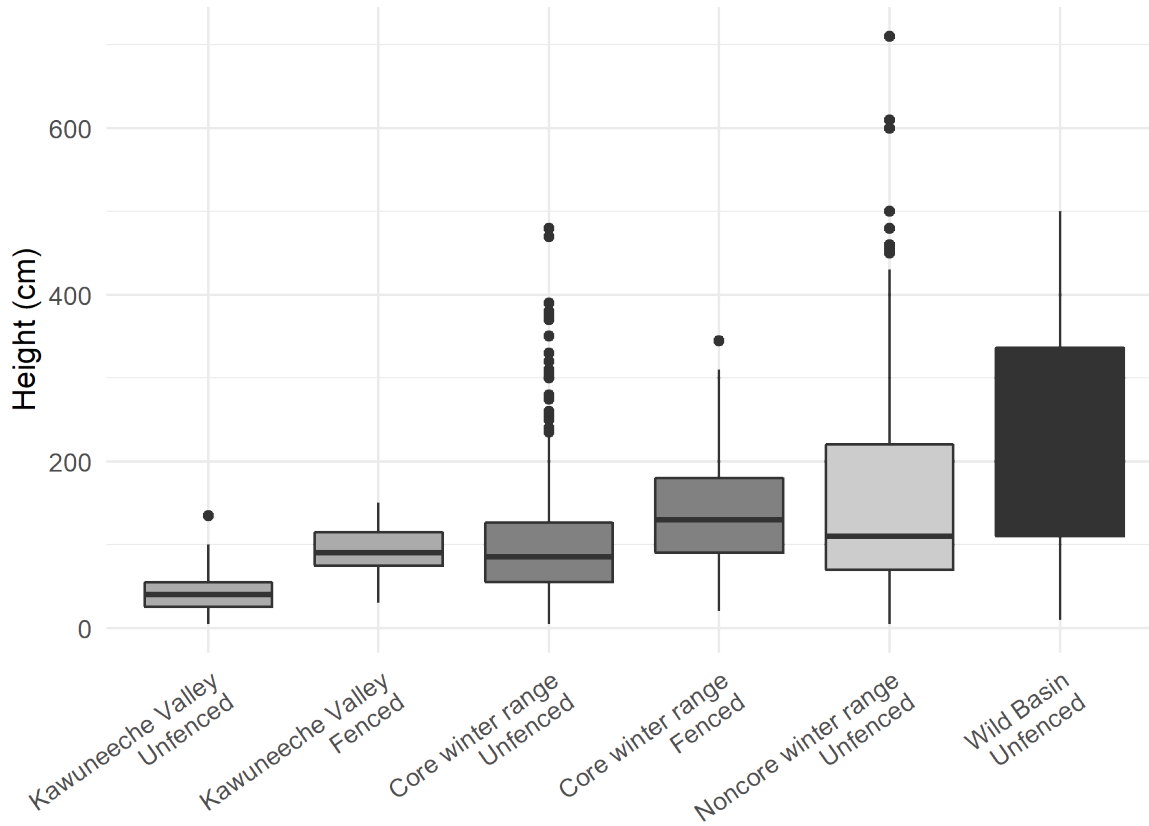


Figure 40. Boxplots comparing willow height distributions (all willow species combined) among Wild Basin, fenced and unfenced core elk winter range, noncore elk winter range, and Kawuneeche Valley plots. Height data are for willows in 2018.

Many riparian areas in the winter range have become drier in the absence of beaver and their dams that raise water tables (Westbrook 2005; Westbrook et al. 2011) and ongoing localized human disturbances (Schweiger et al. 2016). In western Moraine Park, distributary channels hydrologically supported by beaver dam-induced channel avulsions historically supported willows but now support grasses and shrubs (Peinetti et al. 2002). Moraine Park was also once a golf course and while this area has been restored there are likely many residual impacts from intensive historical land use (Schweiger et al. 2016). Channel formation and incision in groundwater and sheet flow-dominated ecosystems like Upper Beaver Meadows have reduced suitability for willow growth and reproduction (Graf 1997). Historic homesteads in the Kawuneeche Valley still contain ditches affecting water table depths. Identifying and remedying such hydrological constraints through active restoration approaches like simulated beaver structures may be useful to accelerate willow height and cover trends (Pollock et al. 2014; Weber et al. 2017; Wheaton et al. 2019). RMNP is currently evaluating the suitability of simulated beaver structures for habitat restoration. Data from these projects can be compared to EVMP data to better understand the potential applications for management.

Riparian ecosystems are spatially and temporally dynamic, and a challenge for monitoring is simultaneously capturing the effects of discrete natural and human-caused disturbances and long-term changes due to factors like climate change. EVMP monitoring is well-positioned to provide such information, especially as the length of the data record increases. Each valley evaluated in the EVMP has a different site history, vegetation structure, and ecological condition, as well as different levels of herbivore use. Moraine Park has arguably been altered the most relative to pre-Euro-American settlement (Peinetti et al. 2002; Baker et al. 2005, 2012). Large areas within Moraine Park that historically supported tall willow communities are now dominated by herbaceous communities and there is little evidence of new willow establishment (Gage and Cooper 2004, 2005; Cooper et al. 2006).

Upland

For combined core and noncore winter range plots, the highest mean upland shrub cover (across all six species combined) was in 2018 (Figure 41A). The PD values indicate low probability of either a positive or negative change in shrub cover over time (2013: PD = 0.55; 2018: PD = 0.63), indicating that major type conversions from herbaceous dominated to shrublands are not occurring at this point. PD values were similar when core and noncore were separated, but noncore upland plots had higher shrub cover than core winter range upland plots across all time periods (Figure 41B). In models of cover, pairwise contrasts were not different from one another for most contrasts.

No change in cover for individual species was observed over time, but height varied over time and between core and noncore winter range (Figure 42). For example, maximum height of *Artemisia tridentata* (ARTR) declined from baseline conditions on the noncore winter range but was unchanged between baseline and 2018 core winter range plots (Figure 42).

