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Monitoring of Vegetation Response to Elk Population and Habitat Management in Rocky Mountain National Park, 2008–14

By Linda C. Zeigenfuss and Therese L. Johnson

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Cover: Elk habitat improvement exclosures in Beaver Meadows, Rocky Mountain National Park, Colorado.

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Conversion Factors

International System of Units to Inch/Pound

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
	Area	
square meter (m^2)	0.0002471	acre
hectare (ha)	2.471	acre
square hectometer (hm^2)	2.471	acre
square kilometer (km^2)	247.1	acre
square meter (m^2)	10.76	square foot (ft^2)
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as °F = (1.8 × °C) + 32.

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as °C = (°F – 32) / 1.8.

Datum

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Altitude, as used in this report, refers to distance above the vertical datum.

Abbreviations

dbh	diameter at breast height
EVMP	elk and vegetation management plan
R^2	coefficient of determination
USGS	U.S. Geological Survey
yr	year

Monitoring of Vegetation Response to Elk Population and Habitat Management in Rocky Mountain National Park, 2008–14

By Linda C. Zeigenfuss and Therese L. Johnson[1]

Abstract

Since 2008, Rocky Mountain National Park in Colorado has been implementing an elk and vegetation management plan with the goal of managing elk populations and their habitats to improve the condition of key vegetation communities on elk winter range. Management actions that have been taken thus far include small reductions in the elk herd through culling of animals and temporary fencing of large areas of willow and aspen habitat to protect them from elk browsing. As part of the park's elk and vegetation management plan (EVMP), a monitoring program was established to assess effectiveness of management actions in achieving vegetation goals. We collected data to monitor offtake (consumption) of upland herbaceous plants and willow annually from 2008 to 2014 and to assess aspen stand structure and regeneration and willow cover and height in 2013, 5 years after plan implementation. Loss of many willow and a few aspen monitoring sites to a fire in late 2012 complicated data collection and interpretation of results but will provide opportunities to observe habitat recovery following fire and in the presence and absence of elk herbivory, which will offer important insights into the use of prescribed fire as an additional management tool in these habitats.

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. By 2013, aspen saplings (stems less than or equal to 2.5 meter height) were recruiting on 29 percent of sampled sites, an increase from 13 percent of sites at baseline, but this was mainly due to growth inside fences. Upland herbaceous offtake dropped below baseline levels (61 percent) on both core and noncore winter range in 2010–14. Less than 10 percent of the upland areas had intense herbivory (greater than 85 percent offtake), and less than 30 percent of the landscape had offtake greater than 70 percent after 2009. Offtake levels in 2013 and 2014 indicated an increase in grazing pressure on upland sites compared to 2010–12 levels, but this change may have been in response to loss of large patches of both herbaceous and woody forage in Moraine Park following the 2012 Fern Lake Fire. Winter willow offtake remained steady from 2009 to 2014, and although there were no substantial increases in offtake, there were also no consistent declines. Winter-range willow offtake was below the baseline level of 35 percent only in 2013 and 2014. Willow heights have stayed at or above baseline levels of 0.9 meter. Average heights of willow increased compared to baseline measures within fenced habitat on the core winter range and on noncore (all unfenced) winter range. Willow cover increased at least 75 percent compared to baseline within core winter-range fenced areas and roughly 25 percent in noncore winter range. Overall, during the first 5 years of implementation, the EVMP at Rocky Mountain

[1] Rocky Mountain National Park

National Park seems to be making steady progress toward the vegetation objectives set out by the EVMP. Habitat fencing has been the most effective means of improving aspen and willow habitat conditions.

Introduction

Rocky Mountain National Park encompasses nearly 108,000 hectares (ha) of high-elevation forest, shrublands, meadows, and alpine tundra and rocklands in north-central Colorado. The park supports numerous species of wildlife, including several large ungulate species. The most abundant ungulate species in the park is *Cervus elaphus* (elk). In recent years, there has been growing concern about the condition of vegetation in the park and conflicts between elk and humans, both inside and outside the park. In particular, condition of *Populus tremuloides* (quaking aspen) and *Salix* spp. (willow) habitats on the winter range of Rocky Mountain National Park have declined.

Many years of research indicated that high elk densities in Rocky Mountain National Park were resulting in the complete loss of aspen clones or reduction of many aspen on core winter-range areas to a shrub-like state (Baker and others, 1997; Suzuki and others, 1999; Weisberg and Coughenour, 2003). Elk browsing was determined to be suppressing growth of or killing all young aspen (those less than 2.5 meters [m] in height—also called "suckers" or "saplings") on the core elk winter range and in some parts of the Kawuneeche Valley (Olmsted, 1979; Baker and others, 1997; Zeigenfuss and others, 2008). As a result, aspen stand regeneration on the elk winter range was limited with few or no suckers growing large enough to be recruited into the tree canopy in recent decades (Suzuki and others, 1999; Binkley, 2008), resulting in overmature, deteriorating aspen stands with no small- or medium-diameter trees (Baker and others, 1997; Binkley, 2008). These stands would likely be permanently lost if the level of elk herbivory remained high, although it is difficult to predict when this would happen (National Park Service, 2007). Research indicated that management actions that enhance aspen recruitment could help sustain and preserve aspen stands in the park over the long term (Kaye and others, 2005).

Elk were also suppressing the growth of willow plants, both in height and areal cover, and tall willow stands were being converted to short willow (Peinetti and others, 2002; Zeigenfuss and others, 2002). Willow is the dominant woody shrub on almost all wet meadow or riparian areas in Rocky Mountain National Park. Heavy browsing inhibited the ability of many winter-range willows to reproduce because few willow plants on the primary elk winter range produced seed. Suitable seedling establishment sites were limited as well, and seedling survival was poor due to downcutting of stream channels and associated lowering of the water table as well as browsing (Gage and Cooper, 2005). Elk were consuming herbaceous plants at extremely high rates on the elk winter range and potentially resulting in the alteration of herbaceous plant communities on the elk range. Annual herbaceous consumption rates on willow and upland shrub communities on the elk winter range averaged 55 to 60 percent, in the 1990s (Singer and others, 2002).

In response to these concerns, Rocky Mountain National Park developed an elk and vegetation management plan (EVMP) to evaluate the effects of a range of alternatives for managing elk and vegetation in the park. The purpose of the EVMP is to guide management actions in the park during a 20-year (yr) time period to reduce the impacts of elk on vegetation and restore, to the extent possible, the natural range of variability in the elk population and affected plant communities (National Park Service, 2007), which includes the following:

- prevention of loss of aspen clones within high elk-use areas;
- restoration and maintenance of sustainable montane riparian willow as indicated by (1) increase of montane riparian willow cover within suitable willow habitat on the primary winter range;

and (2) maintenance or improvement of the condition of riparian willow on the primary summer range; and

- reduction of the level of elk grazing on herbaceous vegetation.

The plan outlines the desired future condition for each of these identified vegetation types (table 1). The management alternative that was selected relies on a variety of conservation tools including temporary fencing, nonlethal redistribution of elk, use of various vegetation-restoration techniques, and lethal reduction of elk (culling).

Purpose and Scope

The EVMP incorporates the principles of adaptive management to assess the effectiveness of management actions. Use of adaptive management in the EVMP means that Rocky Mountain National Park managers will adjust management actions as needed to successfully achieve the EVMP's objectives. Determination of whether vegetation objectives are being achieved requires monitoring and evaluation of target vegetation communities. From 2006 to 2009, an EVMP monitoring program (Zeigenfuss and others, 2011) was established on the elk winter range of Rocky Mountain National Park to monitor the effects of elk management actions (through both population management and habitat management) in key vegetation communities (aspen, willow, upland). This program called for monitoring of (1) vegetation offtake by herbivores in willow (shrub offtake) and upland (herbaceous offtake) communities through annual sampling of a subset of sites, (2) shrub height and cover in willow every 5 yrs through resampling of all sites, and (3) stem density and stem size distribution in aspen every 5 yrs through resampling of all sites. Because of logistical constraints, all monitoring sites could not be established in the same year; however, the bulk of baseline sampling had taken place by 2008 and fence construction began in the fall of 2008, thus 2013 was the fifth year after plan implementation.

The purpose of this report is to provide an analysis of the first 5 yrs of monitoring data collected in association with the EVMP. This analysis will be used as a means to assess progress toward the vegetation goals. We examined the changes that have taken place during this first phase of elk and vegetation management at Rocky Mountain National Park. Additional factors that may be influencing the type and degree of vegetation change on the elk winter range are also addressed. The EVMP monitoring plan, data collection, and this report focus on evaluation of the elk winter range because that has been the focus of plan implementation. For the Kawuneeche Valley, which is primarily summer range, we provide baseline information for a limited number of willow sites established in 2012, 3 yrs of willow offtake sampling, and data from aspen sites 5 yrs after baseline data collection.

Table 1. Vegetation types and their desired future conditions, indicator variables, and thresholds to be used in implementing the Rocky Mountain National Park elk and vegetation management plan.

[table taken from Zeigenfuss and others (2011); ~, about; <, less than; cm, centimeter; dbh, diameter at breast height; m, meter; ≤, less than or equal to; >, greater than]

Vegetation category	Desired future condition	Indicator	Thresholds/objectives
Aspen	At least 45 percent of aspen across the winter range regenerating. Distribution of stem dbh reflects many (~75 percent) small-diameter stems, some (~20 percent) medium-diameter stems, and few (~5 percent) large-diameter stems.	Stem density by height and diameter class	Progressive increase in aspen recruitment above the baseline level of 13 percent (presence of stems <2 cm dbh reaching 1.5–2.5 m tall). Progressive shift in the distribution of stem sizes toward the desired future condition.
Riparian montane willow	At least 31 percent willow cover within suitable willow habitat across the winter range. Average willow height of at least 1.1 m.	Willow consumption	No net increase in annual willow offtake across the winter range above the baseline level of 35 percent.
		Cover	Progressive increase in willow cover across the winter range above the baseline level of 21 percent.
		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 47 percent, with ≤ 25 percent of sites with offtake > 70 percent and ≤ 10 percent of sites with offtake > 85 percent.

Study Area

Most of the Rocky Mountain National Park EVMP monitoring sites are located within the elk winter range on the east side of Rocky Mountain National Park (fig. 1; Zeigenfuss and others, 2011). This winter range encompasses approximately 10,000 ha in five major valleys in the upper montane zone along the park boundary near the town of Estes Park, Colorado. The elk winter range includes areas where elk concentrate during winter (Moraine Park, Beaver Meadows, Horseshoe Park) that are referred to as the "core" elk winter range and other areas that elk use to a lesser degree, referred to as the "noncore" winter range (National Park Service, 2007; fig. 1). Elevation in the area ranges from 2,400 to 2,800 m. Valley bottom vegetation includes *Carex* spp. (sedges), grasses, and riparian shrubs (willow, *Betula* spp. [birch], *Alnus incana* [gray alder]). Slopes contain *Pinus ponderosa* (ponderosa pine)/shrub, mixed conifer (ponderosa pine–*Pseudotsuga menziesii* [Douglas-fir]), *Pinus contorta* (lodgepole pine), aspen, and upland grass/shrub (predominantly *Purshia tridentata* [antelope bitterbrush]) vegetation types. The Kawuneeche Valley is an area of about (~) 825 ha on the western boundary of the park running north to south along the headwaters of the Colorado River (fig. 1). Elevations in the Kawuneeche Valley range from 2,650 m to 2,800 m. The valley is dominated by wet meadows (sedge species, *Calamagrostis canadensis* (bluejoint reedgrass), *Deschampsia cespitosa* (tufted hairgrass), and riparian shrubs (willow, *Betula glandulosa* [resin birch], gray alder).

Elk were extirpated, or nearly so, from Rocky Mountain National Park by human exploitation in the late 1800s but were then reintroduced in 1913 and 1914. The population steadily increased until it reached an estimated 1,000 animals within the park boundaries in 1944 (Packard, 1947). Elk in the park were artificially reduced from 1943 to 1968, but starting in 1968, they were no longer controlled within the park's boundaries, as a result of a change to a U.S. National Park Service management paradigm know as natural regulation (National Park Service, 2007). Elk steadily increased in the park after 1968 and also in the town of Estes Park after the town area was recolonized in the late 1970s, reaching a population of roughly 1,000 animals on the park winter range and 2,000 in the Estes Valley by the late 1990s (Lubow and others, 2002). Since 2001, this population has been declining and by 2008 numbered roughly 650 animals wintering in the park (N.T. Hobbs, Senior Research Scientist, Natural Resources Ecology Lab, Colorado State University, unpub. data, 2014). In addition to the population that winters in the park from November through March, the Rocky Mountain National Park winter range is used in the spring, summer, and fall by elk that migrate between summer range in the park and lower-elevation winter range in the Estes Valley and Front Range foothills (Bear 1989, Larkins 1997). In addition to elk, mule deer (*Odocoileus hemionus*) use elk winter-range areas in the park year-round, tending to concentrate mainly in upland shrub communities. Moose (*Alces alces*) were introduced to North Park, Colo., northwest of Rocky Mountain National Park in the late 1970s and by 1980 had appeared in the Kawuneeche Valley. There is no recent population estimate of moose in the Rocky Mountain National Park; however, Dungan (2007) estimated approximately 100 moose inhabited the west side of the park in the Colorado River drainage during summer 2003, with roughly one-fourth of that population residing in the Kawuneeche Valley. Moose have begun to appear on the elk winter range on the east side of Rocky Mountain National Park in the last decade, though numbers appear to be low. Both elk and moose inhabit the Kawuneeche Valley year-round but are most common in spring through fall.

