Original Article

Fertilizing Western Rangelands for Ungulate Conservation: An Assessment of Benefits and Risks

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ABSTRACT Intensive natural gas development causes habitat loss that reduces nutritional carrying capacity for ungulates and other species of conservation concern. To offset habitat loss from energy development, wildlife managers are experimenting with large-scale sagebrush fertilization on western public rangelands. We synthesize what is known about basic sagebrush ecosystem biogeochemistry and ungulate nutritional ecology to anticipate the benefits and risks of this emerging mitigation tool. Under some environmental conditions, fertilization can temporarily increase sagebrush biomass, and potentially, the nutritional carrying capacity of mule deer limited by digestible energy. However, nitrogen additions are costly and pose environmental risks when applied to arid and semi-arid systems. Fertilization may promote weed invasions, changes to biogeochemical cycles, and potentially irreversible ecosystem shifts that reduce shrub communities. Fertilization also creates excess nitrogen at $2–8 \times$ natural background levels and increases greenhouse gases, ozone pollution precursors, downwind nitrogen deposition, and freshwater degradation. Considering these risks and recognizing that potential benefits are short-term, uncertain, and expensive to achieve, the practice of sagebrush fertilization may pose a net conservation cost.   2015 The Wildlife Society.

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Large expanses of sagebrush (*Artemisia* L.) steppe in the western United States have been lost or modified as a result of energy development (Davies et al. 2011, Naugle 2011). Technological advances such as hydraulic fracturing and the prioritization of domestically produced, clean-burning energy have increased the land area affected by infrastructure and human activity (Walston et al. 2009). In the Intermountain West alone, oil and gas development is projected to directly impact 2.3–5.5 million ha by 2027, the majority of which is public land dominated by sagebrush (Copeland et al. 2009). As a result of these landscape-scale changes, iconic and economically important mule deer (*Odocoileus hemionus*) populations are losing critical habitat (Sawyer et al. 2009) at a time when populations are declining in many areas of Intermountain West (Mule Deer Working Group 2013).

Many migratory mule deer herds in temperate regions congregate in high densities on lower elevation wintering grounds where they have greater access to snow-free vegetation (Gilbert et al. 1970). When these winter ranges are converted into producing gas fields, mule deer experience the cumulative effects of both habitat loss and disturbance associated with development activities, which can displace individuals from preferred habitat (Fig. 1; Sawyer et al. 2006, 2009). Because mule deer show high fidelity to winter ranges, on-site mitigation techniques are a priority.

As a result of these and other disturbances to declining sagebrush communities (Davies et al. 2011), managers are searching for ways to reclaim and enhance sagebrush communities to mitigate impacts from energy development. Given the relatively short drilling phase of most natural gas projects, on-site mitigation in the form of habitat treatments has been suggested as a way to ameliorate impacts until permanent reclamation can occur (Northrup and Wittemyer 2013). Managers have traditionally relied on mechanical thinning and herbicide application to enhance sagebrush vigor, but the efficacy of these treatments in improving wildlife habitat is marginal (Beck et al. 2012). Resource managers are now revisiting the idea of sagebrush fertilization as a means to improve wildlife habitat (USDI BLM 2011*b*). Sagebrush fertilization was initially explored in the

Historically, approximately 5,000 migratory mule deer congregated each winter on the 2,200-m-elevation sagebrush steppe of the Upper Green River Basin in western Wyoming, USA. But as natural gas development accelerated, mule deer numbers declined by >40% between 2001 and 2010 (Sawyer and Nielson 2011). The population decline triggered a federal on-site mitigation requirement, prompting the Bureau of Land Management and Wyoming Game and Fish Department to initiate a pilot sagebrush fertilization project to improve forage for wintering mule deer (USDI 2011a).