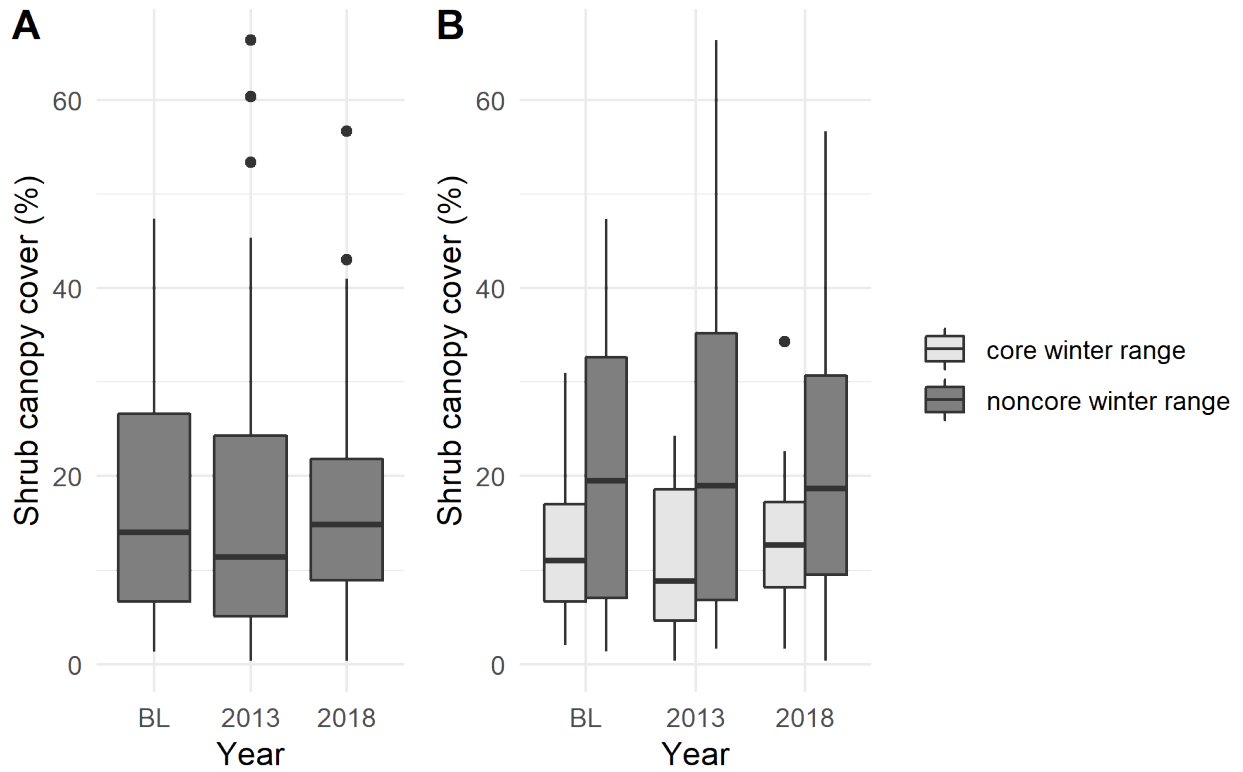


Figure 41. Box plots of percent canopy cover for all upland shrub species pooled in (A) combined core and noncore elk winter range and (B) separated by range type. BL = baseline data (collected 2006–2008).

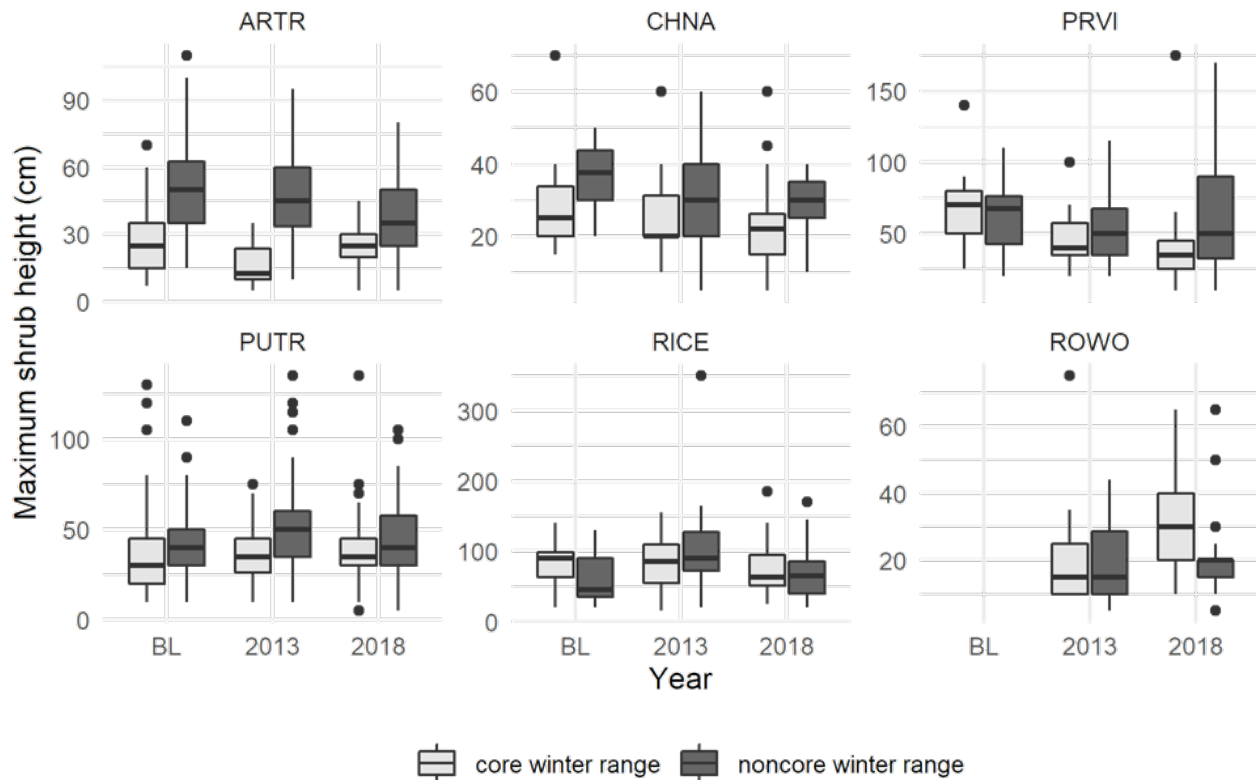


Figure 42. Maximum shrub height along upland line intercept plots for most common shrub species in core and noncore elk winter range areas (ARTR = *Artemisia tridentata*; CHNA = *Chrysothamnus nauseosus*; PRVI = *Prunus virginiana*; PUTR = *Purshia tridentata*; RICE = *Ribes cereum*; ROWO = *Rosa woodsii*). BL = baseline data (collected 2006–2008). Note different Y axis scales in most panels.

Progress Toward EVMP Upland Grassland Goals

A goal of the EVMP was to reduce the level of elk grazing on upland herbaceous vegetation. It also identified maintenance of a range of grazing levels as part of desired future conditions. Management objectives for upland communities in the EVMP include: (1) No net increase in winter upland herbaceous offtake across the winter range above baseline levels of 61 percent by the revised calculation method, with less than or equal to 30 percent of sites with offtake >70% and <10% of sites with offtake greater than 85%. The calculation of baseline herbaceous offtake was originally 47% using a method that included negative offtake values that occur when the amount of biomass outside the grazing cage is greater than that inside the cage. Including these negative values can depress the offtake value and in 2013, a revised method was used to calculate offtake which involved removing these negative data points from the dataset. Baseline offtake values were revised to 61%. These upland herbaceous offtake objectives were met in 2013 and after consultation with the expert panel, the park discontinued monitoring of upland herbaceous offtake. Monitoring of shrub conditions every 5 years at upland grassland sites has continued. The original intent of such monitoring, though outside the scope of the EVMP, was to identify any changes in plant composition which would indicate type conversion from grasslands to shrublands. As of 2018, such type conversion does not seem to be occurring. If resources for monitoring of all EVMP plots becomes limited in the future, the elimination of shrub sampling in upland herbaceous sites should be

considered. However, if elk population numbers begin to increase, patterns of elk use appear to shift away from willow communities due to habitat loss or competition with moose, or increasing use of the upland areas is observed during seasons other than winter, then RMNP managers should re-evaluate collection of upland herbaceous offtake data.