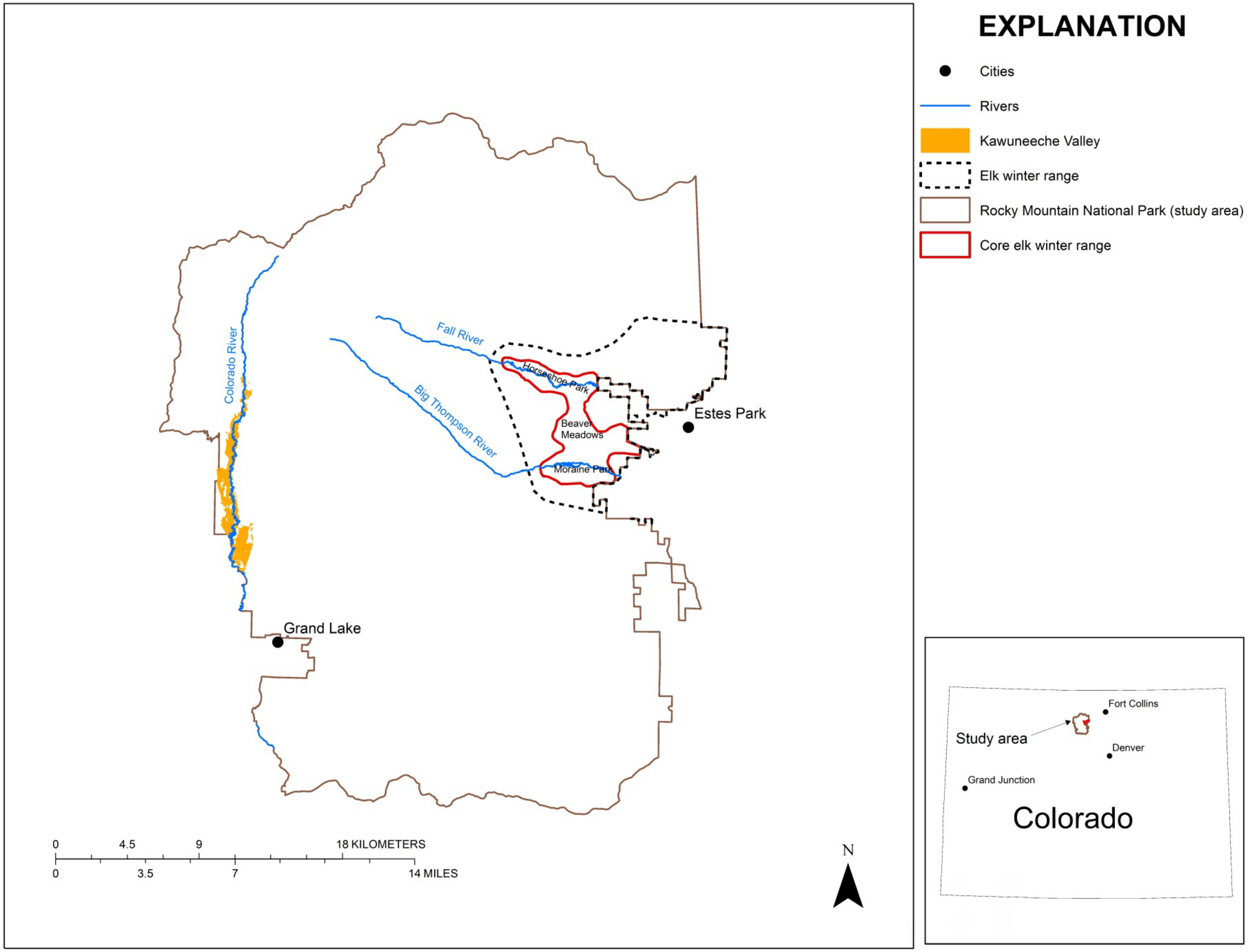

Figure 1. The elk winter range and Kawuneeche Valley of Rocky Mountain National Park, Colorado.

From 2008 to 2013, the park constructed temporary elk-proof fences around patches of key aspen and willow habitat to facilitate vegetation restoration. By 2013, when data for this assessment were collected, approximately 68.5 ha of fenced willow areas and 17 ha of fenced aspen stands had been established on the core elk winter range (fig. 2). In 2011, the park installed an experimental fence around willow in the Kawuneeche Valley, excluding elk and moose from a 6.5-ha area. An additional 1.4 ha of willow habitat and 1 ha of aspen habitat on the winter range, as well as 0.6 ha of Kawuneeche Valley willow, were already fenced, mostly in disjunct small patches, as part of earlier studies and restoration efforts.

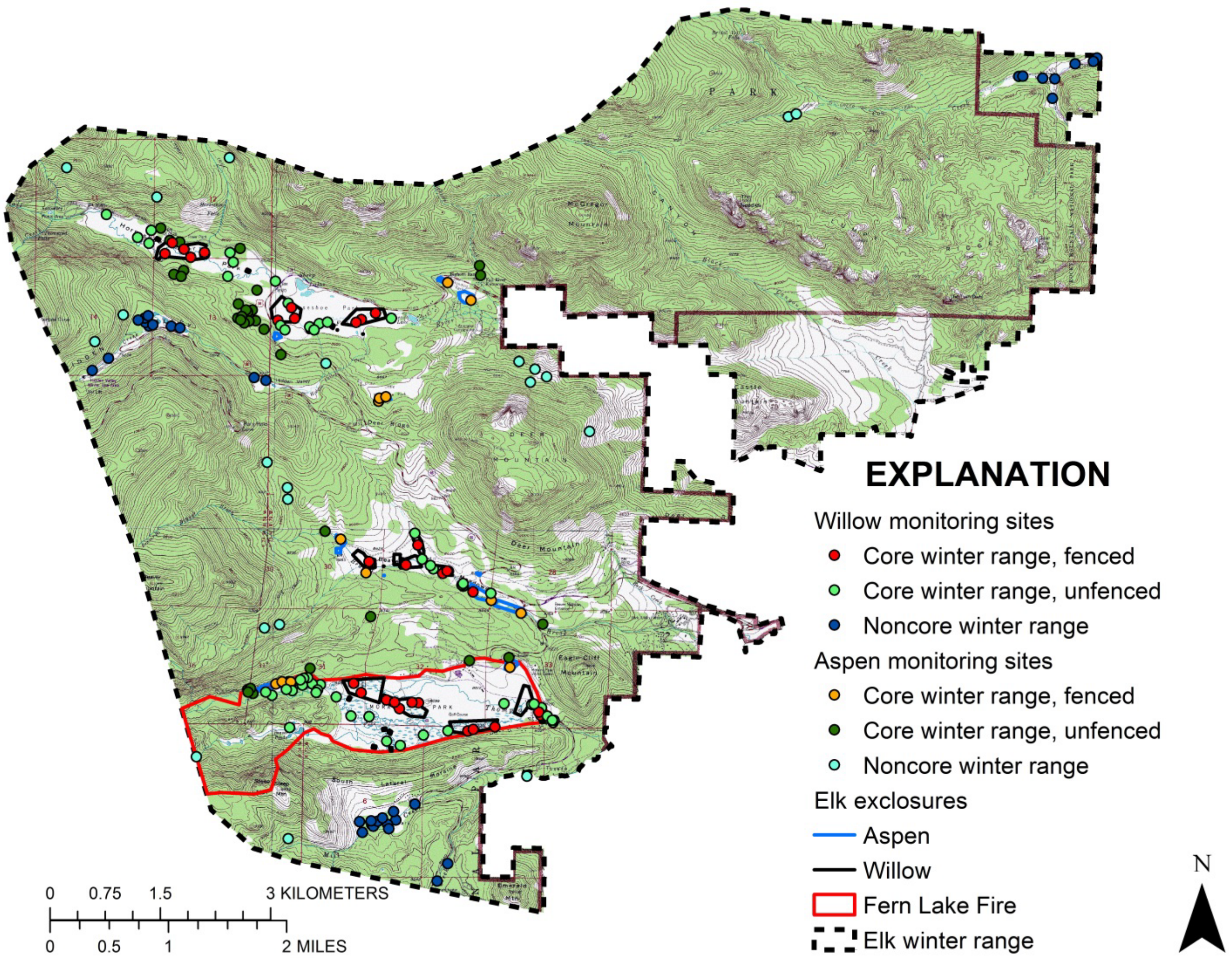

Figure 2. Location of aspen and willow monitoring sites, existing elk exclosures, and the 2012 Fern Lake Fire on the Rocky Mountain National Park elk winter range, Colorado, as of January 2014.

Methods

The EVMP monitoring program was developed using a sampling design that stratified vegetation communities into elk winter-range zones of fenced core winter range, unfenced (open to elk browsing and grazing) core winter range, and noncore winter range (all unfenced). This design was intended to allow assessment of vegetation response to fencing and detection of shifts in habitat use from core to noncore areas (Zeigenfuss and others, 2011).

Twenty sites per elk winter-range stratum (core and noncore winter range) were determined to be adequate to assess the level of change indicated for both fenced and browsed aspen in the EVMP (Zeigenfuss and others, 2011). A sample size of 35 sites for willow monitoring in each elk winter-range zone was needed to adequately measure browsed willow (Zeigenfuss and others, 2011). Nearly one-half the area of core winter-range willow habitat was planned for fencing, and due to the large sizes of these fenced enclosures, several monitoring sites were located in each fenced area; therefore, the sample size in fenced core winter-range willow was limited to 20 sites. A sample size of 25 sites per elk winter-range stratum was determined to be adequate to predict mean values of upland herbaceous offtake within 20 percent of the true mean 90 percent of the time.

Zeigenfuss and others (2011) used a stratified, random sampling design to select potential monitoring sites. Using ArcGIS® 9.2 software program, Zeigenfuss and others (2011) selected the vegetation communities of interest and clipped them to the winter-range boundary. Those vegetation polygons that fell within the core winter range were further separated from the entire winter range to create maps of core and noncore winter-range vegetation communities. Maps of potential fence locations were used to clip vegetation polygons from the core winter-range map. The final results were three maps each for aspen and willow types (noncore winter range (all unfenced); fenced, core winter range; and unfenced core winter range) and two maps for upland vegetation types (core and noncore winter range). Zeigenfuss and others (2011) generated random points in each vegetation type and elk winter-range zone. A minimum distance of 250 m between upland sites was imposed to allow movement of grazing cages over a large enough area to prevent site overlap while also preventing resampling within the same site over a short timeframe. Since willow and aspen monitoring plots were fixed in size, no minimum distance between plots was used, but plot boundaries could not overlap each other or be in close proximity to fences. Not all sites that appeared appropriate on the map were determined to be suitable upon site visit, in which case the site was replaced with another randomly selected site.

The temporal sampling design varied across metrics and community type. Offtake indicators were measured annually due to the high degree of interannual variability related to factors such as weather and ungulate population size. To reduce costs related to sampling all plots every year, a split-panel, serially-alternating design with consecutive year revisits was employed (Urquhart and Kincaid, 1999). The 20-yr monitoring period (2009–28) was divided into five panels, such that each site is visited for offtake measures twice within each 5-yr period. The total number of sites in each vegetation type and elk winter-range zone was divided by five to determine the number of sites assigned to any given panel.

The social nature of an ungulate that moves and feeds in herds (like elk) can result in large spatial heterogeneity in offtake on the landscape (with some locations being more heavily grazed due to number of animals concentrated in a group) in addition to heterogeneity in offtake resulting from the spatial distribution of desirable forage species. Therefore, the temporal aspects of this design were also balanced with the spatial distribution of sites by visually grouping the random sites into spatial “neighborhoods” of five sites (one for each panel) and randomly assigning them to one of the five panels. The “neighborhood” was defined as a group of sites that were spatially close together and linked by topography as much as possible.

Variables such as willow cover and structure, upland shrub cover, and aspen stem density tend to change more slowly so these were measured only every 5 yrs. All monitoring sites were revisited in 2013 to collect measurements associated with these variables.

Vegetation Data Collection

Field sampling methods followed protocols detailed in the park's EVMP monitoring plan (Zeigenfuss and others, 2011), but the protocols are briefly summarized here to provide a context for the results presented in this report. We collected data at the established plots according to the measurement timetables established in the protocols (table 2). Baseline data were collected 2006–9 (Zeigenfuss and others, 2011). Park staff were responsible for collecting all monitoring data as well as establishing and collecting baseline data for sites that were added after 2009. Annual visits to sample offtake from a subsample of willow and upland sites began in 2007 (upland) and 2009 (willow) and continued through 2014. Target sample sizes (table 2) were not always achieved due to a variety of circumstances including weather, inability to relocate sites, labor shortages, and oversight. Data collection dates varied due to snow cover and availability of personnel. Willow offtake measures were typically collected during the months of May and June; however, in 2010, data was collected on some sites as late as mid-July. All other variables (table 2) were measured during the year of site establishment and again during June through September 2013.

Table 2. Summary of data to be used in 5-yr analysis of EVMP monitoring in Rocky Mountain National Park.
[yr, year; m, meter; cm, centimeter]

Vegetation type	Measured variable		Major comparisons	Sampling interval	Original target sample size/interval	Units
Willow	Percent offtake	Winter range	Unfenced core and noncore winter range All yrs 2008–14 yr 1 to yr 6	Annually with one-half of sampled sites measured in consecutive years	14 core 14 noncore (all unfenced)	Current year production removed from 0-100 percent
		Kawuneeche Valley	yrs 2012–14	Annually	4 unfenced	
	Willow cover and height	Winter range	Fenced and unfenced core winter range Unfenced core and noncore winter range yr 1 and yr 5 Burned and unburned	Sampled once every 5 yrs	35 core, unfenced 20 core, fenced 35 noncore (all unfenced)	Cover (percent) from 0–100 percent Willow height (m)—maximum and average Cover and height of willow
		Kawuneeche Valley	Baseline	Annually	4 fenced 4 unfenced	
Aspen	Aspen density and height	Winter range	Fenced and unfenced core winter range Unfenced core and noncore winter range yr 1 and yr 5 Burned and unburned	Sampled once every 5 yrs	20 core, unfenced 20 core, fenced 20 noncore (all unfenced)	Stem size and density of aspen Sapling height (cm)
		Kawuneeche Valley	yr 1 and yr 5		8 (all unfenced)	
Upland	Percent offtake	Winter range	Unfenced core and noncore winter range All yrs 2008–14	Annually with one-half of sampled sites measured in consecutive years	10 core (all unfenced) 10 noncore (all unfenced)	Percent standing crop removed from 0–100 percent
	Shrub cover	Winter range	Unfenced core and noncore winter range yr 1 and yr 5	Sampled once every 5 yrs	25 core (all unfenced) 23 noncore (all unfenced)	Cover (percent) from 0–100 percent Cover by species

A fire in December 2012 burned through Moraine Park affecting 21 core willow monitoring sites, 9 core aspen sites, and 1 noncore aspen site (fig. 2). Of these sites, 17 willow (9 fenced, 8 unfenced) and 4 aspen (3 fenced, 1 unfenced) core sites were burned severely enough to require replacement. The burned plots will be maintained, however, to measure comparative growth inside and outside elk exclusion fences. New sites were established and measured in 2013, which led to changes in the sample sizes and design from the original monitoring plan (table 3). A few other site replacements were made as well due to fence proximity issues after fence construction. An additional year of offtake data collected in 2014 was available by the time this report was finished and is included in this report as well.

