

# **Ecological Impacts of Energy Development and other Human Disturbances in a Rangeland Ecosystem**

Cisneros-Pineda, Alfredo; Aadland, David; and Tschirhart, John

Using a general equilibrium ecosystem model, we estimate the ecological impacts of the Atlantic Rim Natural Gas project in south-central Wyoming, USA. The impacts are measured in terms of species' populations and value of ecosystem services such as species' existence value, hunting net benefits, and cattle-grazing profits. The simulation tracks the biomass consumption patterns and population changes of 13 species in the sagebrush ecosystem. The results are driven by losses of habitat for species, as well as changes in the relative scarcity of prey and forage when species avoid the areas disturbed by energy development (ED). For the study area, the total present-value loss in ecosystem services from ED after coordination among government agencies, and under two different species' population constraints, is between \$60 and \$90 million.

***Keywords:*** ecosystem modelling, economic/ecological integration, sage brush ecosystem, biodiversity, habitat loss, energy development, ecosystem services, benefit transfer

# 1 Introduction

Energy development (ED) in the Intermountain West is proceeding at a rapid pace to meet the increasing local, national, and international demand for energy associated with economic growth. However, ED is also accompanied by externalities whereby people can be negatively impacted in ways for which they are not compensated. One important externality, sometimes referred to as an ecosystem externality (Crocker and Tschirhart 1992), occurs when development alters the equilibrium of an ecosystem and the ecosystem services it provides.

The objective of this research is to identify the ecosystem externalities and change in ecosystem services that result from ED. We do this by tracking the populations of the relevant species in a rangeland ecosystem and estimating changes in the value of three ecosystem services after natural gas development is introduced. Our model shows the complex relationship between ED, biodiversity, and ecosystem services. While species will experience immediate habitat loss from ED, it is not clear to what degree the species will be negatively affected given the ecosystem interactions. We are interested in the degree to which species' behavior, reproduction, and populations in the rangeland ecosystem are impacted by ED, with a focus on the mechanisms leading to these impacts.

The impact on biodiversity is measured by considering the relative abundance of prey and forage in areas disturbed or undisturbed by ED. Studies indicate that species such as the greater sage-grouse (*C. urophasianus*), elk (*C. canadensis*), and mule deer (*O. hemionus*) avoid natural gas wells and the area surrounding them (e.g., Walker 2007; Doherty 2008). To include this effect, we assume that some species do not forage in the area around the natural gas wells. Species' populations are then affected over time in complicated ways because of differential forage and prey availability between disturbed areas (DAs) and undisturbed areas (UAs). Our

approach tracks how changes in prey/forage availability impacts the accumulation of individual organisms' net energies, biomass consumptions, predation rates, natural death rates, and birth rates. The combination of these impacts provides a rich prediction of how ED affects an ecosystem and the associated ecosystem services. The difference in habitat between DA and UA is similar to source-sink habitats examined by Kirol et al. (2015); however, unlike those authors we do not examine other direct affects of ED on species such as loss of breeding grounds or road and fence construction.

Northrup and Wittemyer (2013) reviewed the literature on the ecological ramifications of the disturbances caused by development of alternative and unconventional sources of energy. They advocate that “future studies [should] take a comprehensive approach incorporating a mechanistic understanding to the interplay between development-caused impacts and species ecology that will enable effective mitigation.” (p. 112) This type of comprehensive, integrated approach is the primary contribution of our paper. Our simulation results show that the introduction of natural gas development into a rangeland ecosystem generally has negative effects on biodiversity, although these effects can be mitigated to some extent by adjusting the intensity of other human disturbance such as livestock grazing and hunting.

ED is introduced into the rangeland ecosystem model using an integrated economic and general equilibrium ecosystem model (GEEM). The primary advantage of GEEM (Tschirhart 2000) is that it allows a more complete measure of the ecosystem tradeoffs associated with expanding energy production, rather than just relying on the population effects of one indicator species. In GEEM, the dynamics of plant and animal populations are consistent with ecological principles but modeled with microeconomic methods. Individual plants and animals are assumed to behave as if they maximize the accumulation of net energy over each reproductive period.