Moose and Beaver Presence

Moose

All moose presence in Beaver Meadows, Horseshoe Park, and Endovalley willow was detected outside of exclosures other than one plot inside the Endovalley Alluvial Fan exclosure fence. This fence had been damaged and partially open since the 2013 flooding event, and therefore was not actively excluding animals. Moose presence in Moraine Park, in contrast, was detected in 65% of willow plots outside of exclosures and 21% of willow plots within exclosures. Across all core and noncore winter range plots in 2018, 37% of aspen plots, 63% of willow plots, and 21% of upland plots contained fresh moose sign. Data from Kawuneeche Valley plots are not included in these percentages. At willow plots, moose presence was detected in all major drainages across the EVMP core and noncore winter range in 2018 with a higher concentration of plots containing moose sign observed in noncore winter range (Figure 43). All willow plots in Hollowell Park, all but one in Hidden Valley, and all but one in Cow Creek contained moose sign. In the core winter range, moose sign was most prevalent in willow plots in Endovalley west of Horseshoe Park (100% of plots to the west of Highway 34 in the Endovalley/Horseshoe Park area).

Moose presence was measured in core and noncore winter range aspen plots in 2013 and 2018 (Figure 44). Moose sign increased from presence in 1% of winter range aspen plots in 2013 to 37% of plots in 2018. Only one plot in northwest Beaver Meadows contained moose sign in 2013. The highest concentration of plots with moose sign in 2018 was observed in the Endovalley/Hidden Valley area.

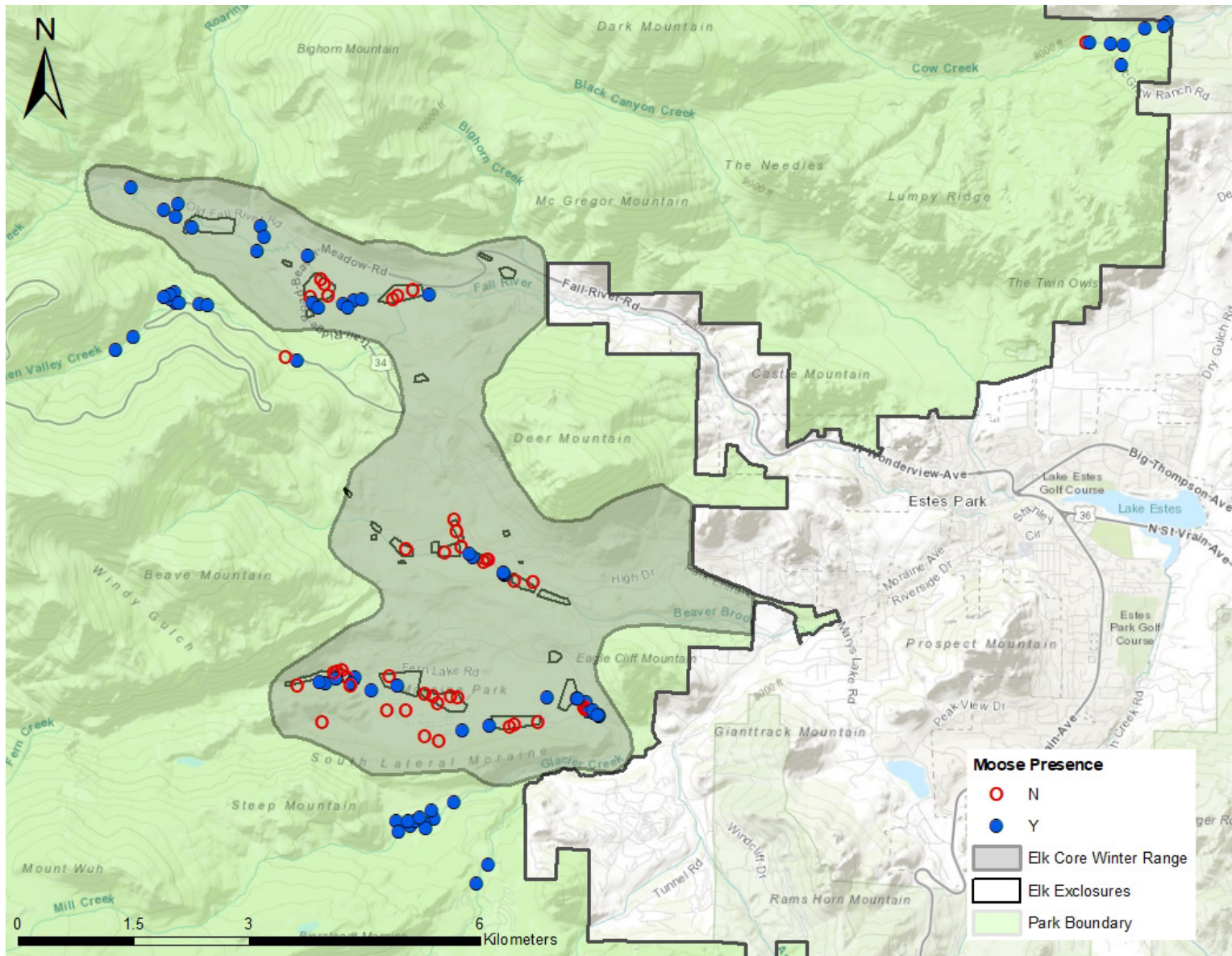


Figure 43. Moose presence detected in Elk and Vegetation Management Plan (EVMP) core and noncore willow plots in 2018.