In 2010, adjacent plots of approximately 100 ha in the northwestern portion of the project area were fertilized with 45 kg/ha (40 lbs/ac) or 90 kg/ha (80 lbs/ac) of pelleted urea $(CO(NH_2)2)$-N via helicopter. Locations near aquatic resources or with known invasive weeds were avoided. In 2011, approximately 405 ha were fertilized at the lower application rate (Map 1). Cost of fertilizer and aerial application was approximately US$135/ha (USDI 2012a). Annual fertilization of up to 12,164 ha on the flanks of the project area has been authorized contingent on results from the pilot study, a projected annual expense of >US$1.6 million (USDI 2011a).
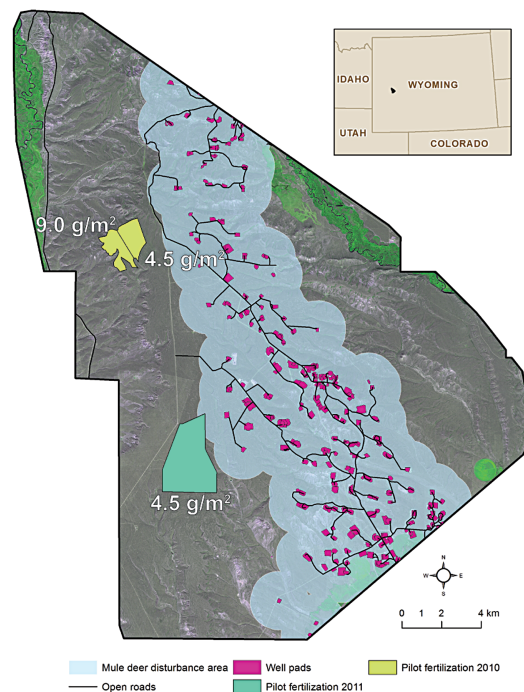
Map 1. The northern extent of the Pinedale Anticline Project Area natural gas development, including direct habitat loss (wells and roads), indirect habitat loss (disturbance area; Sawyer et al. 2009) and fertilization experiments.

Figure 1. A case study in sagebrush fertilization.

1970s, but never developed into a practical or effective tool for large-scale habitat improvement (e.g., Bayoumi and Smith 1976, Barrett 1979, Carpenter and West 1987).

Today, this idea is again being tested on a large-scale fertilization project in the Upper Green River Basin of Wyoming, part of the Greater Yellowstone Ecosystem, where natural gas development has affected the sagebrush winter ranges of mule deer (Fig. 1). Here, mule deer population declines following energy development triggered a requirement in the 2008 Record of Decision for the Pinedale Anticline Oil and Gas Exploration and Development Project (USDI BLM 2008) for on-site mitigation. Along with traditional habitat treatments, managers have initiated a pilot fertilization study in an attempt to enhance sagebrush growth, nutrient quality, and palatability (Fig. 1; USDI BLM 2011a).

Pending results from the 2008 pilot study, a Federal Record of Decision has authorized fertilization of up to 12,164 ha annually in the Pinedale Anticline portion of the Upper Green River Basin (Fig. 1; USDI BLM 2011b). Federal approval is also underway to use sagebrush fertilization to mitigate natural gas development impacts in the Continental Divide–Creston natural gas development near Rawlins, Wyoming (USDI BLM 2012b). As domestic natural gas production increases in sagebrush steppe of the western United States, fertilization of public lands could become more widespread.

Nitrogen fertilization is typically practiced in humid, subhumid, and irrigated areas for the purpose of increasing crop production, forage yield, or protein yield. Such practices have effectively doubled inputs of biologically available nitrogen worldwide, with a number of well-recognized negative consequences (Vitousek et al. 1997). Our understanding of the effects of these inputs on semi-arid ecosystems, and ultimately, on the capacity to improve mule deer population performance, is much less clear. Thus, an assessment of likely ecological benefits and costs of nitrogen fertilization as a mitigation strategy is prudent and timely.

Here, we use an interdisciplinary approach to synthesize what is known about 1) mule deer winter nutrition in sagebrush-steppe ecosystems; 2) the fate of nitrogen fertilizer in these ecosystems, including the necessary conditions for

uptake by sagebrush plants; 3) the potential effects of sagebrush fertilization on mule deer population performance; and 4) the risks of arid-land fertilization.