Individual animals choose the biomass they consume from prey species based on endogenous energy prices for locating, capturing, and handling prey. Individual plants choose to grow biomass depending on the benefits of greater photosynthesis versus the energy price paid for that growth.

The model is calibrated according to the recent Atlantic Rim Natural Gas (ARNG) development in south-central Wyoming, USA. The ARNG region is a typical Western rangeland ecosystem covering approximately 100,000 ha and home to many plant and animal species. The ARNG also provides ecosystem services both to ranchers who graze their cattle on public land managed by the U.S. Bureau of Land Management (BLM), and to elk, mule deer, and antelope hunters. The ARNG development started in 1998 and is scheduled to contain more than 2,000 natural gas wells during its duration (BLM 2017). Even at historically low current natural gas prices, the project is predicted to generate approximately 1.35 trillion ft<sup>3</sup> of natural gas over its lifetime. Annual U.S. consumption of natural gas was about 27 trillion ft<sup>3</sup> in 2017 (BLM 2017). However, the development also represents a substantial disturbance to the rangeland ecosystem.

The ARNG is also typical of a Western rangeland ecosystem because it is under constant pressure from various types of human activities such as ED, transmission lines, roads, housing, etc. (Hanser, et al. 2011, p. 24). While the benefits of such development are fairly well understood and measurable, the ecosystem costs are more nebulous. The approach we apply to the ARNG is a step forward because it can account for the multiple, and possibly complex, relationships between plant and animal species that compete for resources, and it is generalizable to other ecosystems provided sufficient data exists. In addition to ED, our approach could be used to better understand the ecosystem costs and benefits of disturbances such as wildfires, wildlife diseases, urban sprawl, and invasive species, to name a few.

## **2 Methods**

### **2.1 Study Area**

In this section, we provide more details on the ARNG development and how the DAs are introduced into the model. We calibrate our model to the ARNG because the development project represents a large, sudden shock to an area that is a typical Western rangeland ecosystem. Another reason to study the ARNG development is that the U.S. BLM produced an environmental impact statement (EIS) that provides an additional source of information regarding the project and relevant ecosystem.

The ecosystem is a sagebrush-steppe representative of rangeland in the Intermountain West. According to Rowland et al. (2010), the sagebrush-steppe ecosystem covers about 43 million ha in eleven western states and one Canadian province. It is threatened by numerous human activities including ED and livestock grazing that are modeled here, and the authors identified over 350 sagebrush-associated species that are rare or declining.

To keep the model tractable, the food web for ARNG is limited to 12 native species and one non-native species (cattle) as shown in figure 1. There are three trophic levels, with two plants comprising the first trophic level, eight herbivores the second, and three carnivores the third. The arrows in the food web show the direction in which biomass (and thus energy) flows, and the accompanying numbers show the percentage of a predator's diet coming from a prey species. For example, grass makes up 90% of elk's diet. The dashed arrows show two ecosystem services: 1) antelope, mule deer, and elk hunting, and 2) cattle grazing on grass.

### **2.2 Energy Development and Disturbed Areas**

The ARNG development project in Carbon County covers 109,339 ha in south-central

Wyoming that is 64% federal, 31% private, and 5% state owned. It consisted of 627 natural gas wells during 2005-2010, but it is scheduled to contain up to 2,000 wells that will be operated by multiple companies throughout the 30-50 year life of the project. Both the National and the Wyoming Wildlife Federations have initiated a legal challenge to the BLM's ARNG management plan, claiming that the plan does not adequately protect species in the region and adversely impacts ecosystem services such as hunting, fishing, and wildlife viewing (NWF 2007).