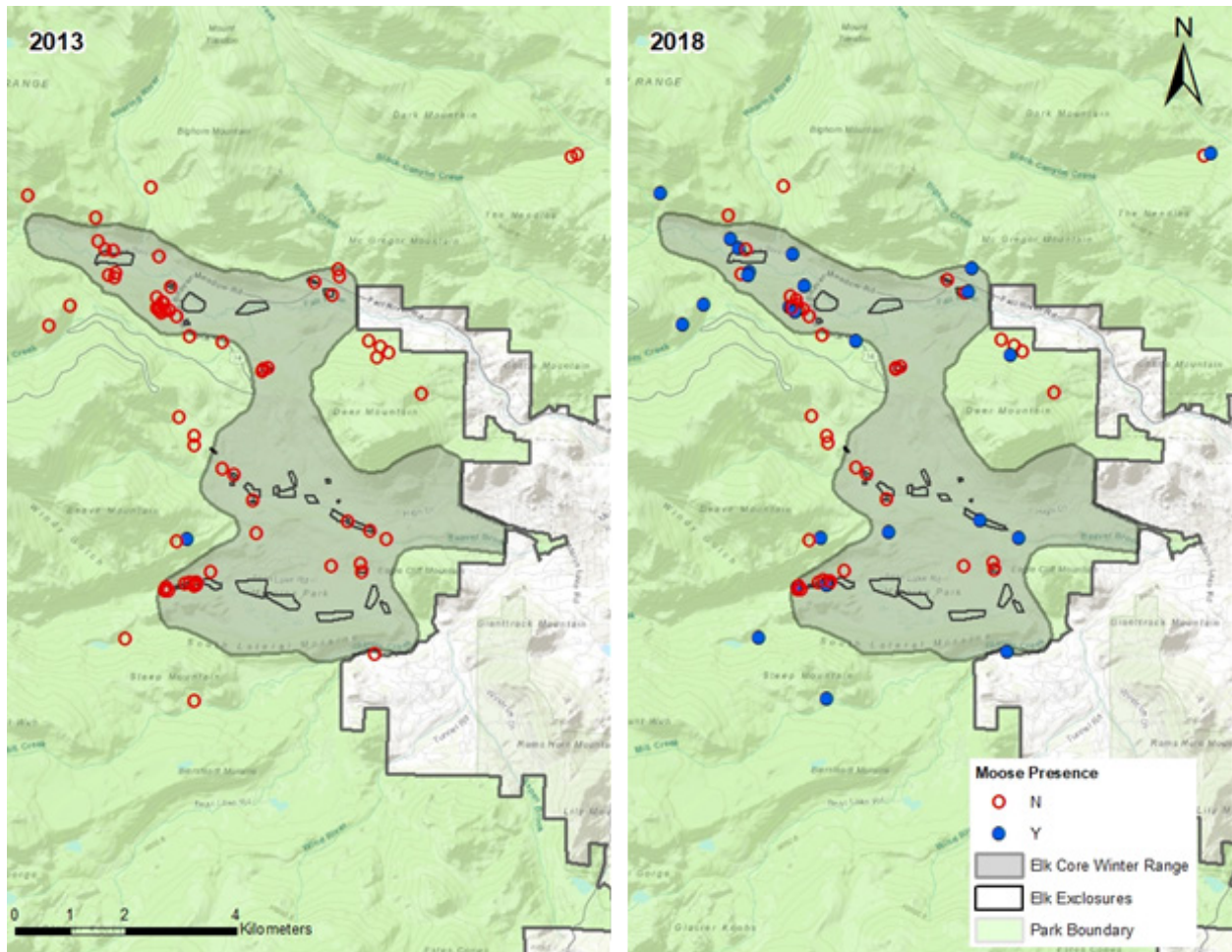


Figure 44. Moose presence in Elk and Vegetation Management Plan (EVMP) core and noncore aspen monitoring plots from 2013 to 2018.

Beaver

Beaver presence was recorded in willow plots during baseline, 2013, and 2018. Overall beaver presence decreased between baseline and 2018 in all major drainages other than Beaver Meadows (Figure 45), which increased only slightly from zero to seven percent. One plot in Hidden Valley contained evidence of a dam in 2013, but this was the only detection of beaver presence in the EVMP Hidden Valley willow plots across all survey years. Beaver presence in Moraine Park decreased from 33% to 0% from baseline to 2013 but increased to 5% in 2018 when cuttings were detected in two plots. Notable decreases were observed in Cow Creek and Mill Creek Basin from baseline to 2018 where presence dropped from 63% to 13% and 64% to 0% of willow plots, respectively.

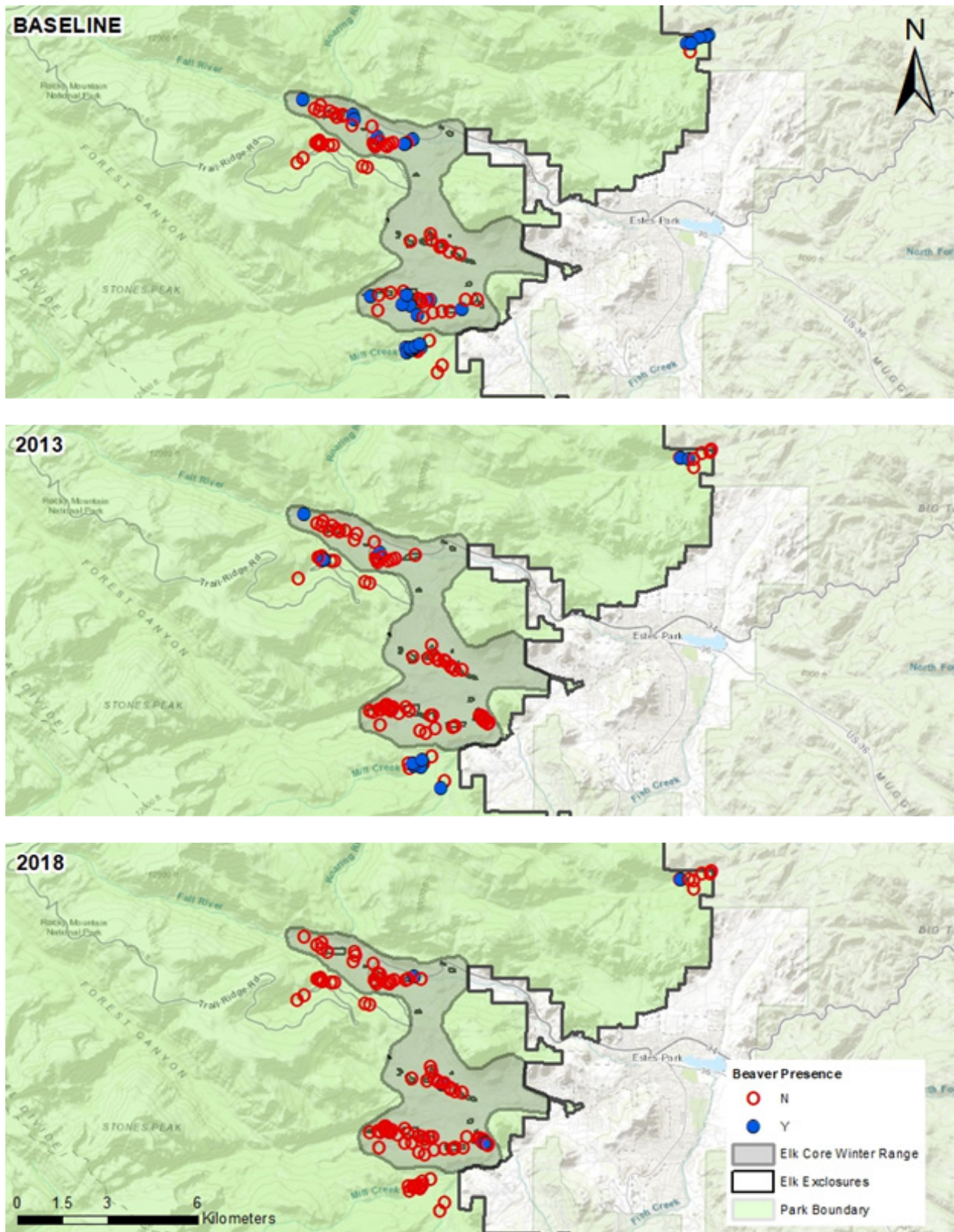


Figure 45. Beaver presence in Elk and Vegetation Management Plan core and noncore willow monitoring plots from baseline to 2018.

Although not statistically analyzed, it appears that moose are increasing at both willow and aspen monitoring sites. Beaver presence appears to have decreased across the winter range willow areas and at the time of this sampling, signs of beaver presence were only detected in a few of the EVMP monitoring sites. However, anecdotal evidence of an increase in beaver activity at or near many of the willow monitoring sites has occurred since 2018 (Nick Bartush, RMNP, personal communication, 2022). It is possible that the lack of beaver activity noted in 2018 was due to floods in the years between 2013 and 2018 which destroyed many of the remaining beaver dams on the winter range.

Conclusions

The EVMP presented a range of desired future conditions and associated thresholds and indicators for monitoring (NPS 2007; Zeigenfuss and Johnson 2015). These differed for aspen, riparian willow, and upland plot types. Monitoring data collected during the EVMP are essential for adaptive natural resources management (Lindenmayer and Likens 2010). The variables and thresholds included in the EVMP were designed to inform adaptive management of elk and vegetation, allowing managers to adjust management as necessary to achieve the management goals (Table 10, NPS 2007). EVMP data could also be useful for assessing long term responses to environmental stressors like climate change and tracking the trajectories of ecological succession in response to management actions like fencing or active vegetation restoration.