Eight willow monitoring sites were established in the Kawuneeche Valley in 2012 in connection with the construction of an experimental exclosure in the fall of 2011. Baseline data were collected at these sites in 2012–14.

Table 3. Number of sites distributed between fencing treatments and areas burned in the Fern Lake Fire on the elk winter range of Rocky Mountain National Park, Colorado, as of January 2014.
[na, not applicable]

Vegetation Type	Total sites	Fenced-unburned	Unfenced-unburned	Fenced-burned	Unfenced-burned
Aspen					
Core winter range	47	11	29*	3	4
Noncore winter range	21	na	20	na	1
Willow					
Core winter range	85	23	34	10	18
Noncore winter range	34	na	34	na	na
Upland					
Core winter range	25	na	25	na	na
Noncore winter range	23	na	23	na	na

*Some of these sites may potentially be fenced in the future.

Aspen

At each aspen monitoring site, all live and dead aspen stems greater than 2.5 m in height falling within a 5-m x 5-m square plot were tallied according to stem-size class. Each stem-size class covered a 2-centimeter (cm) range of diameter at breast height (dbh) from 0 to 2 cm dbh to greater than 34 cm dbh. Based on these stem-size classes, stems were then tallied into three diameter classes: small-diameter (0–10 cm dbh) stems, medium-diameter (10–20 cm dbh) stems, and large-diameter (greater than 20 cm dbh) trees. Aspen stems less than or equal to 2.5 m in height were tallied by height class (0–50 cm, 51–100 cm, 101–150 cm, 151–200 cm, 201–250 cm).

Upland

Upland offtake data were collected in April and May during 2009–14. Herbaceous offtake was measured using movable 1-square-meter (m^2) grazing cages to protect plants from grazers during the grazing season (in this case, winter) and then clipping all herbaceous vegetation from 0.25-m^2 plots protected by the grazing cage and equally sized paired plots outside the cage to determine percentage of offtake (Bonham, 1989). The cages were moved to a new location within each site at the beginning of each sampling period to protect a new plot from grazing. Winter offtake was determined by comparing biomass at the end of winter inside the cage to biomass remaining in the uncaged, grazed plot using equation 1.

$$O_w = 100 \times \left[\frac{(B_i - B_o)}{B_i} \right] \tag{1}$$

where

O_w	is the percent of end of growing season standing crop that is used overwinter,
B_i	is the amount of biomass in caged plot at the end of winter, and
B_o	is the amount of biomass in paired uncaged plot at the end of winter.

Upland shrub species, cover, and height were measured for each shrub whose canopy intercepted a permanently marked 30-m transect line through the site. These measurements were taken from all upland sites at baseline and again in 2013.

Willow

Willow cover and average heights were determined by measuring all willows falling within a 4-m x 4-m square plot (macroplot) at each site as well as those that intercepted a 5.7-m transect line (line intercept) that bisected the square plot from east to west. A more intensive and more accurate method (macroplot) and a less intensive and less accurate method (line-intercept) were used to provide a means for scaling down monitoring effort in the future if funding for data collection was reduced. Both methods were used to collect data at baseline and, in 2013, to provide an appropriate basis for comparison if the future need arises to scale back sampling to the less intensive measures. Because the macroplot method gives a better representation of willow cover, data from this method are reported in the "Results" and "Discussion" sections. The line-intercept results are included in the appendix (tables 1–1 and 1–2). Similarly, two methods of measuring ungulate browsing on willow were incorporated into the willow-monitoring protocols. The "stem-scaled diameter difference method" (also called the "DD2 method" in Bilyeu and others, 2007) calculates offtake as the difference between diameter at bud scar and at browse point scaled by the number of shoots browsed on the entire stem (see Zeigenfuss and others, 2011). The "production-weighted diameter difference method" (DD3) recommended by Bilyeu and others (2007), which accounts for browser preference for more productive shoots, was also used to estimate the percentage of biomass removed based upon shoot size of browsed and unbrowsed shoots (equation 2).

$$DD3 = \left(\frac{b \times B}{b \times B + u \times U} \right) \times \left(\frac{D_p - D_t}{D_b - D_t} \right) \tag{2}$$

where	b	is the number of browsed shoots on the stem,
	u	is the number of unbrowsed shoots,
	D_p	is shoot diameter at the point of browsing,
	D_t	is the average diameter of unbrowsed shoot tips,
	D_b	is the diameter at the base of the shoot,
	B	is average pre-browse mass of browsed shoots, and
	U	is the average mass of unbrowsed shoots.

For clarity of discussion, the DD3 results are used in the "Results" and "Discussion" sections of this report, but the DD2 results have been included in the appendix (table 1–1).

The EVMP included the possibility of taking actions that would encourage elk migration to summer ranges, so summer offtake was anticipated to be minimal. The DD2 and DD3 methods described above estimate total annual willow offtake but were being used as analogous to winter offtake only because a minimal amount of summer willow offtake was anticipated. From the outset of the

EVMP monitoring program, field crews observed summer browsing on willow and presence of elk on the winter range during the summer. At this time, no management actions have been taken to encourage movement of elk from the winter range during the summer; therefore, a rapid survey was conducted in August 2014 to document percent of willow shoots browsed and to determine whether additional monitoring of summer browsing may be needed.

Elk Population and Climate Data

A combination of annual aerial and ground surveys was used annually to estimate the winter population size of elk using low elevation range in the park. Aerial survey data were corrected based on sightability factors and a Bayesian population model was used to estimate trends in elk population size. Precipitation data were acquired from the National Climate Data Center (National Oceanic and Atmospheric Administration, 2014) for COOP weather station #52761 in Estes Park, Colo., Snow depth data were acquired from the Bear Lake Snotel Site #322 (Natural Resources Conservation Service, 2014) located 5 kilometers southwest of the core elk winter range at an elevation of 2,900 m. These data were used to examine the relationship of weather and elk population to patterns in annual offtake data.

Statistical Analysis

The original analysis of baseline data involved making comparisons between core and noncore winter-range areas and presenting the descriptive characteristics of each type (table 2). We modified this structure to include the new burned sites. Statistical tests were carried out using SAS version 9.2 (SAS Institute, Cary, North Carolina). All percentage data were converted to proportions and then transformed using a logit transformation (Warton and Hui, 2011) prior to statistical testing. We tested for differences between baseline and 2013 values for all willow variables, except offtake, using mixed effects regression models (PROC MIXED, PROC GLIMMIX, SAS Institute, Cary, N.C.). Willow and upland offtake were analyzed using linear and quadratic regression models. Baseline upland herbaceous offtake estimates were originally derived using a method that included negative values (Zeigenfuss and others, 2011). Upon recent consultation with a statistician, however, we decided to remove all negative values from the data because they do not allow for the required data transformations and because such values are nonsensical (in other words, there is no opportunity for plant regrowth following grazing in winter, which is the only situation that could logically result in greater production outside the grazing cages compared to inside the cages and thus a negative offtake value). Changing the offtake calculation method allowed for easier comparison of trends and interannual differences but required recalculation of baseline thresholds of distribution of offtake across the landscape. Offtake data were transformed using a logit transformation and were then analyzed for differences using a mixed model.

We used weighted averages based on the amount of aspen, willow, and upland area available in the core and noncore winter range to report winter-range-wide estimates of cover, offtake, 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.

Results

Aspen

Stand Structure

Unfenced, core winter-range aspen stands maintained stable size (age) structure of tree-size (greater than 2.5 m height) stems from 2008 to 2013 (fig. 3). At baseline, 3 of 29 sites had a small-diameter (0–10 cm dbh) component, and by 2013 only two sites had small-diameter trees because all small-diameter trees at one site had moved into the medium-diameter (10–20 cm dbh) class. At many sites, there was a steady progression into the large-diameter (greater than 20 cm dbh) tree class with no concurrent shift of saplings and small-diameter trees into the small- and medium-diameter tree classes. In these unfenced stands, lack of recruitment of regenerating trees into the stand canopy continues while the stand as a whole continues to age with some older, larger trees eventually dying.

Notable change was observed, however, within fenced aspen stands on the core winter range (fig. 3). An increase in the percent of tree-sized stems in small-diameter classes was seen in 8 of 10 fenced sites that were measured at baseline. Only one of these aspen sites had a small-diameter component at baseline sampling. In addition, all of these stands had a substantial proportion of stems that were less than or equal to 2 cm dbh. At baseline sampling, this 0–2-cm class was so rarely encountered (only 1 stand of the original 65 winter-range sites) that stems of this size were not tallied at baseline, only noted.

Overall, little change was observed in the size/age structure of noncore winter-range aspen stands between baseline and 2013 (fig. 4); however, some exceptions were found. One site (site number 2) that had no tree-size stems but did have a cohort of saplings (stems less than or equal to 2.5 m height) at baseline, now has a small-diameter class of trees. This site is located in the Hidden Valley area. Only two other sites had any small-diameter trees in 2013 and these small-diameter trees existed in both sites at baseline; however, the percentage of stems in the small-diameter class has dropped at one site number 23) since establishment and no very small trees (0–2 cm dbh) were observed, which indicated a lack of ongoing recruitment of saplings in this stand. Another two sites (numbers 6 and 10) lost their only living tree-size stems. In one site (site number 21), measurement error in determining whether a single tree was in or out of the site resulted in a situation where no tree-size stems were present at baseline, but a large-diameter tree class was present in 2013 (fig. 4).

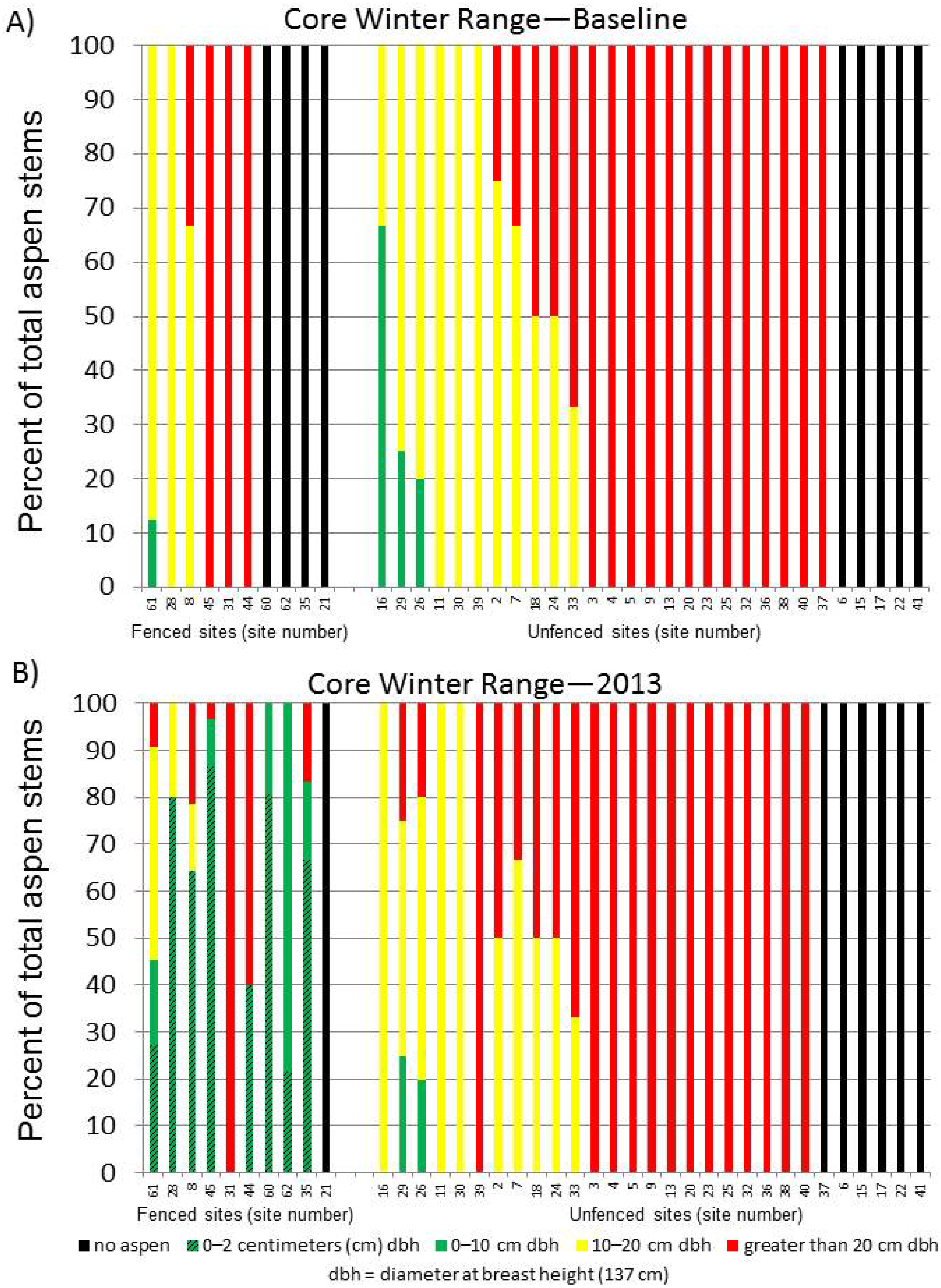

Figure 3. Distribution of aspen tree (height greater than 2.5 meters) stem diameters in monitoring sites on core winter range of Rocky Mountain National Park, Colorado. *A*, baseline measurement (2007/2008 for most sites). *B*, first measurement period (2013).