MULE DEER NUTRITION ON WINTER RANGE

Experimental evidence shows that in some areas, bottom-up factors affecting nutrition govern mule deer population performance (Bishop et al. 2009, Pierce et al. 2012); thus, habitat loss and alteration that reduce foraging opportunities are a particular management concern. Attention has rightly focused on critical summer and transitional ranges that may determine autumn body condition, and ultimately, probability of overwinter survival (Mautz 1978, Tollefson et al. 2011, Monteith et al. 2013). However, habitat loss on already geographically limited winter range may reduce carrying capacity (Sawyer et al. 2006) during a season when forage is limited and unnecessary energy expenditures are costly (Wallmo et al. 1977). As a result, winter nutrition continues to be a management concern.

Shrubs are important nutritional sources in temperate regions, especially where snow cover makes other forage inaccessible (Wallmo et al. 1977). In the sagebrush-steppe ecosystems of the Intermountain West that provide crucial winter range for many mule deer herds, big sagebrush (*A. tridentata* spp.) is often the dominant shrub and can comprise >50% of the mule deer diet (Kucera 1997). Generally, big sagebrush is a high-quality resource preferentially browsed by mule deer even when other forage options are available (Welch and Wagstaff 1992). *In vitro* digestibility of sagebrush varies across locales from 50% to 64% (Wallmo et al. 1977, Kufield et al. 1981, Welch and Pederson 1981), approximating the 50–60% digestible energy necessary for maintenance energy requirements for deer (Ammann et al. 1973, Swift 1983). Even in winter, sagebrush has high dry matter, caloric values, crude fat, and essential minerals (Welch and Pederson 1981). However, depending on the region, time of year, and overall diet composition of individual mule deer, a winter diet comprising mainly sagebrush may still be nutritionally marginal. For instance, big sagebrush can be rich in crude protein (8.3–14.5%; Dietz 1965), but some evidence indicates that protein is less accessible during winter months (Barrett 1979). This is supported by low levels of fecal nitrogen in sagebrush-fed mule deer, suggesting a protein deficit in some populations (Short et al. 1966).

The digestibility and palatability (defined generally as the characteristics of the plant that make it a preferable forage option for the animal; McKell 1989) of sagebrush by ruminants is a subject of debate (e.g., Ngugi et al. 1995, Welch 1997). Traditionally, sagebrush was considered to be a poor forage for ruminants because of the presence of terpenes, which were thought to reduce palatability and impede *in vitro* digestibility by impairing microbial processes in the rumen (e.g., Carpenter et al. 1979, Ngugi et al. 1995). However, *in vivo* studies of sagebrush-adapted herbivores show that terpenes occur in much lower content in the rumen than what was initially ingested, suggesting those compounds are volatized during chewing or digestion (Estell 2010). With palatability and digestibility of sagebrush undecided, it is unclear whether there is an upper limit to sagebrush ingestion by mule deer (Cluff et al. 1982).

More likely, mule deer nutrition on winter range is most constrained by sheer forage quantity, which limits available digestible energy (Dietz 1965). Winter ranges tend to be geographically confined, with high densities of mule deer, pronghorn, livestock, and other ungulates capable of grazing >90% of sagebrush leaders throughout the year (Berteaux et al. 1998). Nutritional carrying capacity may also be limited by geographic location and the aggregate diet composition, deep snow that limits access to forage in the winter (Wallmo et al. 1977), and low precipitation that reduces annual growth of shrubs (Carpenter et al. 1979, Austin and Urness 1985). Under these conditions, forage quantity is likely much lower than necessary to sustain historical herd sizes, particularly when combined with habitat loss.

Whether fertilization can improve nutritional carrying capacity of affected winter range depends on what nutrients are limiting to mule deer and whether fertilization can boost those nutrients sufficiently to improve population performance. These are open questions best answered by first understanding the fate of nitrogen in semi-arid landscapes.