Results in this report demonstrate progress towards the vegetation goals of the EVMP. Trends suggest that many objectives can be met by the end of 2028, the timeframe specified in the EVMP, but not for all sampled sites in the winter range. The increases in plant height and cover inside fenced enclosures suggest that additional fencing of willow and aspen may be needed to achieve these objectives. Roughly a quarter of willow in the core winter range and ~10% of all aspen are protected by fencing (Zeigenfuss and Johnson 2015). Strategically planting and protecting willows may also be a useful strategy to hasten achievement of EVMP objectives.

The reduction in elk population size on the winter range has coincided with reduced overwinter browse intensity on willow and reduced offtake in herbaceous uplands. Wider distribution of herbivory was also documented in 2013, possibly contributing to improved range condition. Elk utilize the winter range as calving areas during the spring and rutting areas during the fall, with some animals continuing to overwinter as well. Both unfenced aspen and willow lag behind fenced communities in recruitment (aspen), cover and height (willow), continuing a trend that was first demonstrated in 2013. Herbivory may still be limiting on plants unprotected by fencing.

This analysis reveals complex patterns of change in vegetation over time. The EVMP sought to promote a progressive increase in willow cover and height and aspen regeneration across the entire elk winter range but recognized that changes in variables may not be linear and may vary widely within different parts of the winter range (e.g., core versus noncore areas), and in response to management interventions like fencing and in response to unanticipated changes like fire. To understand how these factors influence progress towards desired future conditions, we separated the effects of fencing, wildfire, and range type in our analyses. This approach can inform adaptive management better than analyses of plot data combined across these variables.

Table 10. Progress toward desired future conditions in relation to thresholds specified in the Elk and Vegetation Management Plan (NPS 2007; Zeigenfuss and Johnson 2015).

Category	Desired future condition ¹	Thresholds	Current status (as of 2018)
Aspen	At least 45% of aspen stands across the winter range regenerating.	Progressive increase in aspen regeneration above the baseline level of 13% (presence of stems < 2 cm dbh reaching 150–250 cm tall).	Regeneration occurring on 39% of winter range stands.
	Distribution of stem dbh reflects many (~75%) small-diameter stems, some (~20%) medium-diameter stems, and few (~5%) large-diameter stems	Progressive shift in the distribution of stem sizes toward the desired future condition	Percent of stands with >75% of small diameter stems has increased from 4% to 15% since baseline.
Riparian Montane Willow	At least 31% willow cover within suitable willow habitat across the winter range.	Progressive increase in willow cover across the winter range above the baseline level of 21%.	Willow cover across the winter range is 35%.
	Average willow height of at least 1.1 m.	Progressive increase in willow height across the winter range above the baseline level of 0.9 m	Average height of willow across the winter range is 127 cm (1.27 m)—largely driven by increases inside elk exclosures.
	–	No net increase in annual willow offtake across the winter range above the baseline level of 35%.	Detailed offtake measures discontinued 2013. Current measures indicate browse intensity has decreased roughly 5–10% since baseline.
Upland Herbaceous	Reduction of the level of elk grazing on herbaceous vegetation and maintenance of a diversity of grazing levels across the landscape	No net increase in winter upland herbaceous offtake across the winter range above baseline levels of 61% ² , with ≤ 25% of sites with offtake > 70% and ≤ 10% of sites with offtake > 85%.	Achieved as of 2013.

¹Desired future condition of willow are specific to the 20-yr timeframe of the Elk and Vegetation Management Plan.

²Baseline level of offtake in the EVMP Environmental Impact Statement (NPS 2007) was 47%, however, the method for calculating baseline offtake was revised in 2013 resulting in a baseline value of 61% (Zeigenfuss and Johnson 2015).

Key EVMP variables including aspen regeneration, and willow height and cover had positive trends over time, especially in fenced plots protected from herbivory. There was high spatial variability among core and noncore winter range, some of which is likely a consequence of random variation in the placement of sampling points in the EVMP design. It was surprising, however, that fencing had a strong effect even though elk numbers were quite low during the most recent 5 years, particularly given that overwinter elk populations in RMNP are roughly 1/3 their size at the outset of the EVMP, and roughly 9-fold lower than the largest population in the early 2000s. It might be expected that such drastic reductions in wintering elk populations should result in reduced offtake and herbivory impacts to key vegetation communities without the need of fences. Our results may indicate that palatable species like willow and aspen are going to be browsed heavily regardless of overwinter elk population size, either because they are so desirable for elk to eat or because heavy offtake is still occurring on the winter range. Highest levels of use may no longer be occurring during winter, but rather spring through fall when elk that migrate to lower elevations spend several weeks on the winter range as they move to and from their high elevation summer range in RMNP. However, the trend toward taller plants in the unfenced plots might be interpreted as effect of reduced elk numbers.

The results over the first 10 years of the EVMP clearly indicate the value of fencing for both aspen and willow communities. Fenced areas had greater recruitment in aspen stands and greater height and cover in willow stands. However, fencing is not intended to be a permanent solution (NPS 2007; Zeigenfuss et al. 2011). It is important to recognize that other constraints may affect future trends. For example, shifts in temperature, precipitation and aspects of the water cycle like increasing climatic deficit and reduced runoff may have important consequences for annual growth and production (Eckersten et al. 1987; Marshall et al. 2014; Thoma et al. 2020).

The Fern Lake Fire provided conditions that allowed investigation of the effect of wildfire/burning on vegetation response. Burning had a positive effect on aspen regeneration and, potentially, recruitment into tree class. This positive response was strongest inside fenced areas, suggesting that herbivory may still be limiting post-burn recruitment in areas unprotected by elk exclosures. Burning could be utilized in concert with fencing to increase the rate of aspen regeneration. Burning resulted in a negative effect on willow cover, and, as such, would not be recommended as a management tool to meet objectives of the EVMP. Burned aspen plots exhibited higher rates of suckering, but recruitment of stems in areas where ungulate densities remain high is likely to be restricted to exclosures (Romme et al. 1995; Durham and Marlow 2010). RMNP added additional plots in burned areas in 2013, with annual monitoring conducted to provide insights into the effects of fire and herbivory on aspen regeneration and recruitment. Monitoring these sites for the duration of the EVMP will be useful for evaluating long-term effects of fire and herbivory on aspen recruitment.

The EVMP was developed with a focus on the effects of elk on vegetation. However, over the past decade moose have increased in abundance and expanded their range to include all RMNP valleys (Zeigenfuss and Abouelezz 2018). While less numerous than elk, moose are large, obligate browsers that can significantly impact aspen and willows (Dungan and Wright 2005; Butler and Kielland 2008). While elk move and feed in herds creating spatial heterogeneity in herbivory across landscapes (Pearson et al. 1995; Wallace et al. 1995; Binkley 2008), moose remain largely solitary or

in small groups. Because of their novelty in RMNP, the long-term effects of moose are less known, but increased browse pressure from moose may result in changes on RMNP's east side similar to those observed in the Kawuneeche Valley where moose have been present for a longer time.

Recommendations for Future Data Collection and Analyses

A key strength of the EVMP data is its emphasis on rigorous field data collection. Because the EVMP protocols are time consuming and expensive to implement, it may be desirable to integrate less-costly remote sensing approaches to evaluate certain habitat characteristics like willow height. These would not substitute for field-based data collection but could complement it. For example, if airborne LiDAR becomes available for EVMP areas, vegetation height can be effectively assessed at broader spatial scales and with high precision (Farid et al. 2006).