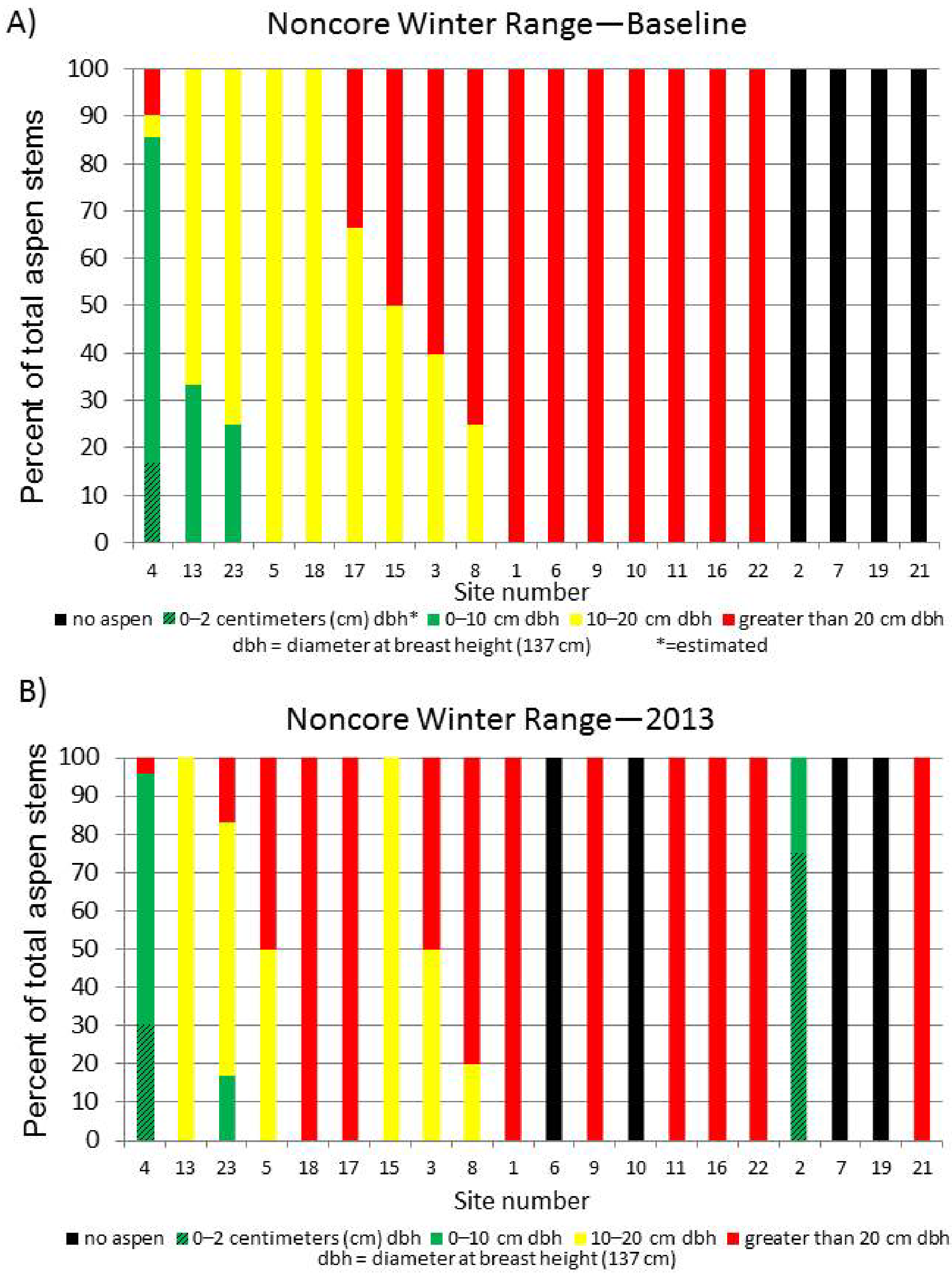

Figure 4. Distribution of aspen tree (height greater than 2.5 meters) stem diameters in sampled sites on noncore winter range of Rocky Mountain National Park, Colorado. *A*, baseline measurement (2007/2008 for most sites). *B*, first measurement period (2013). The 0–2-centimeter class at baseline was estimated.

Almost no change in stand structure was observed in the Kawuneeche Valley since site establishment in 2007 (no aspen are fenced in the Kawuneeche Valley). Only one of the eight sites here had a small-diameter (0–10 cm dbh) class in 2013 and none of these small-diameter stems were in the very smallest (0–2 cm dbh) class that would indicate that saplings are recruiting into the stand. In two other sites, medium-diameter trees moved into the older age/larger size class (fig. 5).

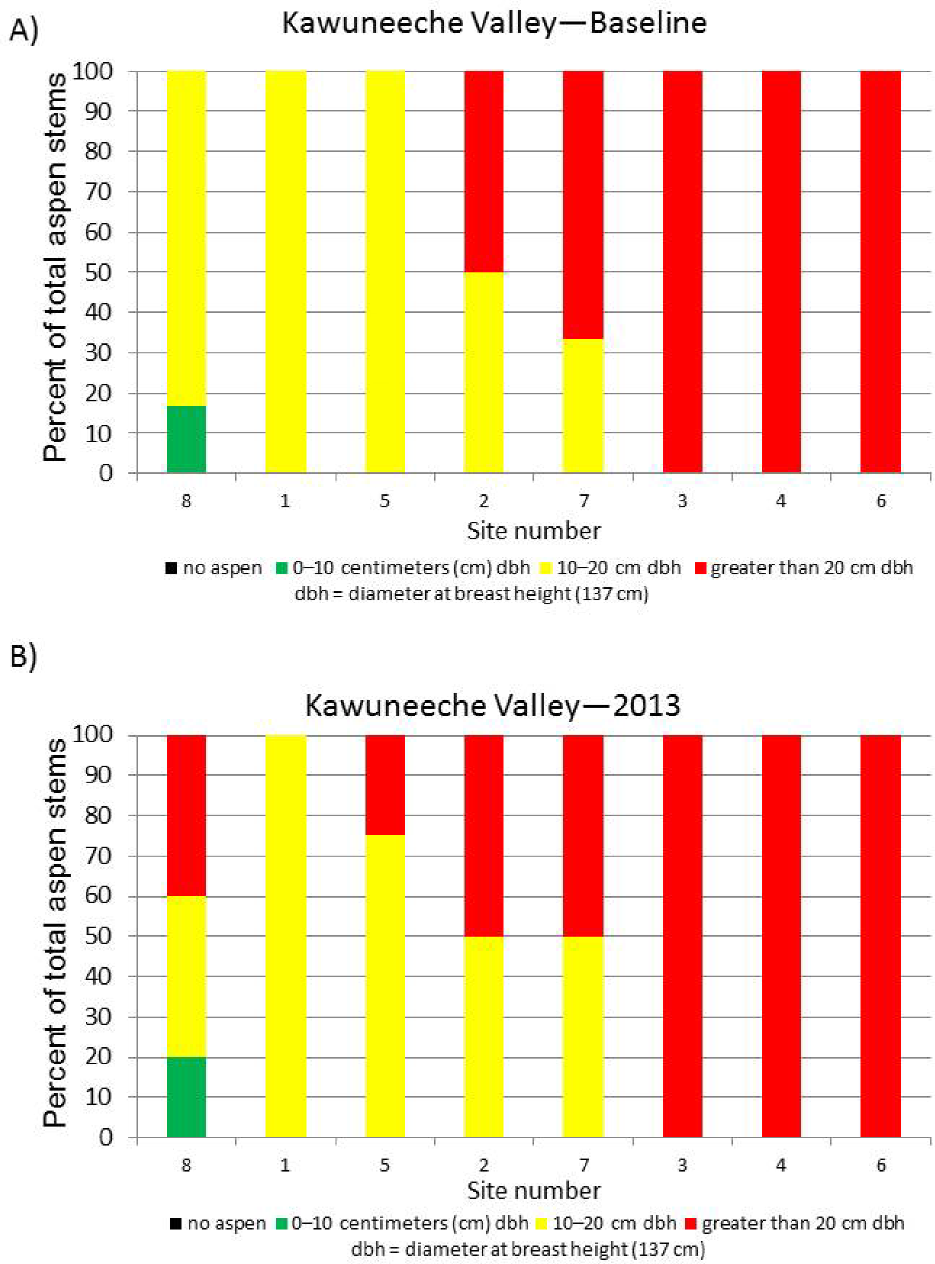

Figure 5. Distribution of aspen tree (height greater than 2.5 meters) stem diameters in sampled sites in the Kawuneeche Valley of Rocky Mountain National Park, Colorado. *A*, baseline measurement (2007/2008 for most sites). *B*, first measurement period (2013).

Sapling Regeneration

Short sapling (less than or equal to 1.5 m height) density was not different between 2013 and baseline for noncore winter range sites (table 4, *p*-value = 0.265). There was a nearly significant (*p*-value = 0.073) increase in these short saplings on unfenced core winter-range aspen sites from baseline to 2013, whereas density of short saplings decreased on fenced core winter-range aspen sites (*p*-value = 0.047, table 4); however, the 2013 sapling densities include several sites that were burned in the Fern Lake Fire and had an extremely high amount of aspen suckering post-fire.

On the core winter range, density of tall aspen saplings (stems 1.5–2.5 m in height) was greater in 2013 than 2008 on sites that were fenced (table 4, *p*-value = 0.003). Eighty percent of the monitoring sites that had been protected by fences had high density (greater than 1,000 stems per acre) of tall (1.5–2.5 m height) saplings (table 4, fig. 6) by 2013. On the noncore winter range, there was little change in either short or tall sapling density since the baseline sample (*p*-value greater than or equal to [≥] 0.21, table 4).

At baseline there was no difference in tall aspen sapling density between the core and noncore sites (*p*-value = 0.240), but by 2013, density of tall aspen saplings was greater on fenced sites compared to unfenced sites, both in the core and noncore winter range sites (*p*-value less than [<] 0.001, fig. 6). Although noncore winter-range sites had the highest densities of tall saplings at baseline, by 2013 fenced core winter-range sites had significantly greater density of tall saplings than noncore winter-range sites (*p*-value < 0.001). Tall saplings remained nonexistent on Kawuneeche Valley sites during the time since site establishment (table 4).

Table 4. Least squares means of aspen sapling (plants with height less than or equal to 2.5 meters) density at all aspen monitoring sites in Rocky Mountain National Park, Colorado. Baseline data collected 2006–9, with most of the sites collected in 2007 or 2008. P-values report differences of least squares means of log-transformed density between baseline and 2013 samples. In 2013, this sample includes four sites burned by the Fern Lake Fire.

[s.e., standard error; n, number sites sampled; m, meter; na, not applicable]

Winter range zone	Aspen sapling density (stems/acre)									
	Short (stems ≤ 1.5 m height)					Tall (stems 1.5–2.5 m height)				
	Baseline (mean ± s.e.)	n	2013 (mean ± s.e.)	n	*P*-value[1]	Baseline (mean ± s. e.)	n	2013 (mean ± s.e.)	n	*P*-value[1]
Core winter range, unfenced	6,667 ± 2,483	32	8,317 ± 2,481	33	0.073	5 ± 8	32	20 ± 8	33	0.179
Core winter range, fenced	7,269 ± 1,600	13	4,664 ± 1,550	14	0.047	25 ± 513	13	1,851 ± 495	14	0.003
Noncore winter range	1,902 ± 438	20	2,329 ± 433	21	0.265	131 ± 93	20	162 ± 93	21	0.210
Kawuneeche Valley	1,640 ± 478	8	1,984 ± 478	8	0.783	0	8	0	8	na

[1]Reported *p*-value is for log-transformed sapling density variable.

Figure 6. Comparison of two fenced aspen sites on the elk winter range of Rocky Mountain National Park, Colorado. Colored arrows indicate landmarks for reference. *A, B*, baseline measurement prior to fencing. *C, D*, after 4 years of protection from ungulate browsing (2013).

Effects of Burning

Although the Fern Lake Fire caused mortality of some small-diameter aspen trees in the burned sites (mortality of one-half or more of measured stems less than 10 cm dbh, fig. 7), burning seemed to have a positive effect on sapling regeneration. Extensive suckering was observed on nearly all the burned sites. The seven core winter-range sites measured in 2013 that were burned had high density of short (less than or equal to 1.5 m height) saplings (table 5, figs. 8 and 9). Three of the four newly established burned sites had over 25,000 suckers per ha, indicating the potential for good recruitment in the next few years as these suckers continue to grow. All three of these sites with high sucker densities were located outside of fences. One noncore site that was burned had almost no suckering. This site was in a moist area where abundant growth of grasses and forbs may have outcompeted new aspen saplings. Tall aspen saplings (1.5 to 2.5 m height) had the highest density in unburned, fenced core winter-range sites compared to all other sites (p-value < 0.001). The fire may have removed this size on burned sites inside fences and many new suckers likely did not have enough time to reach this taller height in one season after burning.

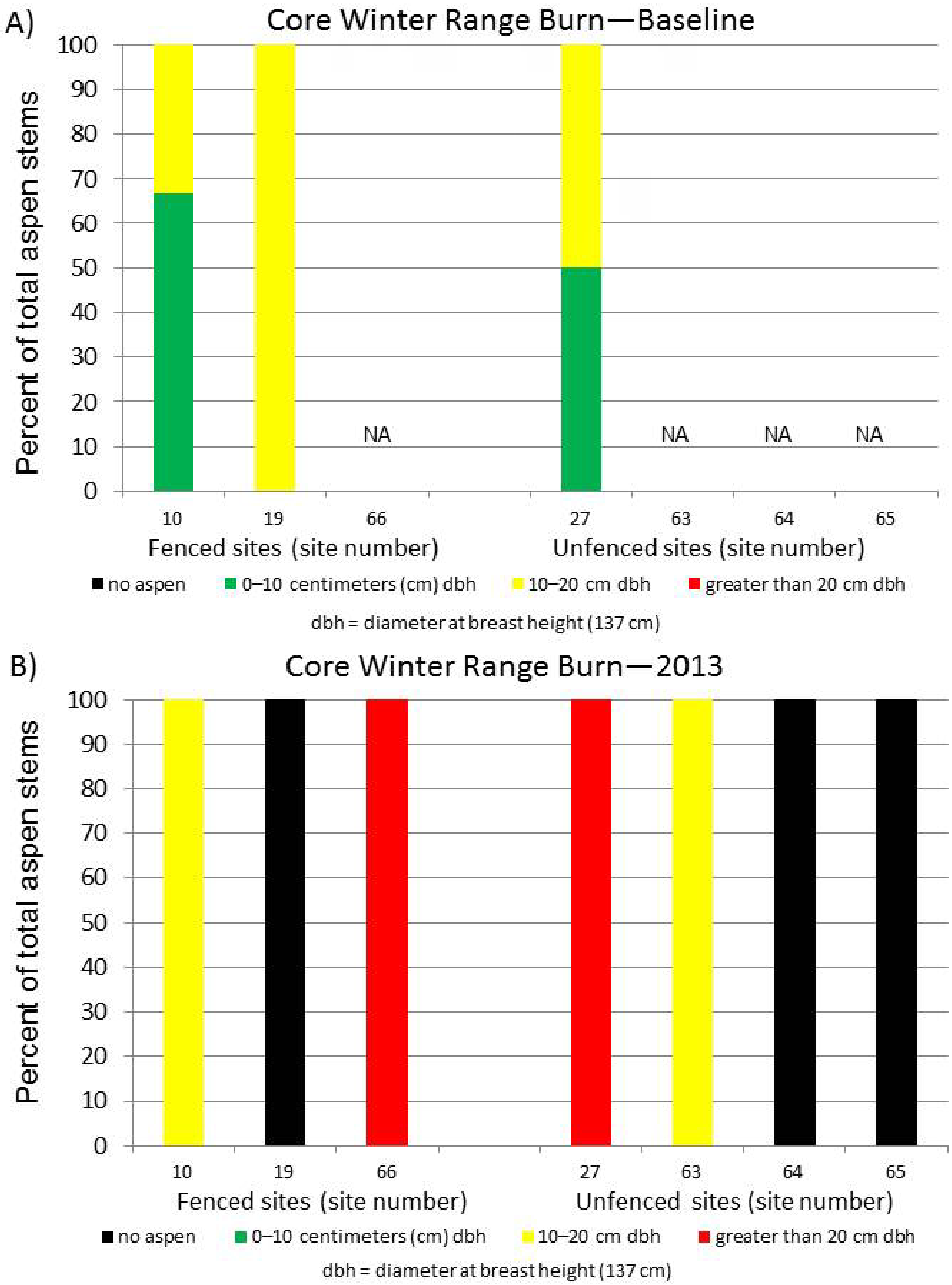

Figure 7. Tree (height greater than 2.5 meters) stem distribution of aspen in sites burned by the 2012 Fern Lake Fire. Four of the seven sites displayed were established after the fire to assess effects of burning on aspen sites. No trees with 0–2-centimeter diameter at breast height [dbh] stems were found. *A*, at baseline. *B*, in 2013.