THE FATE AND EFFECT OF NITROGEN IN SAGEBRUSH-STEPPE ECOSYSTEMS

For nitrogen (N) fertilization to improve sagebrush winter habitat, a number of conditions are necessary (Fig. 2). First, the additional N must be taken up by woody or herbaceous plants. Nitrogen absorbed by plants must then result in a higher quantity or quality of forage relative to unfertilized sagebrush, and this enhancement must last through the winter and be available to browsers. The biogeochemistry behind each of these steps is poorly understood, but a N budget can be pieced together to provide a framework for anticipating the effects of N additions (Table 1).

Typical of arid and semi-arid ecosystems, the quantity of N stored in pools (plant biomass, litter, soil) is much greater than the N fluxes (i.e., the amount cycled within, added to, or lost from the ecosystem on an annual basis). Although sagebrush ecosystems store 2,500–3,500 kg N/ha in plant biomass and soils to a depth of 15 cm, the rate of N released from soils in a form available to plants (N mineralization), is estimated at only 5–30 kg N/ha/year (Burke 1989). Data for inputs (atmospheric deposition and N fixation) and outputs (N gas emissions or leaching in groundwater) are limited, but are estimated to be 0–40 kg N/ha/year (inputs) and 10–20 kg N/ha/year (outputs; Table 1). Hence, fertilization rates of 40–80 kg N/ha/year represent an increase of 2–4 × relative to the background N inputs by atmospheric deposition and N fixation, and 2–8 × higher than the background N release (mineralization) rate (Table 1). What are the potential fates for this excess N?
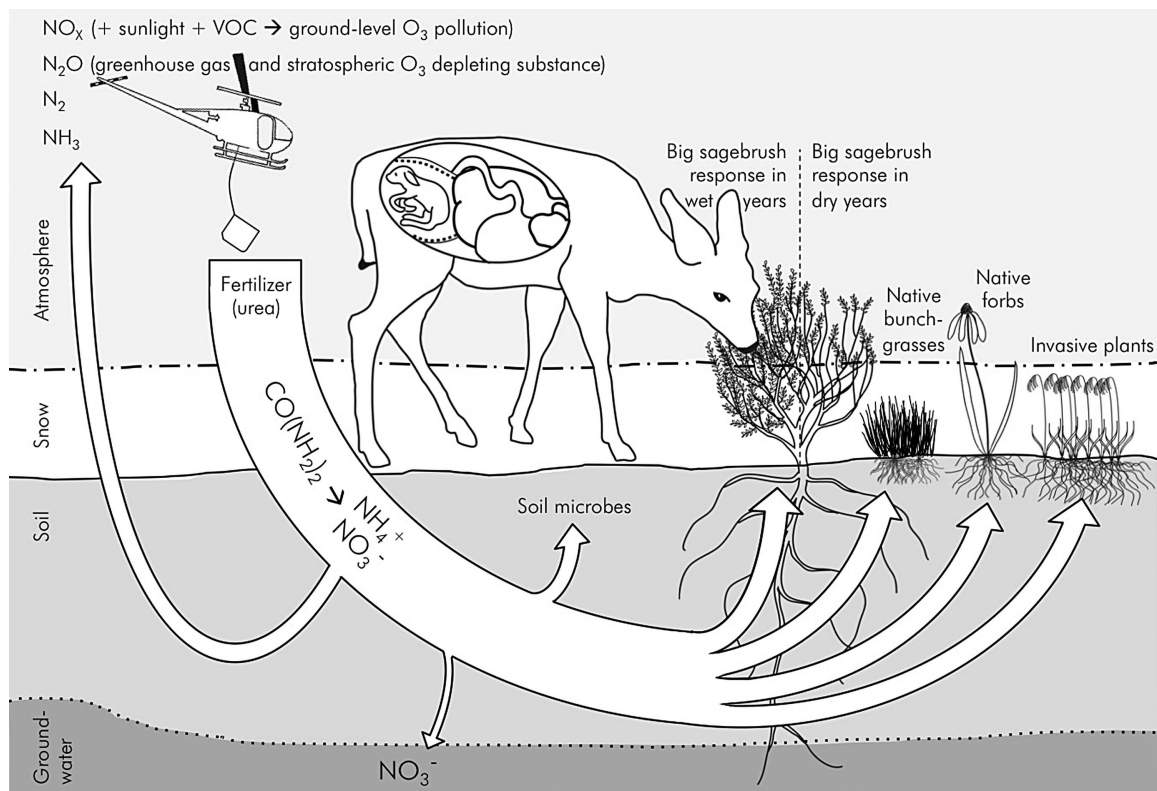
Figure 2. Conceptual model of multiple potential nitrogen pathways following sagebrush fertilization. For fertilization to boost mule deer population performance, nitrogen must improve sagebrush quantity or quality sufficiently to increase survival of mule deer adults or fawns (VOC: volatile organic compounds).