In contrast to elk, moose have increased in number and expanded into new areas of RMNP since adoption of the EVMP. While elk have been studied in RMNP for decades, the drivers of moose expansion and resulting consequences for RMNP ecosystems are less well understood. Therefore, it's important that the results of ongoing efforts by RMNP and outside collaborators to census moose and characterize their spatial distribution and seasonal dynamics be incorporated into the implementation of the EVMP.

More extensive sampling in the Kawuneeche Valley and Wild Basin may prove especially valuable through time. This may be most efficiently obtained through partnership with other long-term programs such as conducted by the ROMN that have sentinel site focused monitoring efforts in both these areas (Schweiger et al. 2015). Because of distinct ecological histories, patterns of ungulate use, and high intrinsic value to RMNP, these valleys are both worthy of monitoring on their own as well as providing benchmarks for comparison of trends in sites included in the EVMP. Wild Basin is the most structurally diverse and highly functioning montane willow area in RMNP. In addition to dense and tall willows, anecdotal evidence suggests it has continuously supported beaver while their populations have declined in the winter range and Kawuneeche Valley. Monitoring here is essential as moose populations in Wild Basin have been increasing.

The density of EVMP sampling in the Kawuneeche Valley is inadequate to capture the spatial variability in riparian structure and condition. Conditions outside research ungulate exclosures show a decline in condition compared to areas inside exclosures. Additional effort should be directed towards increasing the number of sampling locations within the Kawuneeche Valley, perhaps in partnership with the ROMN and their 180 long term monitoring sites across ROMO. In addition to the habitat management exclosure erected in 2011, three existing research exclosures that were established in 1997, also could provide valuable information on browsing impacts on willow in the Kawuneeche Valley where moose have been present for several decades. Continuing to maintain the fences on these exclosures, as well as adding some monitoring plots inside the fences, may provide important information on condition of willow in this region in the absence of moose and elk browsing. This information will supplement the findings of the role of exclosures in this valley.

The park is a currently a partner in the Kawuneeche Valley Ecosystem Restoration Collaborative. This group of federal, state, and local agencies and non-profits is in the planning stages of restoration

actions in the Kawuneeche Valley that could include the use of exclosures to facilitate willow growth. The EVMP's vegetation monitoring data should be used to inform design. The EVMP's methods could also be incorporated into monitoring the trajectory of this project's restoration efforts.

Inconsistencies in the collection of plot-level moose and beaver presence made displaying temporal trends in beaver and moose presence challenging. While the monitoring protocol was not originally designed to be utilized in this way, it offers an opportunity to document changes in animal presence over time. During future data collection efforts, presence of beaver, moose and elk should be consistently documented at the plot level.

Management exclosures constructed to limit elk browsing have helped with recovery in some portions of the site as evidenced in EVMP data, but full recovery of willow community structure and ecological processes like new willow establishment and beaver occupancy will likely take many years. Large herbivores like elk and moose remove catkin-producing stems, reducing seed production and dispersal with potential consequences for riparian willow establishment dynamics (Gage and Cooper 2005; Cooper et al. 2006). Although not measured in the EVMP, seed production may have increased in fenced areas with taller unbrowsed willow stems. Areas not protected by fencing will take longer to reach desired future conditions laid out in the EVMP, or may not reach those desired conditions at all, if the limited progress documented in these analyses hold into the future. Of the management techniques utilized, these analyses demonstrate that fencing is the most successful technique for increasing both willow height and cover, and aspen recruitment. Reductions in the population size of elk overwintering in RMNP also appear to be resulting in modest improvements to habitat outside of exclosures, but these improvements may be negated by the impacts of larger populations of migratory elk that linger on winter range during other seasons.

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Appendix: Statistical Approach

This appendix provides supporting information on statistical analyses of Elk Vegetation Management Plan (EVMP) data collected through the 2018 sampling season in the NPS NRR Report “Vegetation Response to Elk Population and Habitat Management in Rocky Mountain National Park—Analysis of Elk and Vegetation Management Plan monitoring data: 2008–2018.” Separate analyses were run for plots established in aspen, willow and upland communities and for the elk winter range (core and noncore areas) and the Kawuneeche Valley. For information on sampling and the broader analysis and interpretation of the results, refer to the body of the report, past analyses (Zeigenfuss and Johnson 2015), and the original EVMP monitoring plan (Zeigenfuss et al. 2011).

Statistical analyses were aimed at answering three main questions. First, is there evidence of trends in key variables since the initiation of monitoring over the entire winter range? Secondly, how do variables of interest differ among management subgroups (e.g., core winter range vs. noncore winter range, fenced vs. unfenced plots, burned vs. unburned plots)? Lastly, is there evidence of similar trends among management subgroups over time?

To provide inferences on differences between time periods, range type, and fencing/burning subgroups, we fit Bayesian Generalized Linear Models to core winter range, noncore winter range, and Kawuneeche Valley subsets of the data using the R statistical software (R version 4.0.3). Random effects models with plots treated as random factors were developed using the ‘stan_glmmer’ function in the rstanarm package (Goodrich et al. 2020; Brilleman 2018). In addition, models were fit to particular subgroups of data. For aspen data, these included short and tall saplings; for willow and upland data, different groups of species (e.g., all willows, all shrubs, etc.) were separately analyzed.

Bayesian estimation was performed via the Hamiltonian Markov chain Monte Carlo (MCMC) algorithm, a method for numerically sampling from a probability distribution, adding independent priors specific to the data type being modeled on regression coefficients. Four chains with 8000 iterations and a burn-in of 1000 iterations were run and convergence evaluated using graphical methods and MCMC diagnostic statistics. Burn-in refers to discarding samples at the initial start of the MCMC chain and is intended to give the algorithm time to reach its equilibrium distribution. Weakly informed priors were used for different response types (e.g., willow height, aspen stem count) and detailed below.

Continuous response variables (e.g., willow height) were fit with a gamma distribution, which restricts possible responses to positive real numbers. Aspen stem counts were modeled as Poisson processes, while proportion data (e.g., cover) were modeled using a beta distribution (Ferrari and Cribari-Neto 2004), which constrains values from 0 to 1. Additional model details and diagnostic plots are found below.

Functions in the BayestestR package in R were used to describe the posterior distributions of parameters from Bayesian models. We report measures of centrality, uncertainty, effect existence, effect significance, and effect size. The median of the posterior distribution was used as the index of centrality; in probabilistic terms, this means that there is 50% probability that the effect is higher or

lower. We used the 95% Highest Density Interval (HDI) credible interval to evaluate uncertainty in parameter estimates.

Assessments of effect significance were based on a threshold of 0.05 of the standard deviation (SD) of the outcome variable and a threshold of $0.3 \cdot \text{SD}$ of an outcome variable was used to assess whether effect size was large. Note, both thresholds are inherently subjective (as is the $\alpha=0.05$ typically used in the frequentist null hypothesis testing). Differences between factors (e.g., year, fenced/unfenced) were compared using functions in the ‘emmeans’ R package (Lenth 2021). More details on statistical models are provided in appendices.

Bayesian models view a model parameter θ as a [random variable]. In contrast, frequentist models treat model parameters as unknown *constant*. Rather than estimating an unknown constant, Bayesian modeling focuses on an unknown distribution of parameter values.