Previous research has found that various species avoid natural gas wells, but the magnitude of the impact depends on the intensity of the development, the affected species, and the season. A study of the Powder River Basin in Wyoming and Montana shows that coal-bed natural gas (CBNG) development has a negative impact on sage grouse leks, with the DA ranging somewhere between 0.8 km and 3.2 km from the wells (Walker et al. 2007). Doherty et al. (2008) estimate that sage grouse were 1.3 times less likely to occupy sagebrush habitat in a 4 km<sup>2</sup> area around CBNG wells (equivalent to a circle with radius of 1.13 km). Similarly, studies suggest that elk avoid oil and gas development areas (Hebblewhite 2011). In particular, elk avoided areas within 2 km of major roads and active gas/oil wells during the summer and 1 km during winter (Powell 2003).

To account for the impact of ED, we assume that elk and sage grouse do not forage inside the areas surrounding the natural gas well which are labelled "disturbed areas" (DAs). All other species in the rangeland ecosystem forage and hunt in both the DAs and the area outside the DAs labelled the undisturbed area (UA). Although other species may be affected by ED, such as mule deer (Hebblewhite 2011), we only focus on elk and sage grouse given their importance for ecosystem services. A simple schematic of habitat loss is shown in figure 2.

Figure 3 shows the location of the active natural gas wells in 2010 and the total DA in the ARNG region from 1980 to 2010 assuming a 1 km disturbance radius around each well. Figure 4 shows the total DA for all wells from 1980 to 2010. Although the total DA has generally been growing, it started increasing sharply in 1998. Given the 1 km radius for the DAs, the total affected area is equal to 15,387 ha or 14% of the ARNG area. Interestingly, the maximum allowable DA in the EIS is significantly smaller (BLM 2007), but may have only considered actual cleared habitat and not the avoided area around gas wells. The EIS plan calls for active reclamation that would stabilize disturbed soils and vegetation communities, while allowing for no more than 3,075 ha of total DA (3% of the total ARNG area) at any time.

In establishing habitat loss, we simplify the analysis by assuming that the DA is identical for sage grouse and elk. Walston et al. (2009) did something similar by assuming that indirect impacts of natural gas development in the Pinedale Anticline Project Area in western Wyoming have a 1,000 m “buffer” around wells. Van Dyke et al. (2012) studied the behavioral responses of elk to habitat alteration near natural gas wells in Colorado, and they selected sites that were less than 1,000 m from operating wells. Holloran (2005) found that for sage-grouse in western Wyoming the number of displaying males declined as distances from leks to drilling activity decreased, where leks located less than 1,000 m were the most affected. For tractability, another simplification is that all gas wells new since 1999 are assumed to be introduced simultaneously. Absent this assumption, all the DAs in the model would need to be indexed by their age.

### **2.3 General Equilibrium Ecosystem Model (GEEM)**

In this section, we provide a non-technical overview of GEEM and more thoroughly discuss hunting, grazing and the introduction of DAs around natural gas wells. The basic premise

behind GEEM is that ecosystem properties, and species' populations in particular, are ultimately determined by individual organisms' behaviors; therefore, in ecological models population-updating equations should be derived from individual behaviors. GEEM accomplishes this by assuming species' individuals are energy optimizers, similar to optimal foraging theory, and the results of the optimization problems, including biomass consumptions and energy transfers, are used to construct population-updating equations. Abrams (1999) pointed out that most studies employing optimum foraging ignore population dynamics, and most studies of population dynamics ignore optimum foraging. GEEM is an attempt to bridge the gap. A richer summary of the model is in Section 1 of the Appendix, as well as in Tschirhart (2000, 2002, 2009) and Finnoff and Tschirhart (2008, 2009).

In GEEM, the sun is the source of all energy, and solar energy is converted to biomass by the two groups of primary producers (grass and shrubs). There is a fixed amount of land available where plants compete for light, acquire essential macronutrients (such as nitrogen), grow, and reproduce. The animals in the food web consume grass, shrubs, and other animals. Thus, solar energy enters the system through the plants and indirectly to the animals through herbivore foraging and predator-prey relationships. All the solar energy entering the ecosystem is accounted for as energy leaving the system through respiration or as energy stored in the plants and animals (Tschirhart 2000). Predation risk is incorporated in the prey net energy objective functions because it has been shown to be important in sage-brush ecosystems (e.g., Gude et al. 2006).