Hydrolysis and Volatilization

Urea is made available to plants and microbes following hydrolysis to ammonia (NH_3; Fig. 2). This ammonia gas can be lost to the atmosphere (volatilized), unless it dissolves in water to form ammonium (NH_4^+), a plant-available nutrient. A risk of autumn application of urea is that it may be hydrolyzed to ammonia and lost to the atmosphere before spring when ammonium is needed for plant growth. At least 20% of urea fertilizer added to upland soils is lost as gaseous NH_3 (Schlesinger and Hartley 1992), and likely more in the neutral-to-alkaline pH soils of semi-arid shrublands. Emissions of NH_3 lead to downwind N deposition, which is projected to be the major source of nitrogen load exceedances in U.S. national parks and other protected Class I airsheds in the future (Ellis et al. 2013).

Gaseous Losses From the Biological Soil N Cycle

Ammonium N is further converted to another plant-available nutrient, nitrate (NO_3^-), via the process of nitrification. Intermediate steps in this pathway result in N losses from the ecosystem via emissions of nitrogen oxides (NO and NO_2, collectively known as NO_X) and the greenhouse gas nitrous oxide (N_2O; Hall et al. 1996). The average NO loss following fertilization is typically <1% of the added N, but may be much higher when fertilizer is applied as urea prior to the dormant season when plant demand is low (Hall et al. 1996). Emissions of NO_X and N_2O from another biogeochemical pathway (denitrification of nitrate to N_2 gas) may be exacerbated in carbon-poor arid soils.

Nitrogen fertilization can increase harmful N gas emissions and reduce soil greenhouse gas uptake, thus boosting concentrations of ozone precursors and greenhouse gases in the atmosphere (reviewed in Mosier et al. 1996). NO_X is a regulated air pollutant that, under specific environmental conditions, can react with volatile organic compounds (elevated by gas production operations) to form the tropospheric air pollutant ozone (O_3; Carter and Seinfeld 2013); this is an emerging problem in some natural gas developments such as the Upper Green River Basin, where air pollution can rival that in urban areas (Schnell et al. 2009).

Hydrologic Loss

Together with leaching and infiltration, surface runoff of agricultural fertilizer is a major source of N contamination in lakes and rivers (Carpenter et al. 1998). In a semi-arid climate in which <10% of precipitation goes to deep drainage, little N is likely to be leached in an average year. However, during higher precipitation years, notable deep-water drainage can occur (Schlaepfer et al. 2012), with potential consequences for freshwater acidification and eutrophication (Vitousek et al. 1997).

Soil

Nitrogen that makes it into soil (i.e., is not lost to gaseous emissions) tends to stay there; studies that follow N through ecosystems recover much more in soil and soil microbes than in vegetation (Templer et al. 2012). Especially if urea-N is redistributed by wind to beneath the canopy of sagebrush where soil organic matter concentrations are higher, uptake

Table 1. Ecosystem nitrogen balance of semiarid Wyoming, USA, big sagebrush ecosystems, as estimated from a variety of sources (min. and max. values rounded to one significant digit).