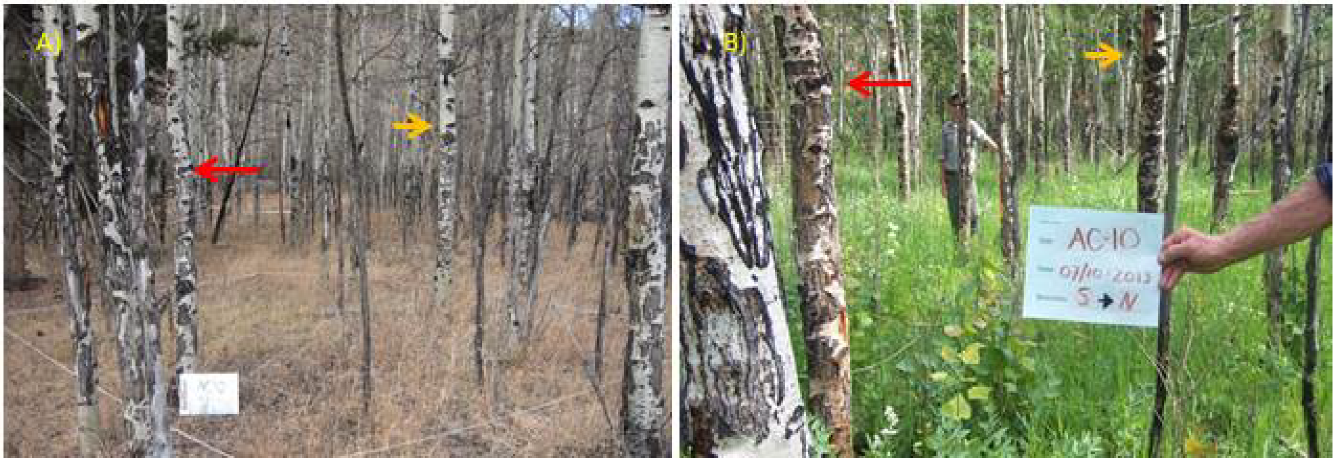

Figure 8. Comparison of moderately burned aspen site on the elk winter range of Rocky Mountain National Park, Colorado. Colored arrows indicate landmarks for reference. *A*, baseline measurement prior to fencing and burning. *B*, after 4 years of protection from ungulate browsing in the first season post fire (2013).

Figure 9. New aspen site established in 2013 in area burned by 2012 fire, Rocky Mountain National Park, Colorado. This site had extremely dense regeneration and was unprotected by fences.

Table 5. Least squares means of aspen sapling (plants with height less than or equal to 2.5 meters) density at aspen monitoring sites at sites burned in the 2012 Fern Lake Fire compared to unburned sites in Rocky Mountain National Park, Colorado. Baseline data collected 2006–9, with most of the sites collected in 2007 or 2008.

[s.e., standard error; n, number sites sampled; m, meter; na, not applicable]

Winter range zone	Aspen sapling density (stems/acre)									
	Short (stems ≤ 1.5 m height)					Tall (stems 1.5–2.5 m height)				
	Baseline		2013		*P*-value	Baseline		2013		*P*-value
	(mean ± s.e.)	n	(mean ± s.e.)	n	(test between sample dates)[1]	(mean ± s.e.)	n	(mean ± s.e.)	n	(test between sample dates)[1]
Core winter range, unfenced										
Burned	324	1	35,505 ± 17,386	4	na	162	1	40 ± 40	4	na
Unburned	3,275 ± 709	31	4,736 ± 723	29	0.135	0 ± 6	31	17 ± 7	29	0.068
Core winter range, fenced										
Burned	5,506 ± 607	3	7,881 ± 607	3	0.061	0 ± 0	3	0 ± 0	3	na
Unburned	7,899 ± 2,051	10	3,798 ± 1,983	11	0.013	32 ± 618	10	2,355 ± 589	11	0.004
Noncore winter range										
Burned	1,296	1	162	1	na	0	1	162	1	na
Unburned	1,930 ± 452	19	2,437 ± 448	20	0.214	138 ± 98	19	162 ± 97	20	0.358

[1]Reported *p*-value is for log-transformed sapling density variable.

Upland

Offtake

Winter offtake levels on upland herbaceous plants were highly variable from 2007 to 2014, declining from 2009–11, but then increased from 2012 through 2014 (fig. 10). Overall, upland offtake dropped from an average 61.8 ± 3.8 (least squares mean ± standard error) percent in 2007–9 to 44.5 ± 2.9 percent in 2010–14 (p-value = 0.012). Prior to 2013, the overall trend was a similar and slightly significant downward trend in both core (coefficient of determination [R^2] = 0.584, p-value = 0.077) and noncore (R^2 = 0.672, p-value = 0.046) winter-range sites. Both offtake estimation methods yielded similar results with comparable patterns and trends in offtake levels. Although the offtake values were different for core compared to noncore sites, similar patterns in offtake trends were observed on both core and noncore winter range, except in 2009 and 2014, (table 6). Few direct correspondences could be determined between elk population size in the park and upland offtake (fig. 10). One interesting pattern in upland offtake involves an apparent shift in offtake from the core winter range to the noncore winter range. By 2014, offtake on the noncore winter range was much greater than core winter range, though this difference was not significant (p-value = 0.126).

Table 6. Annual upland herbaceous offtake on elk winter range in Rocky Mountain National Park, Colorado, 2007–14. Original 2007–8 data were estimated using a method that included offtake values of -100 to 100. Current analysis removed all negative values from the dataset. * denotes difference between core and noncore winter range offtake in that year (p-value less than 0.05).
[yr, year; s.e., standard error; n, number sites sampled]

Sample yr		Entire winter range		Core winter range		Noncore winter range	
		Percent offtake (mean ± s.e.)	n	Percent offtake (mean ± s.e.)	n	Percent offtake (mean ± s.e.)	n
Baseline (2007–8)	Original	47.1 ± 4.3	75	52.4 ± 3.9	44	40.3 ± 5.9	31
	Revised	61.3 ± 1.8	43	61.3 ± 2.6	45	60.0 ± 3.2	30
2009		63.7 ± 4.5	17	72.1 ± 3.8*	10	47.3 ± 6.8*	7
2010		45.9 ± 4.8	18	49.6 ± 6.5	9	40.8 ± 7.4	9
2011		32.6 ± 4.0	18	31.9 ± 5.3	9	33.4 ± 6.5	9
2012		42.4 ± 6.3	17	41.5 ± 6.5	10	44.4 ± 13.5	7
2013		52.8 ± 5.6	15	51.1 ±7.4	10	57.3 ± 8.2	5
2014		48.8 ± 5.4	15	41.4 ± 5.6	8	60.5 ± 8.8	7

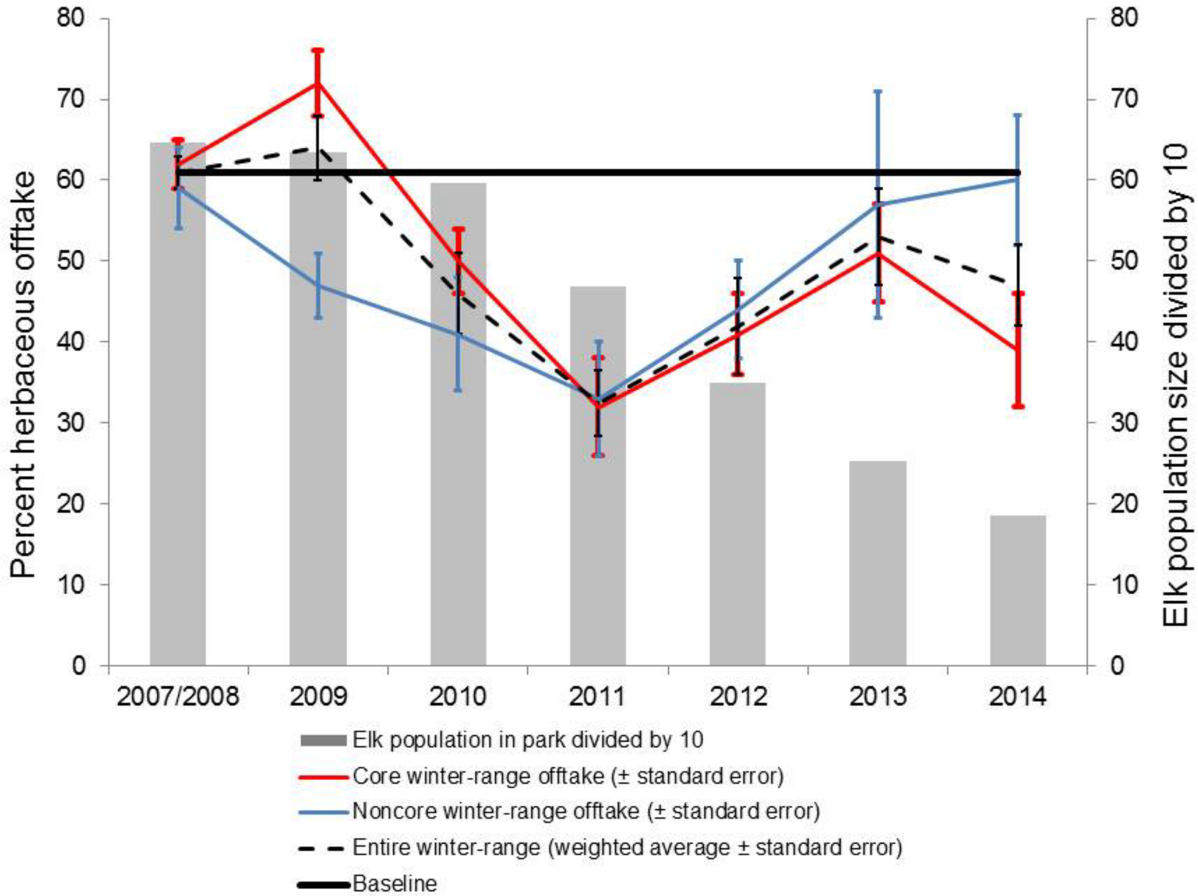

Figure 10. Patterns of upland herbaceous offtake relative to estimated park elk population size on the elk winter range of Rocky Mountain National Park, Colorado, 2007–14. Population estimates from N.T. Hobbs, Senior Research Ecologist, Natural Resources Ecology Lab, Colorado State University (unpub. data, 2014).

We also looked for patterns in offtake related to snow depth recorded at the Bear Lake Snotel site and April–July precipitation as recorded at the Estes Park weather station. Most upland sites are located on south- and west-facing slopes that remain snow free or have low snow accumulations for most of the winter and, therefore, might be subject to grazing when other parts of the winter range are snow-covered. Precipitation is very important to production on these dry upland sites.

No obvious connection was noted between snowpack and offtake, and offtake was not correlated with maximum snow depth at the Bear Lake Snotel site; however, offtake on the entire winter range was lower in years with higher April–July precipitation (p-value = 0.001, R^2 = 0.86, fig. 11). Offtake rates were significantly greater in core winter range in 2009 compared to noncore winter range (p-value = 0.006), but not in any other year. A prescribed burn conducted in the Upper Beaver Meadows and Deer Ridge areas in fall of 2008 burned approximately 208 ha, and this may have reduced available upland forage on burned areas during winter 2008–9, resulting in higher concentration of elk use on other upland areas of the core winter range.

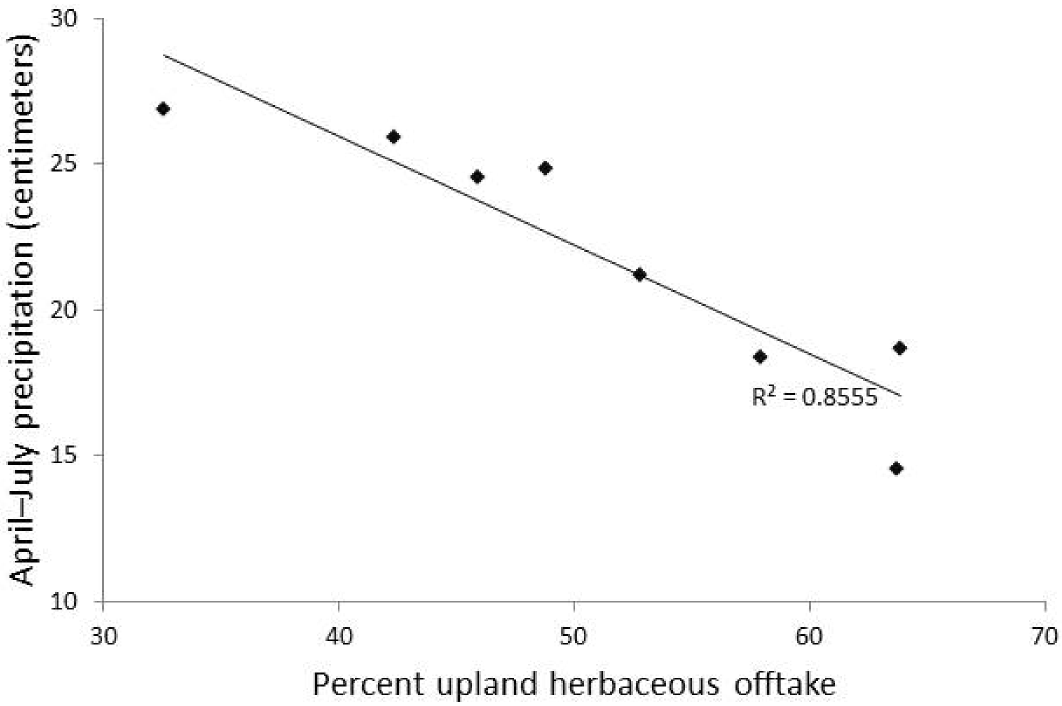

Figure 11. Relationship between upland herbaceous offtake and growing season precipitation (April–July) on the elk winter range of Rocky Mountain National Park, Colorado.