Each species  $i$ ,  $i = 1, \dots, 12$ , is comprised of  $n_i$  identical individuals that act as if they maximize the accumulation of net energy ( $R_i$ ) in every period ( $t$ ) by choosing their biomass accumulation via photosynthesis ( $x_i$  for plants), or biomass consumption via grazing and



predation ( $x_i$  also for animals). Plants (animals) choose their  $x_i$  in response to the energy they must expend to grow biomass (locate, capture and handle prey biomass). The energy prices paid by plants are referred to as the shading energy loss (SEL), a measure of the energy that a plant's leaf loses when sunlight is blocked by other leaves. Whenever a plant increases the size of its leaf biomass, it creates shade for its own leaves and the leaves of other plants. Therefore, the space occupied by the leaves of other plants imposes a cost on each individual plant; this explains why the population and biomass accumulation of grass affects shrubs, and vice versa. The SEL energy price can also be interpreted as measuring the scarcity of vertical space. Each animal and plant maximizes net energy by balancing the gains and losses of biomass consumption. Animal gains come from energy embodied in the animal or plant biomass consumed, and the losses come from paying the energy prices, from predation avoidance, and from respiration that includes territory defense, locomotion, feces, and reproduction. Plant gains come from growing photosynthetic biomass and their losses come from respiration, foraging, and the energy lost from shade created by its leaves and the leaves of other plants.

In every time period, a set of equations taken from the optimization problems of individual plants and animals, and from the balance conditions that equate biomass demands of predators and grazers with biomass supplies from prey and plants, are solved to yield the energy prices and optimum biomasses. When the transfers of biomass between species is in balance and when all individuals are maximizing net energy, the ecosystem is said to be in general equilibrium. Similar to perfect competition in microeconomic theory, individual agents (individual plants and animals) cannot influence the final energy prices as they are determined by the market forces of demand for and supply of biomass aggregated over all species' individuals. For example, in a predator-prey relationship, when the predator population is large (small), other

things equal, the general equilibrium energy price individual predators pay for prey is high (low). The reverse holds for prey populations.

Populations are updated each period using the optimum biomass transfers and energy prices determined in the general equilibrium calculations. These biomasses and prices are substituted into individuals' net energy functions,  $R_i$ , which are used to determine the birth rate per capita. The death (or mortality) rate per capita is influenced by how much an individual exposes itself to predation by consuming more biomass. The growth rate per capita is simply the birth rate per capita minus the death rate per capita and is used to update the population of each species at the end of each period.

If the updated populations of all species are unchanging over time then a steady state (SS) is said to obtain. After a perturbation to a SS, the general equilibrium biomass and price values change which drive new population changes. If out of SS, there are forces that tend to move the system to SS. That is, when individuals in a species have high (low or negative) optimum  $R_i$ , they will have high (low) birth rates and populations will increase (decrease) in subsequent periods until a new SS is attained. The speed at which this occurs largely depends on the magnitude of the perturbation and on the responsiveness of plants.

### **2.3.1 Energy Prices and Competition**

The energy prices are particularly important because they can be used as indicators of intra- and inter-specific competition. The individual plant or animal maximizes net energy by choosing biomasses in response to the prices it observes but which it has no control over as an individual. The energy prices are emergent properties determined by the aggregated behaviors of plant and animal populations determined via the general equilibrium calculations. Moreover, the

initial energy prices obtained in calibrating GEEM are dependent on data taken from field and laboratory studies. This approach may be more realistic than traditional Lotka-Volterra competition models that start with populations and species-level parameters and that omit individual behavior (Tschirhart 2009).

With respect to the energy prices, consider an elk for whom the energy price it pays measures the cost of locating and handling grass. If the elk population increases in one period, this energy price increases in next period's general equilibrium calculations because more elk increase the demand for the same supply of grass. Therefore, in the next period each elk will consume less grass which will lower biomass consumption and optimum net energy, which in turn will lower the birth rate in the following period. Hence, a higher energy price to elk leads to a lower elk population consistent with intra-specific exploitative competition. Additionally, an increased elk population will decrease the supply of grass in the next period which will increase the energy price paid by jackrabbits and lower jackrabbit optimum net energy and birthrate. In this way the energy prices also indicate inter-specific competition.