Ecosystem nitrogen	Component of the N cycle	Range for native sagebrush ecosystems[a]	Likely response to added N
Inputs: kg N/ha/year	Deposition	0–20[1]	No change
	Fixation	0–40[2]	Decrease
Pools: kg N/ha	Shrub aboveground biomass	20–100[3,4]	Increase
	Other plant biomass	10–20[4]	Increase
	Litter	100–200[4,5]	Increase
	Soil (0–15 cm)	2,000–3,000[2,6]	Increase or no change
	Soil microbial biomass	200–300[7]	Increase
Internal fluxes: kg N/ha/year	Production	2–6[8,9]	Increase
	Litterfall	2–100[5,10,11]	Increase
	Litter decay	0–70[5,10,11]	Decrease
	Net mineralization	5–30[6]	Increase
Outflows: kg N/ha/year	Trace gas emissions	−10 to −20[1,2,12–15]	Increase
	Leaching	0[16]	0; No change

[a] Source references: [1] Matson et al. 1991; [2] Charley 1977; [3] Pearson 1965; [4] Cleary et al. 2010; [5] West 1983; [6] Burke 1989; [7] Burke et al. 1989; [8] Pearson 1965; [9] Harniss and Murray 1973; [10] Mack 1977; [11] Woodmansee et al. 1978; [12] Parton et al. 1988; [13] Smart et al. 1999; [14] Schlesinger and Hartley 1992; [15] Stark et al. 2002; [16] Schlaepfer et al. 2012.

of N into soil microbial biomass and soil organic matter (i.e., immobilization) would be favored.

Vegetation Uptake

Any available N that is not immobilized, or is mineralized from soil, is available to plants which, given sufficient water, will compete with one another for that N. A change in N availability, such as is caused by fertilization, alters competitive dynamics and plant community composition. Specifically, N increases forage quantity and protein yield of cool-season grasses, such as invasive cheatgrass (*Bromus tectorum*), relative to warm-season species and forbs (Goetz 1969). Forbs are rare relative to grasses and shrubs in these systems; therefore, they are more likely to be lost both because of their initial low abundance, but also because of competitive effects of faster growing grasses (Suding et al. 2005). Especially when applied in autumn or winter, fertilization may favor plant species that are active earlier in the spring (e.g., sagebrush and also cheatgrass; Witwicki et al. 2013).

In other semi-arid systems, N additions have been shown to promote invasion by exotic plant species and long-term changes in plant composition (Milchunas and Lauenroth 1995, Vinton and Burke 1995, Fenn et al. 1998). Invasive weeds such as cheatgrass tend to be highly responsive to nitrogen; thus, N addition can exacerbate invasions caused by surface disturbance (e.g., from well pads, roads, and pipelines). A consequence of increased invasive grasses in semi-arid systems is an increase in fire frequency (Rao et al. 1982). In a shortgrass steppe experiment, additions of N and water altered plant community structure, an effect that persisted over decades. N additions reduced dominance of native species and increased exotic species including *Bassia scoparia* (kochia), creating a feedback that increased soil nitrogen availability (Milchunas and Lauenroth 1995, Vinton and Burke 1995, Burke et al. 2013). Similarly, application of biosolids to a sagebrush-steppe ecosystem disturbed by oil shale development in the 1970s showed persistent effects to soil fertility and a long-term reduction in

sagebrush, dependent on initial soil conditions (Paschke et al. 2005). Evidence that up to 15% of N from agricultural fertilizers can persist in soils for 30 years following application (Sebillo et al. 2013) suggests that a long-lasting N pool may continue to drive ecosystem changes for decades after application. Semi-arid systems are susceptible to irreversible changes in N cycling and species composition as a result of relatively small (Fenn et al. 1998) or short-duration N pulses.