Distribution of offtake across the landscape (core and noncore sites combined) varied from 2008 through 2014 (table 7). We used the revised offtake calculation method to estimate baseline offtake and then revised distribution objectives to align them with this new method. The revised objectives are (1) no net increase in winter upland herbaceous offtake across the winter range above baseline levels of 61 percent, (2) with less than or equal to 30 percent of sites with offtake greater than 70 percent and (3) less than or equal to 10 percent of sites with offtake greater than 85 percent. Much less upland winter range was grazed intensively (greater than 85 percent offtake) from 2010 to 2014 than at baseline or 2009.

Table 7. Distribution of annual upland herbaceous offtake on elk winter range of Rocky Mountain National Park, Colorado.

[Gray shading highlight indicates years when offtake targets for high levels of browsing set in the Elk-Vegetation Management Plan (National Park Service, 2007) were met.]

Offtake level	Annual percent of sites within offtake levels (all negative offtake values removed)						
	Baseline	2009	2010	2011	2012	2013	2014
0–50 percent	26	24	56	78	65	40	53
50–70 percent	44	29	28	22	18	40	40
70–85 percent	19	41	17	0	12	20	0
>85 percent	11	6	0	0	6	0	7

Shrub Cover

Mean shrub cover on upland monitoring sites did not change between baseline sampling in 2007 and 2013. Core winter-range sites had mean shrub cover of 9.4 ± 2.9 percent (mean ± standard error) and noncore sites had a mean shrub cover of 21.1 ± 4.2 percent (fig. 12) in 2013. Noncore upland winter-range sites had greater shrub cover both at baseline and in 2013 compared to core winter-range sites. Using transformed data variables, this difference was not significant at the baseline sampling but was significant by 2013 (*p*-value = 0.046), which may reflect removal of shrubs by prescribed burns conducted on the core winter range during the monitoring period, rather than any effect of herbivory.

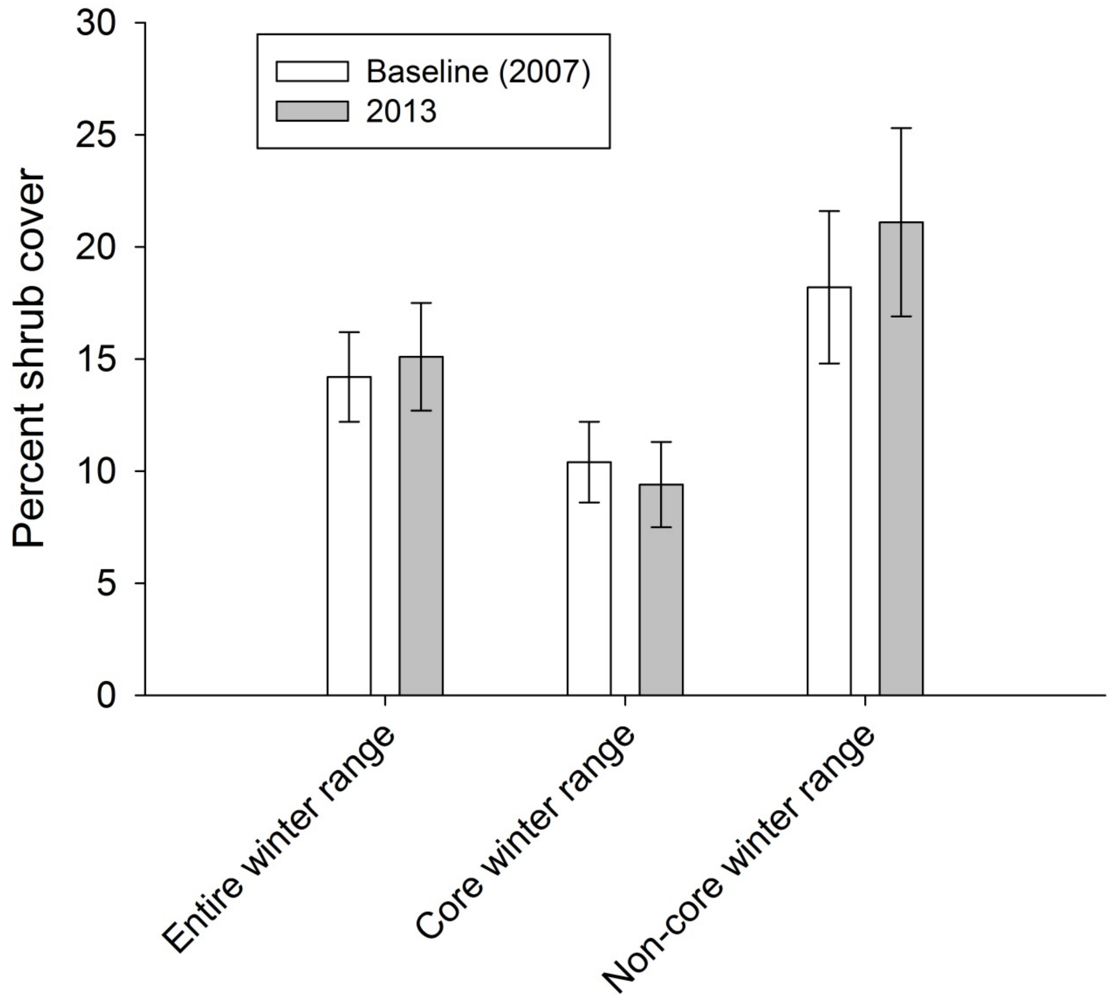

Figure 12. Mean shrub cover on upland sites on elk winter range in Rocky Mountain National Park, Colorado.

Willow

Offtake

Willow offtake was highly variable from 2009 to 2014 on the core winter range and the winter range as a whole but remained fairly consistent on the noncore winter range from 2009 to 2011 and then began to decline (fig. 13). Offtake was significantly lower throughout the winter range in 2013 (p-value = 0.045) but increased again in 2014. Although offtake on the noncore winter range seemed to track the decline in elk populations, a similar pattern was not evident for the core winter range (fig. 13). Willow offtake was higher on the core winter range than the noncore winter range throughout the 5 yrs evaluated (p-value = 0.004). Patterns in offtake for the entire winter range paralleled those on the core winter range because most of the winter-range willow areas are on the core winter range (fig. 13). Willow offtake was not correlated to elk population size, snow depth, total water year precipitation, or growing season (April–July) precipitation. Patterns in offtake were similar using the DD2 offtake estimation method (appendix, table 1–1).

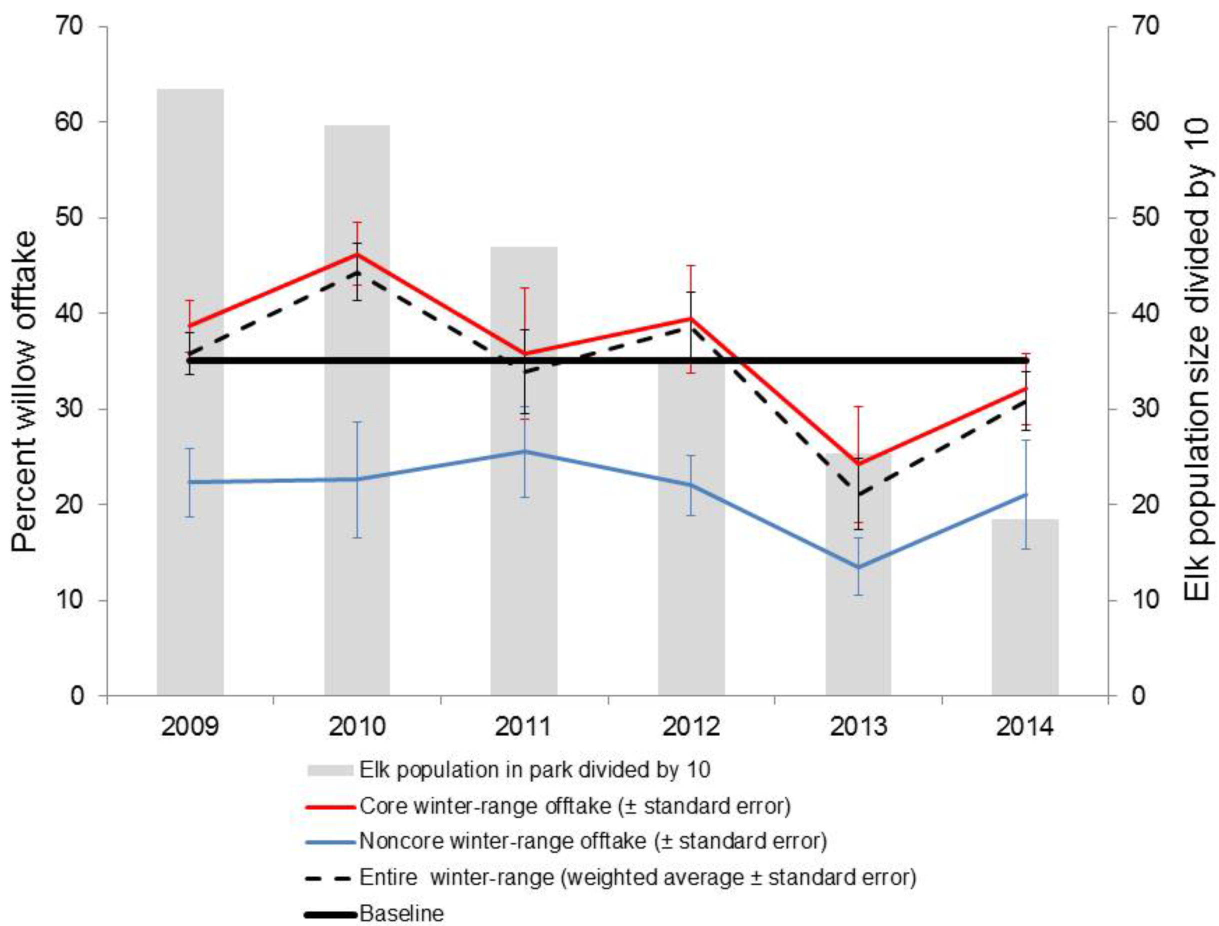

Figure 13. Mean winter willow offtake relative to park elk population size (divided by 10) on the elk winter range of Rocky Mountain National Park, Colorado, 2007–14. Population estimates from N.T. Hobbs, Senior Research Ecologist, Natural Resources Ecology Lab, Colorado State University (unpub. data, 2014).

Willow Height

Heights of unfenced core winter-range willows did not change significantly during 2008–13 (p-value = 0.304), but willow heights did increase on fenced areas of the core winter range as well as on the noncore winter range (p-value ≤ 0.009, table 8). Average heights of fenced willows were roughly 30 cm taller in 2013 than 2008. The same was true for maximum willow heights with maximum heights of fenced willows increasing roughly 40 cm (table 8). Height gains were similar in the noncore winter range.

Noncore willows were taller than core willows in 2008 and 2013, even after 5 yrs of fencing (p-value < 0.001). In 2008, core winter-range willows that were eventually fenced had lower maximum heights than willows on sites that were not fenced in the core winter range, though this difference was not significant (p-value = 0.009). It is not surprising that fenced sites had lower baseline heights because the willow sites selected for fencing were those with the poorest willow condition. Similar statistical results were observed using both the macroplot and the line intercept method, although the actual size estimates were often quite different (see appendix, table 1–2).

Shrub Cover

Between 2008 and 2013, the percentage of original winter-range willow sites (those not affected by the Fern Lake Fire) that had living willow plants dropped slightly from 94.5 percent to 93 percent. No significant change in willow cover was observed in the unfenced, core winter range between the baseline samples and 2013 (p-value = 0.181; fig. 14; table 8), but willow cover in fenced, core winter-range sites had a nearly significant (p-value = 0.081) increase and cover of noncore winter-range willow increased significantly (p-value = 0.009; fig. 15; table 8).

At baseline, there was greater cover in the noncore winter-range willow sites compared to sites destined for fencing (p-value = 0.034). By 2013, noncore winter-range sites had greater willow cover than both fenced and unfenced, core winter-range sites (p-value ≤ 0.008).

The line-intercept method resulted in similar statistical results compared to the macroplot method of measurement, and the line-intercept method did not pick up the difference in cover for fenced willows. Sample size was not sufficient to adequately test differences between core winter-range sites in Horseshoe Park, Moraine Park, and Beaver Meadows because this was not the original intent of the monitoring; however, Horseshoe Park had substantially greater percent willow cover (30 ± 5) in 2008 than sites in Moraine Park (8 ± 2) or Beaver Meadows (8 ± 3), fairly similar to noncore sites (35 ± 5). By 2013, willow cover in Beaver Meadows and Moraine Park (including those sites burned in the Fern Lake Fire) had increased to 10 ± 3 percent and 18 ± 4 percent, respectively, but were still much lower than Horseshoe Park (41 ± 5 percent) and noncore winter-range sites (44 ± 6 percent).

Table 8. Average and maximum willow height and percent willow cover on the elk winter range of Rocky Mountain National Park, Colorado, at baseline measurement (2008–9) and at first 5-yr sampling (2013). Estimates are for the macroplot methods described in Zeigenfuss and others (2011) and include sites that were burned in 2013. Least squares means and standard errors are presented for individual winter range categories, and weighted means and related standard errors based on proportion of winter range in each category are presented for the entire winter range (bottom row).