In GEEM, an absence of competition is also possible. For example, if the elk population decreased substantially, then the supply of grass may exceed the elk demand for grass in which case the balance condition that equates biomass demands of elk with biomass supplies from grass becomes an inequality. In the general equilibrium calculations, relaxing the equality yields a zero energy price paid by elk. This follows using Kuhn-Tucker conditions from non-linear programming. In effect, the elk are satiated in grass, their optimum net energies are at a maximum, and birth rates at their highest possible level. In GEEM, we say there is no competition when an energy price for a predator chasing prey or an herbivore consuming forage is zero. An example of no competition is provided in the results section below.

### **2.3.2 GEEM and Plant's Uptake of Nitrogen**

Here we discuss the role of macronutrient restrictions in GEEM. The availability of nitrogen is the primary factor limiting plant growth in many ecosystems (Chapin et al. 1986; Tilman et al. 1996), and in a prairie system the competition between grasses and shrubs is strongest for nitrogen (Köchy and Wilson 2000). In the model, if there are no other resources needed by the plants besides light, any deviation from a SS causes the population of one plant species to continuously grow and displace the other species. This result is consistent with the resource/ratio hypothesis which states that the number of competing plant species can be no greater than the number of limiting resources (Tilman 1985; Miller et al. 2005). GEEM is consistent with the resource/ratio hypothesis (Finnoff and Tschirhart 2009); therefore, because there are two plant species in the ARNG model, nitrogen is introduced as a second resource to allow coexistence of the plant species. In the model there is a restriction that limits how much nitrogen can be absorbed from different layers of soil according to the grass versus shrub root structures and biomass accumulations. This absorption of nitrogen then regulates plant growth and biomass accumulation. See the Section 2.2 of the Appendix for further details regarding how nitrogen is introduced into the model and how it impacts plant growth.

### **2.3.3 GEEM and Cattle Grazing**

Since much of the land in the ARNG area is owned by the BLM, ranchers are required to obtain cattle grazing permits. Cattle consume only grass as shown in figure 1, which puts them in direct competition with the four native grass-eaters. We assume that cattle effectively take the place of bison in the original rangeland food web before bison were hunted to near extinction in

the 1800s. Cattle and bison are similar in size, primarily grass eaters, and are good substitutes for grazing management (Peden et al. 1974). There are differences in their feeding behaviors noted by Plumb and Dodd (1993), but the differences are not pertinent for this GEEM that omits spatial details. Therefore, cattle are calibrated as part of the “natural” GEEM SS. By natural SS, we refer to the SS where the population of all the native species of the ecosystem correspond to the initial densities used in the calibration.

The twelve native ARNG species use a portion of their energy to reproduce in GEEM. Whenever prey or forage is abundant, these species expend less energy and are able to reproduce at a greater rate. Cattle, however, are assumed to use their energy to gain weight rather than reproduce. Ranchers employ a stocker operation where they acquire young adult steers or heifers and graze them for one season before they are sent to market. Since much of this grazing occurs on public land under grazing permits, the density of cattle on the rangeland ecosystem is, at least in part, under the control of the BLM.

Following the literature, a stocking rate for steers or heifers of 0.22 (0.57, 0.90) per ha is considered low (heavy, very heavy) density (Hart and Ashby 1998, Derner and Hart 2007). For the ARNG density, according to the EIS there are 31 relevant BLM grazing allotments in the ARNG and the surrounding area, which allows for a total of 39,695 animal unit months (AUMs). Assuming the number of AUMs inside the ARNG is proportional to its acreage results in 21,135 AUMs. Assuming each animal is allowed to graze for 5 months implies a total of 4,227 individual steers and heifers. Since there are 109,297 ha in the ARNG, the estimated grazing density is  $4,227/109,297 = 0.0387$ . This density is much lower than what the literature considers low density, but many of those studies refer to more fertile land and dense grass. We use our estimation to calibrate the parameters in GEEM (see Section 2.4 of the Appendix for more

information).