Allocation of N in Sagebrush

The most common effect of N additions on big sagebrush is an increase in leader length (new annual growth), although treatment effects are variable. In studies of fertilization rates similar to those used in the Upper Green River Basin (45–90 kg N/ha; Fig. 1) leader growth was 30–103% greater for fertilized than for unfertilized plants (Bayoumi and Smith 1976, Barrett 1979, Carpenter and West 1987). No treatment effect was observed at N application rates ≤31 kg N/ha in one study (Carpenter and West 1987) and treatment effects at N concentrations >84 kg N/ha were inconsistent in 2 studies (Bayoumi and Smith 1976, Barrett 1979).

Sagebrush response to nitrogen addition is highly dependent on timing and amount of precipitation, with a greater growth response during wet years and minimal response in dry years (Hooper and Johnson 1999). At moderate fertilizer application rates, benefits to new leader growth decline sharply beginning in the second year after fertilization, requiring annual application to maintain treatment effects (Bayoumi and Smith 1976, Barrett 1979). Fertilization does not appear to have a significant effect on concentrations of terpenoid compounds and subsequent palatability of sagebrush (Sneva et al. 1983). Some evidence suggests that fertilization can increase foliar crude protein of sagebrush from 14% to 43% relative to controls. However, crude protein levels in fertilized plants do not differ from controls when sampled in autumn (Barrett 1979, Witwicki et al. 2013), suggesting that additional N is either allocated to

deciduous leaves and then lost with the shed leaves or translocated to less digestible woody portions of the plant in the autumn (Dietz 1965, Knight 1994). In either case, there is little increase in winter forage quality in terms of available protein content. If crude protein is limiting to mule deer, a better understanding of the contribution of fertilizer N to crude protein of sagebrush is important.

MULE DEER RESPONSE TO ENHANCED FORAGE

If all of the biogeochemical factors required for fertilization to promote additional leader growth are met, the next requirement for successful mitigation is for individual animals to find and utilize enhanced sagebrush plants. Ungulates are able to sense and exploit forage with higher nutritional value (Swift 1948, Barrett 1979) and have been shown to preferentially browse in fertilized versus control plots of sagebrush (Berteaux et al. 1998, Wambolt 2004). Mule deer should thus preferentially browse fertilized plants if they are nutritionally superior to controls.

Whether fertilization is likely to fill a nutritional deficit depends on whether protein or energy is the limiting factor. If limited by protein, fertilization is unlikely to fill a nutritional deficit given the minimal overwinter effect of fertilization on crude protein in leaves and stems (Barrett 1979, Witwicki et al. 2013). If limited by digestible energy, fertilization may fill a nutritional deficit when sufficient precipitation allows for greater leader growth and given that new growth has not already been browsed at the onset of winter when migratory mule deer arrive. However, this benefit assumes there is no upper limit to sagebrush ingestion due to inhibition of microbial processes. At 33.6 kg/ha of ammonium nitrate fertilization, an additional 73.4 kg/ha of sagebrush forage produced (Bayoumi and Smith 1976, Barrett 1979) could support an additional 52.4 deer-days/ha (assuming a 65-kg F), representing a potential increase in nutritional carrying capacity.

Ultimately, fertilization will have mitigation value to mule deer if nutritional enhancements improve the body condition of individual animals sufficiently to boost key vital rates. In long-lived mule deer, population growth rate is most sensitive to survival of adult females (Gaillard et al. 1998), but may also be driven by higher fawn survival, a vital rate that shows greater variability and responsiveness to treatments than adult survival (Bishop et al. 2009). An *ad libitum* artificial diet was shown to increase population growth rate of mule deer through higher fetal survival, overwinter fawn survival, and annual survival of adult females (Bishop et al. 2009). The effects of habitat improvements on mule deer nutrition are no doubt less pronounced than the effects of an artificial diet, but have not been well-studied. However, Bergman et al. (2014) found that removal of encroaching pinyon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*), together with shrub reseeding and weed control, improved habitat quality enough to increase survival of mule deer fawns in Colorado. We note that these examples show the potential for demographic responsiveness to nutritional improvements in the absence of important anthropogenic factors such as energy development, which might lessen habitat treatment effects. Although increased leader growth from sagebrush fertilization could potentially provide additional calories, the realized demographic benefits to mule deer have not been assessed.