[cm, centimeter; s.e., standard error; yr, year; na, not applicable]

Winter range zone	Willow height (cm)			Maximum willow height (cm)			Willow cover (percent)		
	Baseline (mean ± s.e.)	2013 (mean ± s.e.)	*P*-value (yr)	Baseline (mean ± s.e.)	2013 (mean ± s.e.)	*P*-value (yr)	Baseline (mean ± s.e.)	2013 (mean ± s.e.)	*P*-value[1] (yr)
Core winter range, unfenced	92 ± 14	102 ± 14	0.304	196 ± 24	168 ± 21	0.131	21 ± 4	21 ± 3	0.181
Core winter range, fenced	65 ± 6	98 ± 9	<0.001	99 ± 14	140 ± 13	0.004	16 ± 5	28 ± 5	0.081
Noncore winter range	160 ± 20	185 ± 20	0.009	256 ± 27	300 ± 27	0.001	35 ± 5	44 ± 6	0.009
Entire winter range (weighted average)	79 ± 7	102 ± 9	na	157 ± 14	177 ± 14	na	20 ± 2	24 ± 3	na

[1]Reported *p*-value is for transformed willow cover variable.

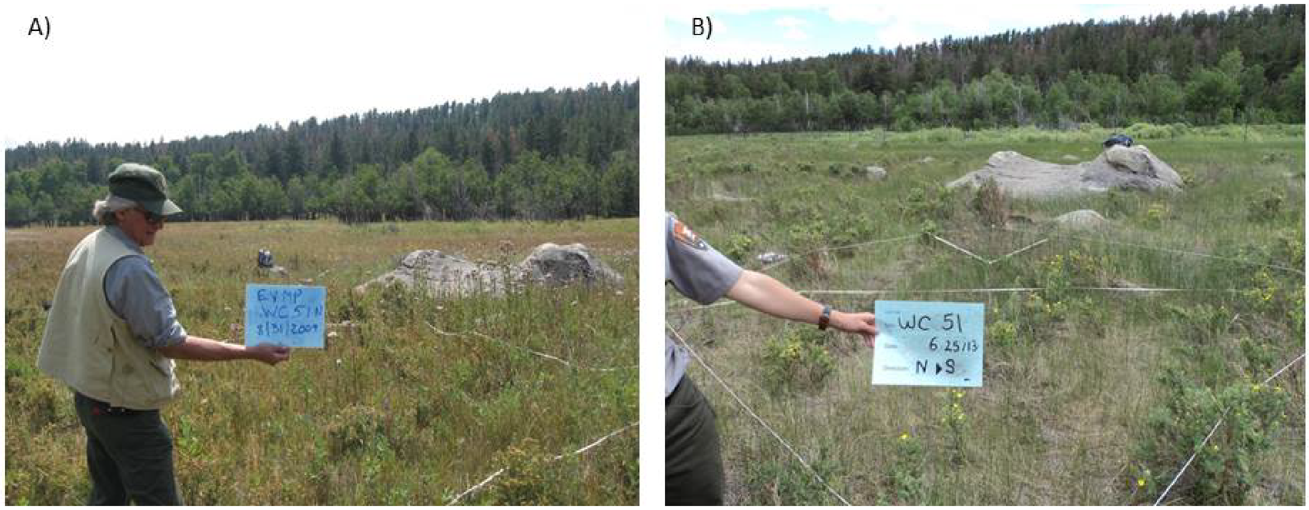

Figure 14. Comparison of unfenced willow site on the elk winter range of Rocky Mountain National Park, Colorado. *A*, baseline measurement (2009). *B*, 4 years after baseline measurement (2013).

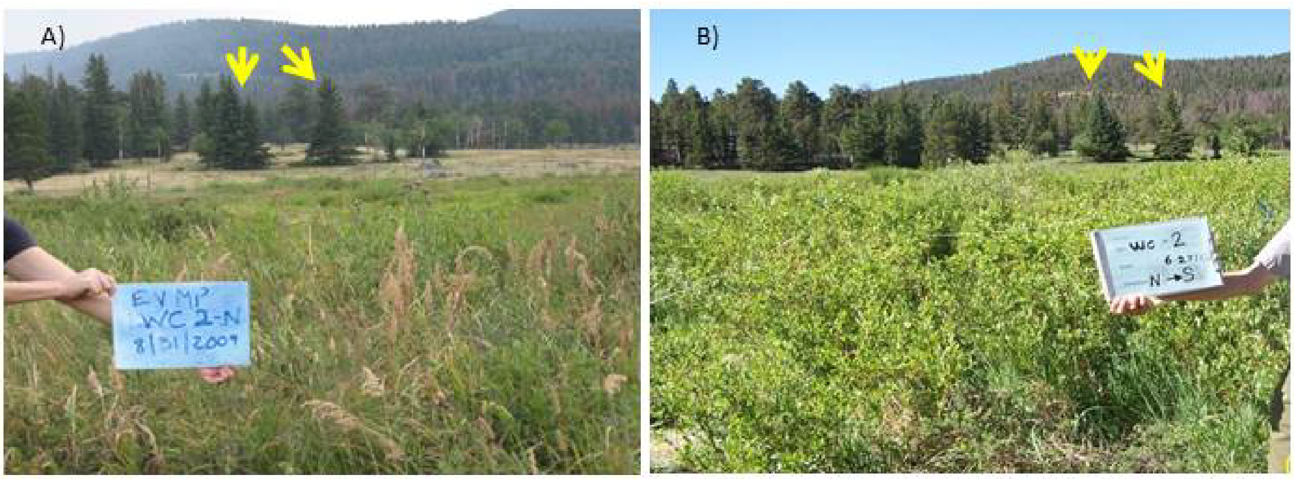

Figure 15. Comparison of unburned willow site on the elk winter range of Rocky Mountain National Park, Colorado. Colored arrows indicate reference landmarks. *A*, one growing season after being fenced (baseline measurement). *B*, after 5 years of protection from ungulate browsing (2013).

Effects of Burning

The Fern Lake Fire moderately to severely burned 81 percent of the willow sites in Moraine Park and 30 percent of all sites on the core winter range. Although willow cover on individual burned sites may have decreased substantially, burned sites had much lower cover (p-value = 0.002) from the outset because they were all in Moraine Park, which generally had lower cover than winter range in Horseshoe Park. Fenced, burned sites had particularly low cover at the outset because sites selected for fencing were those in the worst condition. It is possible that there was willow growth inside the fences prior to the 2012 fire, but such growth would have likely been lost to the fire. As a result, willow cover on these sites did not change significantly between baseline and 2013 (table 9).

In 2013, willow cover was lower at both fenced and unfenced, burned sites, than all unburned winter-range willow sites (p-value ≤ 0.038). No differences in average willow height were evident at baseline between plots that were eventually burned in 2013 (p-value ≥ 0.172) in the core winter range, but all core winter-range willow sites had shorter willows than noncore sites (p-value ≤ 0.003); however, after the fire in 2013, all burned sites, whether fenced or not, had shorter willows than unburned sites (p-value ≤ 0.032).

Burning had mixed effects on willow growth in this first season post-fire, with many sites having complete consumption of older plants, no regeneration of new plants (K. Kaczynski, Postdoctoral Research Associate, Department of Forest and Rangeland Stewardship, Colorado State University, written commun., June 2014), and low resprouting of older plants (figs. 16 and 17). Sites within fences seem to have burned more completely than those outside of fences in many cases (fig. 17), possibly due to accumulation of herbaceous litter inside fences.

Table 9. Least squares means of average and maximum willow height and percent willow cover on burned sites compared to unburned sites on the elk winter range of Rocky Mountain National Park, Colorado, at baseline measurement (2008 for most sites) and at first 5-yr sampling (2013). Estimates are for macroplot methods described in Zeigenfuss and others (2011). Least squares means and standard errors are presented for individual winter range categories, and weighted means and related standard errors based on proportion of winter range in each category are presented for the entire winter range (bottom row).
[cm, centimeter; s.e., standard error; yr, year]

Winter range zone	Willow height (cm)			Maximum willow height (cm)			Willow cover (percent)		
	Baseline (mean ± s.e.)	2013 (mean ± s.e.)	*P*-value (yr)	Baseline (mean ± s.e.)	2013 (mean ± s.e.)	*P*-value (yr)	Baseline (mean ± s.e.)	2013 (mean ± s.e.)	*P*-value (yr)[1]
Core winter range, unfenced									
Burned	67 ± 14	42 ± 9	0.137	91 ± 18	56 ± 12	0.122	15 ± 6	12 ± 6	0.242
Unburned	119 ± 19	118 ± 19	0.922	242 ± 30	226 ± 27	0.350	25 ± 5	26 ± 4	0.767
Core winter range, fenced									
Burned	46 ± 13	46 ± 13	0.993	53 ± 14	57 ± 13	0.852	2 ± 2	5 ± 2	0.797
Unburned	71 ± 10	121 ± 9	<0.001	121 ± 15	177 ± 13	0.001	21 ± 6	38 ± 5	<0.001
Noncore winter range									
(all unburned)	160 ± 20	185 ± 20	0.009	256 ± 27	300 ± 27	0.001	35 ± 5	44 ± 6	0.009

[1]Reported *p*-value is for transformed willow cover variable.

Figure 16. Comparison of burned willow site on the elk winter range of Rocky Mountain National Park, Colorado. Colored arrows indicate reference landmarks. *A*, baseline measurement (2009). *B*, 4 years after baseline and following one season of post-fire growth (2013).

Figure 17. Comparison of heavily burned willow site on the elk winter range of Rocky Mountain National Park, Colorado. Colored arrows indicate reference landmarks. *A*, prior to fencing and burning (baseline measurement). *B*, after 4 years of protection from ungulate browsing and first season post-fire (2013). No regrowth of willow was indicated in this site in 2013.

Kawuneeche Valley

Willow offtake at the Kawuneeche Valley willow sites was compared across the 3 yrs the valley was sampled. Baseline (2012) willow offtake was 30 ± 3 percent (mean ± standard error) and remained nearly constant in 2013 (28 ± 4 percent) and 2014 (29 ± 5 percent). The eight Kawuneeche Valley sites also had low willow cover (17 ± 5 percent), mean height (50 ± 4 cm), and maximum height (98 ± 12 cm) at their baseline measure in 2012. These sites were all established in relation to one exclosure site, so they are not representative of the entire Kawuneeche Valley but only of the location of the exclosure.

Discussion

Progress Toward Vegetation Goals

Aspen

Objectives.—Progressive increase in aspen regeneration above the baseline level of 13 percent to at least 45 percent of winter range stands (presence of stems less than 2 cm dbh reaching 1.5–2.5 m tall). Progressive shift in the distribution of stem sizes toward the desired future condition of 75 percent small-diameter stems, 20 percent medium-diameter stems, and 5 percent large-diameter stems.

In 2013, there were signs of a progressive shift toward the desired future condition of a more natural distribution of stem sizes to reflect recruitment of younger (smaller diameter) trees inside fences that excluded ungulates (figs. 3 and 4). In unfenced areas, however, on both core and noncore winter range, stand structure was stagnant, with more trees moving into larger (older) size classes, and no replacement of small-diameter trees.

At baseline, 13 percent of sampled aspen sites on the winter range had a recruiting sapling class (had at least 400 stems per ha that were 1.5–2.5-m height). By 2013, 29 percent of sampled aspen sites had measurable sapling recruitment (table 10). Although an increase was noted across all the winter range (both core and noncore), the most substantial increase was inside fenced areas of the core winter range (tables 4 and 10, fig. 6). The increase in tall saplings inside fences was accompanied by a decrease in the number of short (less than 1.5-m height) saplings, likely due to movement of these short saplings into taller classes. Competition for space and light resources also likely contributed to a decrease in density in smaller size classes as density of larger size stems increased. In unfenced sites of the core winter range, these shorter (0–1.5-m height) saplings had a nearly significant (p-value=0.073, table 4) increase, but much of this increase is attributable to the high level of suckering on the four burned sites.

In 2013, there was still no recruiting sapling cohort in the Kawuneeche Valley stands. No fences were constructed in Kawuneeche Valley aspen. It is unknown whether the lack of sapling recruitment is related to the combined effects of moose and elk browsing in the Kawuneeche Valley or other factors such as disease or water stress. Signs of moose presence (scat, tracks, or browsing) were observed in 80 percent of the Kawuneeche Valley sites, whereas less than 2 percent of winter-range sites had signs of moose presence.

Table 10. Percent of aspen monitoring sites on elk winter range and Kawuneeche Valley of Rocky Mountain National Park, Colorado, that had a recruiting sapling cohort (at least 400 stems per hectare of height between 1.5 and 2.5 meters) at baseline sampling and in 2013. Entire winter range numbers are weighted by the percent area in each location.

Location	Sample date	
	Baseline	2013
Entire winter range	13 percent	29 percent
Core winter range total	7 percent	32 percent
Core, fenced		64 percent
Core, unfenced		12 percent
Noncore winter range total	15 percent	28 percent
Kawuneeche Valley	0 percent	0 percent

It was likely too soon to observe a positive effect of the December 2012 Fern Lake fire on tall saplings (those 1.5–2.5-m height); however, short sapling (stems with height less than or equal to 1.5 m) density on some burned plots on the core winter range exceeded 35,000 stems/acre (table 5, fig. 9). In burned aspen stands, particularly those protected by fences, the large amount of regeneration in the first post-burn year may lead to increased small-diameter trees within the next sampling interval, but high levels of browsing in unfenced, burned areas could also prevent these saplings from being recruited into the stand. Overall, there has been progress toward the aspen vegetation goals between 2008 and 2013, but this progress is primarily in fenced areas on the winter range.

Upland

Objectives.—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 greater than 70 percent and less than or equal to 10 percent of sites with offtake greater than 85 percent.

Herbaceous offtake in upland sites increased above the baseline level on the core winter range in 2009, but dropped below baseline for all other years on both core and noncore winter range. A spike in 2013 was likely in response to loss of large patches of both herbaceous and woody forage in Moraine Park following the 2012 fire and associated shift in habitat use to adjacent upland areas.

After 2009, all years met the objectives for distribution of offtake across the landscape such that not more than 30 percent of the landscape was heavily grazed (greater than 70 percent offtake) and not more than 10 percent of the landscape was very intensively grazed (greater than 85 percent offtake). The stability of shrub cover on upland sites indicates no apparent major shifts from herbaceous communities toward shrub communities due to grazing. Reduction and redistribution of herbaceous offtake in line with EVMP objectives were achieved during recent years.

Willow

Objectives.—No net increase in annual willow offtake across the winter range above the baseline level of 35 percent. Progressive increase in mean willow height across the winter range above the baseline level of 0.9 m to at least 1.1 m. Progressive increase in willow cover across the winter range above the baseline level of 21 percent to at least 31 percent.