Using *Bayes’ Law*, model parameters are estimated:

$$P(\theta|y, X) = \frac{P(y|\theta, X) \cdot P(\theta)}{P(y)} \propto P(y|\theta, X) \cdot P(\theta).$$

Where:

- X : The observed data
- y : The outcomes
- $P(\theta|y, X)$: Our view of the possible value of our model parameters after seeing the data
- $P(y|\theta, X)$: The likelihood of seeing the current outcome conditioned on the data in hand and a specific parameter value
- $P(\theta)$: Our view of the possible values of our model parameters before observing any actual data, i.e., the *prior*
- $P(y)$: The unconditional probability of the outcome, i.e., the probability after considering all possible parameter.

The objective is to estimate the posterior density of parameter $P(\theta|y, X)$, typically described in terms of a point estimate of the expected value of posterior density (e.g., the median of the distribution) and a measure of the variance. To solve for $P(\theta|y, X)$ analytically, the data likelihood $P(y|\theta, X)$, prior $P(\theta)$, and marginal likelihood of outcome $P(y)$ are needed, but because there is usually no closed-form analytic solution, Markov chain Monte Carlo (MCMC) methods are used to numerically solve the posterior density by directly generating random draws of parameters.

Aspen Data

For analyses of aspen count data, a random effects model was used for making inferences in means:

$$\mu_i = \exp(\alpha + \beta_1 x_{1,i} + \beta_2 x_{2,i} + \beta_3 x_{3,i} + \beta_4 x_{4,i} + \beta_5 x_{5,i} + \gamma_i)$$

$$\gamma_i \sim \text{normal}(0, \sigma^2)$$

$$y_i \sim \text{poisson}(\mu_i)$$

where y_i is the count of aspen stems in the i th plot; α is the model intercept; $x_{1,i}$ is a factor (indicator variable) coded numerically as 1 when the observation in plot i was from 2013 and 0 otherwise; $x_{2,i}$ is a factor coded numerically as 1 when the observation in plot i was measured in 2018 and 0 otherwise; $x_{3,i}$ is a factor coded numerically as 1 when the observation in plot i was fenced and 0 otherwise. Analyses included both burned and unburned plots. Additional models were developed explicitly including burning as a factor by adding an additional term, β_4, x_i coded numerically as 1 when the observation in plot i was burned and 0 otherwise.

For each of the specific models developed (e.g., short saplings in the combined winter range, tall saplings in the core winter range, etc.), the intercept $\exp(\alpha)$ represents the mean count of aspen in unfenced plots at baseline; $\exp(\beta_1)$ is the multiplicative change in mean counts that occurred during the time interval between the baseline year and 2013; $\exp(\beta_2)$ is the multiplicative change in mean counts that occurred during the time interval between the baseline year and 2018; $\exp(\beta_3)$ is the multiplicative change in mean counts that was caused by fencing; $\exp(\beta_4)$ is the multiplicative change in mean counts that was caused by burning.

For an observation y that is assumed to follow a Poisson distribution (e.g., aspen stem count data), the likelihood for one observation can be written as:

$$1/y! \lambda^y e^{-\lambda},$$

where $\lambda = E(y|\mathbf{x}) = g^{-1}(\eta)$ and $\eta = \alpha + \mathbf{x}^\top \boldsymbol{\beta}$ is a linear predictor. For the Poisson distribution it is also true that $\lambda = \text{Var}(y|\mathbf{x})$, i.e., the mean and variance are both λ . The rate parameter λ must be positive, so with a Poisson GLM, the *link* function g maps between the positive real numbers \mathbb{R}^+ (the support of λ) and the set of all real numbers \mathbb{R} . When applied to a linear predictor η with values in \mathbb{R} , the inverse link function $g^{-1}(\eta)$ returns a positive real number. See Gabry and Goodrich (2020) for details on rstanarm implementation.

The standard link function for a Poisson GLM is the log link $g(x) = \ln(x)$. With the log link, the inverse link function is the exponential function and the likelihood for a single observation is:

$$\frac{g^{-1}(\eta)^y}{y!} e^{-g^{-1}(\eta)} = \frac{e^{\eta y}}{y!} e^{-e^\eta}.$$

Prior distributions for $f(\alpha)$ and $f(\boldsymbol{\beta})$ were represented by normal distributions with a mean zero and a small standard deviation (scale)). With independent prior distributions, the joint posterior distribution for α and $\boldsymbol{\beta}$ in the Poisson model is proportional to the product of the priors and the N likelihood contributions, the posterior distribution drawn from when using MCMC:

$$f(\alpha, \boldsymbol{\beta}|\mathbf{y}) \propto f(\alpha) \times \prod_{k=1}^K f(\beta_k) \times \prod_{i=1}^N \frac{g^{-1}(\eta_i)^{y_i}}{y_i!} e^{-g^{-1}(\eta_i)}.$$

Willow and Upland Data

Height

Bayesian repeated measures analyses of shrub height were fit separately for combined core and noncore winter range, core winter range, and noncore winter range plots. Separate models were also fit to different groups of species. Shrub height models were fit using weekly informative priors and a gamma distribution with the “stan_glmr” function in the “rstanarm” package (Brilleman 2018; Goodrich 2020). Gamma regression is commonly used when the response variable is continuous and positive. A random effects model was used for making inferences in mean height:

$$\mu_i = \exp(\alpha + \beta_1 x_{1,i} + \beta_2 x_{2,i} + \beta_3 x_{3,i} + \beta_4 x_1 x_3 + \beta_5 x_2 x_3 + \epsilon_i)$$

$$\epsilon_i \sim \text{normal}(0, \sigma^2)$$

$$y_i \sim \text{gamma}(\mu, \phi)$$

where y_i is shrub height in the i th plot; α is the model intercept; $x_{1,i}$ is a factor (indicator variable) coded numerically as 1 when the observation in plot i was from 2013 and 0 otherwise; $x_{2,i}$ is a factor coded numerically as 1 when the observation in plot i was measured in 2018 and 0 otherwise; $x_{3,i}$ is a factor coded numerically as 1 when the observation in plot i was fenced and 0 otherwise. ϕ represents a shape parameter. Separate models were developed to explicitly include burning as a factor by adding an additional term, $x_{4,i}$, coded numerically as 1 when the observation in plot i was burned and 0 otherwise.

For each of the specific models developed (e.g., all willows in the combined winter range, all willows in the core winter range, etc.), the intercept $\log(\alpha)$ represents the mean height of shrubs in unfenced plots at baseline; $\log(\beta_1)$ is the multiplicative change in height that occurred during the time interval between the baseline year and 2013; $\log(\beta_2)$ is the multiplicative change in height that occurred during the time interval between the baseline year and 2018; $\log(\beta_3)$ is the multiplicative change in mean height that was caused by fencing. In models explicitly incorporating burning as a factor, an additional main effect term was added representing the multiplicative change in mean height caused by burning. Interaction terms between main factors were also included.

For a simple GLM, the likelihood for one observation is a conditionally normal probability density function (Gabry and Goodrich 2020a):

$$\frac{1}{\sigma\sqrt{2\pi}} e^{-\frac{1}{2}\left(\frac{y-\mu}{\sigma}\right)^2},$$

where $\mu = \alpha + \mathbf{x}^\top \boldsymbol{\beta}$ is a linear predictor and σ is the standard deviation of the error in predicting the outcome, y . A linear predictor $\eta = \alpha + \mathbf{x}^\top \boldsymbol{\beta}$ is related to the conditional mean of the outcome via a link function g , which for the gamma modeling of height was the log link function.