DISCUSSION

The semi-arid rangelands of the western United States occupy a nexus of world-class energy reserves and iconic wildlife species. Effective techniques for mitigating negative effects of energy development in sagebrush ecosystems are needed now more than ever. Whether sagebrush fertilization is a worthwhile strategy for sustaining wildlife alongside domestic natural-gas development, however, requires a full accounting of the benefits and risks, some of which are only poorly understood. Our systems approach to understanding the trade-offs of rangelands fertilization reveals a set of risks and costs not necessarily evident from a single-species approach.

The proximate goal of fertilization is to increase the quantity, quality, and palatability of sagebrush plants (USDI BLM 2011a). Under some environmental conditions, sagebrush fertilization may increase forage quantity through greater leader growth, which could improve nutritional carrying capacity of treated areas. However, potential nutritional benefits are transitory, likely lasting 1 or 2 years postapplication and requiring repeated, expensive N additions (e.g., US$135/ ha; Fig. 1). Any boost in sagebrush crude-protein content following fertilization disappears from edible plant parts by the crucial winter months (Barrett 1979, Witwicki et al. 2013). Likewise, there is limited evidence for any effect of fertilization on concentration of terpenoid compounds (Sneva et al. 1983) and still debate as to whether those compounds reduce palatability or digestibility to begin with.

Notable scientific unknowns exist at every step of the nitrogen pathway (Fig. 2). The timing, amount, and form of precipitation necessary for sagebrush growth response are unresolved, as well as whether sagebrush or invasive species are more likely to benefit from fertilization. Once captured by sagebrush plants, the role of N in creating digestible energy versus protein, seasonal translocation of N within the plant (with implications for availability to browsers), and the magnitude of fertilizer effects on leader growth are still uncertain. Whether enhanced sagebrush results in the ultimate goal of improved population performance of mule deer depends on whether digestible energy or protein is limiting, whether mule deer are capable of digesting large quantities of sagebrush efficiently, and the magnitude of the effect on mule deer fitness—all questions that remain unanswered. In short, the benefits of sagebrush fertilization are uncertain, transitory, and costly.

The potential benefits of rangeland fertilization are further tempered by ecological risks. Fertilization creates the conditions favorable to plant community shifts by promoting annual grasses and invasive plant species while reducing forbs. These unintended outcomes could diminish sagebrush habitat quality and undermine the goal of fertilization. As occurs in agricultural systems where fertilization is traditionally practiced, fertilizer nitrogen could contaminate

streams, rivers, lakes, and groundwater in wet years in these semi-arid rangelands. And finally, the creation of air-polluting byproducts is a potential side effect of annual N application at large spatial scales. Given these uncertainties and risks, the ecological trade-offs of sagebrush fertilization are substantial and may represent a net conservation cost.

MANAGEMENT IMPLICATIONS

The capacity of rangeland fertilization to improve mule deer habitat is uncertain and contingent on a complex biogeochemical pathway. Given the high monetary cost, short duration of treatment effects, and suite of ecosystem risks, there is little evidence to indicate that fertilization is an effective or sustainable mitigation approach. Accordingly, we encourage managers to consider alternative mitigation options. The concerns associated with rangeland fertilization do not, however, justify artificial feeding of mule deer, which has its own well-documented drawbacks, including disease transmission, expense, and ethical issues. Instead, the challenge of mitigating habitat loss *post hoc* reinforces the importance of avoiding impacts to irreplaceable mule deer winter range through improved planning and siting. Where that is not possible, protection and effective management of summer and transitional ranges may provide nutritional opportunities that better allow animals to survive difficult winter conditions.

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