Winter willow offtake remained steady during the past 5 yrs, and although there were no substantial increases in offtake, there were also no consistent declines. On the core winter-range, willow offtake was below the baseline level of 35 percent only in 2013, and this decline was likely due to the loss of forage in Moraine Park due to the Fern Lake Fire and associated shift in habitat use out of Moraine Park in winter 2012–13. Offtake did not increase on the noncore winter range as a result of fencing large portions of willow habitat (fig. 13), but pressure on core winter-range habitats has not lessened overall either. It is possible that winter willow offtake has actually decreased during 2008–14, but that an increase in summer browsing has masked this change because the spring measurement method currently being used to monitor offtake only provides data on annual browse offtake and cannot distinguish season (winter as compared to summer) of browsing. A 2014 survey of summer browsing on the winter range indicated an average of 14.5 percent of twigs were browsed by late summer on the core winter range (Moraine Park, Horseshoe Park, and Beaver Meadows), whereas the noncore winter range was only very lightly browsed (about 3.7 percent of twigs browsed). These data indicate that summer browse levels are much greater than anticipated and may need to be addressed at some point in the future. It should be noted that percent offtake of current year growth would likely be lower when accounting for removal of biomass per twig, but summer browsing may have greater physiological effects on willows than winter browsing. Danell and others (1994) observed increased shoot size, leaf size, and leaf nutrient concentrations following winter browsing, but decreases in most of these variables following summer browsing of birch by moose.

By all metrics, willow heights have stayed at or above baseline levels of 0.9 m. Although there has been a slight increase in average willow heights above baseline measures during these first 5 yrs in the core winter range, this increase was due to height increases within fenced habitat (table 8). No willow height increases were observed in unfenced core winter-range areas, but noncore winter-range willows had a slight height increase, which may be indicative that lower populations of elk in the park during the past 5 yrs have also contributed to observed height increases on the winter range as a whole. The increased willow heights on the noncore and fenced core winter range have resulted in a 29-percent increase in average willow height on the entire winter range to 1.1 ± 0.1 m (mean ± standard error); therefore, the stated objective of increasing willow heights on the winter range by 10 percent has been achieved in this 5-yr span. These heights (1.1 m), however, are still much shorter than those being observed on the taller, less browsed, noncore winter range (1.6–1.8 m [160–185 cm], table 8). Knowing that this objective was achievable in a 5-yr timeframe, the park may want to adapt their stated objectives to try to continue 10 percent height growth each year until conditions on the entire winter range are closer to those seen on the noncore winter range at baseline.

The loss of both fenced and unfenced willows to the Fern Lake Fire in Moraine Park in 2012 may have masked height gains made prior to 2013 within these sites because burned willows were shorter in 2013 than unburned willows regardless of whether they were fenced (tables 8 and 9). Willows in the sites that burned were shorter at baseline than willows in sites that did not burn, however, so this effect is not completely attributable to the fire. Presuming that willows do recover in burned areas (or at least in fenced areas within the burn), it is likely that willow heights within fences will increase by the 2018 sampling based on the increases seen in fenced areas that were not burned.

As with willow heights, willow cover showed a 75-percent increase within the fenced areas since baseline measurement (table 8). The burned sites, on average, had much lower cover (about 5 percent cover) than unburned winter-range sites (roughly 30 percent cover, tables 8 and 9). Over the entire winter range, willow cover increased roughly 20 percent between 2008 and 2013 reaching an average of 24 ± 3 (± standard error) percent cover by 2013, which is considerable progress toward the objective of 31 percent willow cover.

Again, the loss of much willow to the Fern Lake Fire likely influenced the overall results for the core winter range and the recovery of willow in burned areas is still to be determined. Continued monitoring of the burned sites will be necessary to determine the longer-term impacts of the fire on these willow communities. It would be ideal to be able to compare willow in the three major valleys (Moraine Park, Horseshoe Park, and Beaver Meadows) of the core winter range, but the original sampling design did not provide for such comparisons and it would require a substantial effort and establishment of numerous additional sites to provide for such comparisons.

It should be noted that in some sites, where there is a potential for both willow and aspen, growth of one species may be limited by the other (fig. 18). In such cases, increases in one of the target species may not be observed due to increasing dominance of the other and should be evaluated on a case by case basis to determine if the sites are now better suited to be included in another sampling stratum.

Figure 18. Comparison of willow site in Beaver Meadows on the elk winter range of Rocky Mountain National Park, Colorado. Heavy aspen regeneration that has taken place in this site. *A*, prior to fencing (baseline measurement). *B*, after 4 yrs of protection from ungulate browsing (2013).

Current State of Elk-Vegetation Management on Winter Range

Management actions that have been employed since 2008 in Rocky Mountain National Park include temporary fencing of critical habitat to exclude elk and elk culling to maintain a target population size. After 5 yrs of these actions, there is progress toward the vegetation objectives of the EVMP, but this largely seems to be in response to fencing. Willow cover and heights have increased inside fences as has aspen regeneration. Many fences had been in place less than 5 yrs at the time of the 2013 sampling and, therefore, it is anticipated that increases in willow height and cover and aspen sapling density will continue into the 2018 sampling. Herbaceous offtake has decreased in some years, but this response seems to be correlated to growing-season precipitation rather than lower elk numbers.

The Fern Lake Fire, which burned much of Moraine Park in December 2012, added a dimension of complexity to the EVMP monitoring program but also has provided opportunities to examine the response of aspen and willow to burning. The use of prescribed fire as a potential management tool was included in the EVMP (National Park Service, 2007) and although the 2012 fire was neither prescribed nor intended as a habitat management tool, the presence of monitoring plots within burned and unburned areas will provide further insights into using prescribed fire as a tool in these vegetation types.

Similar to other studies (Kay and Bartos, 2000; Kay, 2001; Smith and others, 2001), aspen recruitment increased on core winter range primarily within elk exclosure fences. The high degree of suckering in burned aspen stands 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. Romme and others (1995) found high densities of sprouting aspen on burned sites for up to 2 yrs following fire in Yellowstone National Park, but by the end of the third year, sprout density in burned and unburned stands was similar; however, in areas where ungulate density was less than one animal per square kilometer, sucker densities were observed to remain high for several years following burning (Durham and Marlow, 2010). To determine aspen responses to both fire and herbivory, additional burned sites were added to the monitoring program in 2013. Annual monitoring of these sites for at least the next 5 yrs will provide information on the short-term effects of fire on aspen regeneration and recruitment in the presence and absence of elk herbivory. Keeping these sites in the 5-yr rotation of site measurements for the duration of the EVMP will provide information on long-term effects of fire and ungulate herbivory.

Herbaceous offtake did not respond linearly to reductions in elk population size on the winter range. There may be several contributing factors to this lack of response. There was a strong relation between upland herbaceous offtake and growing season precipitation. Plant primary production is strongly influenced by the amount and distribution of precipitation (Sala and others, 1988). In wet meadows of Rocky Mountain National Park winter range, herbaceous plants rely more heavily on precipitation water than groundwater (Alstad and others, 1999), and this is likely true to an even greater degree in these dry upland sites where water tables are farther from the surface. Furthermore, increased precipitation can mediate the effects of grazing on aboveground net primary production (Augustine and McNaughton, 2006), whereas drought can exacerbate the effects of grazers on aboveground net primary production and plant cover (Fahnestock, 1998; Fahnestock and Detling, 1999; Zeigenfuss and others, 2014). Consumption of plant biomass from a more productive plant will result in a lower percentage of biomass removed (percent offtake) than consumption of the same amount of biomass from a less productive plant. In this way, offtake (percent of biomass consumed) could actually decrease while utilization (mass of plant biomass consumed) stays the same.

Also, the elk population estimate only accounts for elk that winter in the park. A larger population of elk spend time on the winter range in spring and fall as they migrate from higher-elevation summer range in the park to lower-elevation winter range in the Estes Valley and foothills at the edge of the Colorado plains (Ben Kraft, Wildlife Biologist, Colorado Parks and Wildlife, oral commun., August 2014). These elk contribute to the offtake on the winter range and the timing and duration of their residence on the winter range varies based on weather conditions; therefore, if the number of elk wintering on the winter range decreased, but the number of elk using the winter range during spring and fall migration increased or the length of time spent on the winter range during migration increased, it could essentially cancel out the effects of a smaller wintering herd on overall offtake. Based on a visual assessment of upland areas in 2014, high variability in annual offtake, and lack of a strong correlation between offtake and elk population size, a recommendation was made by a committee of grazing and wildlife ecologists in September 2014 to discontinue collection of upland herbaceous offtake data.

Increases in average willow height and cover on the winter range within fenced sites indicate that fencing has been successful in allowing willow growth; however, the loss of sites to the Fern Lake Fire makes it difficult to discern how much change in willow growth may have taken place within fences in Moraine Park. Few studies of fire effects on willow in the Rocky Mountains are available for comparison, probably because intense fires are less likely to burn the lush vegetation associated with riparian areas where large patches of willow are found. Norland and others (1996) determined that resprouting burned willows in Yellowstone National Park had higher protein levels and greater digestibility, longer and heavier shoots, and greater leaf surface area than unburned willows; however, their height did not increase, likely due to intense browsing on these highly nutritious plants. It is possible that resprouting willows in Rocky Mountain National Park may also be subject to high levels of herbivory due to increased nutritional value of burned plants. Current studies of Moraine Park willow have determined that 45 percent of studied individuals had resprouted following the fire, but that those burned plants subject to browsing had 64 percent less biomass than unbrowsed, burned willows (Kaczynski and Cooper, 2015).

At some point, recovery inside the fences will likely plateau as available resources (space, light, water) are used. The density of living willow is low in some areas, including fenced areas, so growth of existing willow may not be enough to meet goals for willow cover. Some of these areas also lack suitable sites for willow establishment due to competition with herbaceous species, so willow may need to be planted to reach goals. In addition, if willow outside exclosed areas do not recover, even with reduced numbers of wintering elk, it may also be necessary to fence more willow, which may be particularly true in Moraine Park where recovery of willow from the Fern Lake Fire is compounded by intense browsing of new growth in burned areas. Annual offtake sampling of burned willow sites will provide information on response of offtake rates to burning. Annual monitoring of willow height and cover on burned plots, both fenced and unfenced, for the next 5 yrs, coupled with the data from the Kaczynski and Cooper (2015) study, will provide information on the short-term effects of fire on these willow communities. Continuing to monitor these sites as part of the regular monitoring program rotation after 2018 will provide data on the long-term responses of willow communities to fire.

Meaningful reduction of willow offtake seems to be the most difficult of the vegetation goals to achieve. As in upland sites, willow offtake is tied not only to overall number of elk on the winter range but their distribution and the duration of their residence on the winter range, which in turn can be driven by other factors such as temperature, annual and seasonal precipitation, snowfall, and forage productivity and availability. However, weather conditions may cause summer offtake to be substantial in some years (for example, late snows might keep elk on the winter range into the calving season and then for a post-parturition period extending into the summer, or drought may cause elk to abandon summer range earlier because of low forage availability). Alternatively, portions of the elk herd may alter their behavior over time, which could lead to decreased migration to the summer range, despite management actions intended to encourage seasonal migration. If more elk begin to summer on areas that have traditionally been winter range, park managers may feel the need to begin to evaluate summer willow offtake annually. Kaczynski and Cooper (2015) observed summer offtake on willows in Moraine Park in 2013, and data collected in summer 2014 indicated that elk are browsing in summer throughout the core winter range.

In conclusion, the elk-vegetation management program at Rocky Mountain National Park seems to be making slow but steady progress toward the vegetation goals set out by the EVMP. At this first evaluation point, it seems highly likely that plan objectives can be met by the end of the EVMP timeframe in 2028, mainly through increases in plant density, height, and cover within the fences; however, it might require additional fencing of willow and aspen to achieve these objectives, as well as planting and protecting willow in some locations. The EVMP called for fencing of as much as 105 ha (260 acres) of willow and 64 ha (160 acres) of aspen on the winter range (National Park Service, 2007). At the time of this analysis (2014), roughly 26 percent of willow on the entire winter range was fenced, all of it on the core winter range. Sixty-five percent of the potential amount of willow area that could be fenced under the EVMP has already been fenced. Ten percent of all aspen on the winter range were fenced by 2014, representing 25 percent of the potential amount of aspen that might be fenced under the EVMP. A substantial amount of willow and aspen could still be fenced if progress toward vegetation objectives remains limited outside fenced areas. The dedication of the Rocky Mountain National Park resources management staff to the collection of monitoring data has greatly helped to compensate for the difficulties presented by nature and provides the opportunity to potentially learn from these events.

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Appendix

Table 1–1. Willow offtake using alternate stem-scaled diameter-difference (DD2) calculation method (Bilyeu and others, 2007).

Year	Percent offtake (mean ± standard error)		
	Core winter range	Noncore winter range	Entire winter range (weighted average)
2009	34.4 ± 2.9	18.0 ± 3.2	32.3 ± 2.2
2010	43.4 ± 4.2	22.2 ± 6.3	40.2 ± 3.6
2011	34.5 ± 7.7	20.9 ± 3.1	30.5 ± 4.4
2012	44.5 ± 4.8	19.1 ± 2.6	37.2 ± 3.6
2013	26.1 ± 6.6	11.6 ± 2.8	19.9 ± 3.7
2014	33.4 ± 3.6	19.4 ± 5.4	30.5 ± 3.0

Table 1–2. Least squares means of willow cover and heights based using line intercept method in Zeigenfuss and others (2011).

[cm, centimeter; s.e., standard error; yr, year]

Winter range zone	Willow height (cm)			Maximum willow height (cm)			Willow cover (percent)		
	Baseline (mean ± s.e.)	2013 (mean ± s.e.)	*P*-value (yr)	Baseline (mean ± s.e.)	2013 (mean ± s.e.)	*P*-value (yr)	Baseline (mean ± s.e.)	2013 (mean ± s.e.)	*P*-value (yr)
Core winter range, unfenced	85 ± 18	89 ± 18	0.810	121 ± 25	116 ± 25	0.841	20 ± 4	22 ± 4	0.460
Core winter range, fenced	47 ± 12	83 ± 12	<0.001	60 ± 14	98 ± 14	<0.001	17 ± 5	24 ± 5	0.213
Noncore winter range	143 ± 23	200 ± 23	0.001	185 ± 28	239 ± 28	0.002	29 ± 5	37 ± 5	0.008