Prior distributions for $f(\alpha)$ and $f(\boldsymbol{\beta})$ were represented by normal distributions with a mean zero and a small standard deviation (scale)). The joint posterior distribution for α and $\boldsymbol{\beta}$ is proportional to the product of the priors and the N likelihood contributions (Gabry and Goodrich 2020a):

$$f(\boldsymbol{\beta}|\mathbf{y}) \propto f(\alpha) \times \prod_{k=1}^K f(\beta_k) \times \prod_{i=1}^N f(y_i|\eta_i),$$

where \mathbf{X} is the matrix of predictors and η the linear predictor, i.e., the posterior distribution that `stan_glmr` draws from using MCMC.

Cover

Bayesian random effects models of shrub cover were fit separately for combined core and noncore winter range, core winter range, and noncore winter range plots and to different groups of species (e.g., all willow species, all shrub species, etc.). Cover values are limited to values between 0 and 1, so regression models were based on the beta distribution commonly used to model plant cover (Damgaard and Irvine 2019). The log-link function was used with coefficient values and were interpreted as multiplicative changes in mean cover. The general model used follows:

$$\begin{aligned} \mu_i &= \log(\alpha + \beta_1 x_{1,i} + \beta_2 x_{2,i} + \beta_3 x_{3,i} + \beta_4 x_1 x_3 + \beta_5 x_2 x_3 + \epsilon_i) \\ \epsilon_i &\sim \text{normal}(0, \sigma^2) \\ y_i &\sim \text{beta}(\mu_i) \end{aligned}$$

where y_i is cover in the i th plot; α is the model intercept; $x_{1,i}$ is a factor (indicator variable) coded numerically as 1 when the observation in plot i was from 2013 and 0 otherwise; $x_{2,i}$ is a factor coded numerically as 1 when the observation in plot i was measured in 2018 and 0 otherwise; $x_{3,i}$ is a factor coded numerically as 1 when the observation in plot i was fenced and 0 otherwise. Separate models were developed to explicitly include burning as a factor by adding an additional term, $x_{4,i}$ coded numerically as 1 when the observation in plot i was burned and 0 otherwise. Indicator variables and regression coefficients not relevant to a particular model (e.g., subsets of data like noncore range plots lacking fencing) were not part of models.

For each model (e.g., all willows in the combined winter range, all willows in the core winter range, etc.), the intercept $\log(\alpha)$ represents mean cover in unfenced plots at baseline; $\log(\beta_1)$ is the multiplicative change in cover that occurred during the time interval between the baseline year and 2013; $\log(\beta_2)$ is the multiplicative change in cover that occurred during the time interval between the baseline year and 2018; $\log(\beta_3)$ is the multiplicative change in mean cover that was caused by fencing. In models explicitly incorporating burning as a factor, an additional β term was added for the main effect representing the multiplicative change in mean cover caused by burning. Interaction terms between main factors were also included (e.g., β_4).

The likelihood for one observation is a conditionally normal probability density function (Gabry and Goodrich 2020b):

$$\frac{1}{\sigma\sqrt{2\pi}} e^{-\frac{1}{2}\left(\frac{y-\mu}{\sigma}\right)^2},$$

where $\mu = \alpha + \mathbf{x}^\top \boldsymbol{\beta}$ is a linear predictor and σ is the standard deviation of the error in predicting the outcome, y . A linear predictor $\eta = \alpha + \mathbf{x}^\top \boldsymbol{\beta}$ is related to the conditional mean of the outcome via a link function g , which for the gamma modeling of height was the log link function.

Prior distributions for $f(\alpha)$ and $f(\boldsymbol{\beta})$ (intercept and vector of regression coefficients) were represented by normal distributions with a mean zero and a small standard deviation (scale)). The joint posterior distribution for α and $\boldsymbol{\beta}$ is proportional to the product of the priors and the N likelihood contributions (Gabry and Goodrich 2020c):

$$f(\boldsymbol{\beta}|\mathbf{y}) \propto f(\alpha) \times \prod_{k=1}^K f(\beta_k) \times \prod_{i=1}^N f(y_i|\eta_i),$$

where \mathbf{X} is the matrix of predictors and η the linear predictor, i.e., the posterior distribution that `stan_glm` draws from using MCMC.

Cover data were modeled using the beta distribution with the likelihood (Gabry and Goodrich 2020c):

$$f(y_i|a, b) = \frac{y_i^{(a-1)}(1-y_i)^{(b-1)}}{B(a, b)}$$

where $B(\cdot)$ is the beta function. The shape parameters for the distribution are a and b and enter into the model according to the following transformations,

$$a = \mu \cdot \phi$$

$$b = (1 - \mu) \cdot \phi$$

If $g_1(\cdot)$ is the link function, the specification of the shape parameters, $\mu = g_1^{-1}(\mathbf{X}\boldsymbol{\beta})$, where \mathbf{X} is a $N \times K$ dimensional matrix of predictors, and $\boldsymbol{\beta}$, is a K dimensional vector of parameters associated with each predictor. ϕ is a scalar parameter. With the shape parameter values included, the likelihood takes the form (Gabry and Goodrich 2020c):

$$f(y_i|\mu, \phi) = \frac{y_i^{(\mu\phi-1)}(1-y_i)^{((1-\mu)\phi-1)}}{B(\mu\phi, (1-\mu)\phi)}$$

Analysis requires specifying prior distributions $f(\boldsymbol{\beta})$ and $f(\phi)$ for the vector of regression coefficients and ϕ . When modeling ϕ with a linear predictor a full Bayesian analysis requires specifying the prior distributions $f(\boldsymbol{\beta})$ and $f(\boldsymbol{\gamma})$.

With a single set of explanatory variables, the posterior distribution of $\boldsymbol{\beta}$ and ϕ is proportional to the product of the likelihood contributions, the K priors on the β_k parameters, and ϕ (Gabry and Goodrich 2020c),

$$f(\boldsymbol{\beta}, \boldsymbol{\phi} | \mathbf{y}, \mathbf{X}) \propto \prod_{i=1}^N f(y_i | a, b) \times \prod_{k=1}^K f(\beta_k) \times f(\boldsymbol{\phi})$$

For two sets of explanatory variables (e.g., year and fencing), the posterior distribution of $\boldsymbol{\beta}$ and $\boldsymbol{\gamma}$ is proportional to the product of the likelihood contribution, the K priors on the β_k parameters, and the J priors on the γ_j parameters (Gabry and Goodrich 2020c),

$$f(\boldsymbol{\beta}, \boldsymbol{\gamma} | \mathbf{y}, \mathbf{X}) \propto \prod_{i=1}^N f(y_i | a, b) \times \prod_{k=1}^K f(\beta_k) \times \prod_{j=1}^J f(\gamma_j)$$

Probability of Direction (PD)

The Probability of Direction (PD) was used to represent certainty associated with the most probable direction (positive or negative) of the effect (e.g., year, fencing). The PD is correlated with the frequentist p-value, with a two-sided p-value of respectively .1, .05, and .01 approximated by a PD of 95%, 97.5%, and 99.5% (Makowski, Ben-Shachar, and Lüdtke 2019).

Additional supplemental materials, containing R code, results and figures not presented in the report are available by request from the Rocky Mountain National Park Resource Stewardship Division.

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The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

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