### **2.3.4 GEEM and Hunting**

Once GEEM is calibrated to be consistent with the nitrogen restriction and the natural rangeland ecosystem, hunting and ED are introduced. Hunting is modeled as a percentage reduction of the elk, antelope, and mule deer populations at the end of each period. The percentage, or hunting rate, is fixed but the number of harvested animals varies over time. To calibrate GEEM for recent hunting rates, we collected 2014 data from published Wyoming Game and Fish Department (WGFD) harvest reports (WGFD 2014). The hunting areas (or units) intersect with the ARNG region but none falls entirely inside. We calculated the percentage of each unit that falls inside the ARNG region, and then assumed that the same percentage of elk, antelope, and mule deer from the total animals hunted were harvested inside the ARNG region. Detailed elk, antelope, and mule deer hunting data for the ARNG are in Section 2.5 of the Appendix.

## **3 Results**

Given the lack of complete ecological and economic data for the ARNG region, the model is calibrated using parameter values, economic parameters, and functional forms that are sometimes imprecise. Therefore, we quantify the inherent uncertainty in the model's predictions by undertaking a thorough sensitivity analysis whereby the parameter values are varied up and down by a fixed percentage. The results of the sensitivity analysis are shown in Section 4 of the Appendix and will be discussed in more detail below.

Figure 5 shows the simulated time paths for plant biomasses and the populations of all

species for two scenarios: (1) where hunting and ED are introduced and (2) where only hunting is introduced, the base line (BL). Introducing hunting separately is necessary to identify the individual impacts of hunting and ED. The ecosystem starts at the natural SS; then hunting is introduced in period 3 for both scenarios. After a hunting SS is reached (around period 100), ED is introduced.

### **3.1 Responses to Hunting in GEEM**

The impacts on the animal species can be examined by their predator-prey and competitive interactions. There are no detailed field studies of these interactions for the ARNG, but GEEM does yield theoretical predictions for predator-prey interactions in terms of biomass and energy transfers and population changes, as well as energy prices that are a novel quantification of competitive interactions as explained in section 2.4.1.

The introduction of elk, antelope and mule deer hunting in period 3 immediately reduces the populations of these species as shown in figure 5. This disturbance then triggers a myriad of feedback responses within and among species. Consider grass first. With fewer elk after hunting, the population and individual biomass of grass increase briefly (from periods 3 to 5). However, nitrogen limits the total biomass (individual biomass times population) of grass such that a new SS is attained with a larger population of smaller plants as shown by the solid ( $n$ ) and dashed ( $x_i$ ) lines in the first row of figure 5. Meanwhile, with fewer antelope and mule deer, the population and biomass of shrubs also briefly increase in periods 3 to 5. However, after period 5 the SEL price for shrubs increases because of interspecies competition fueled by the higher total biomass of grass. This in turn leads to a smaller population of smaller shrubs. Even though both plant species experience less pressure from herbivores, grass partially displaces shrubs because the

release of pressure on grass is greater. This follows because after hunting, total elk consumption (individual elk consumption times elk population) of grass biomass as a percent of total grass biomass decreases by 1.42%, whereas total deer plus antelope consumption of shrub biomass as a percent of total shrub biomass decrease by only 0.18%. After the initial responses from hunting, grass and shrub populations converge slowly to new baseline SSs.

Hunting in period 3 immediately removes about 600 elk, or about 11% of the population. Consequently, the energy price elk expend for grass decreases by about 15% indicating less intra-specific competition as discussed in section 2.4.1. (Only if the energy price were to fall to zero would intra-specific competition cease to exist; however, other simulations not presented here indicate that to obtain a zero energy price would require removing about 72% of the elk.) With less intra-specific competition, each surviving elk consumes more grass and produces more offspring which offsets some of the hunting losses. Nevertheless, with fewer elk to hunt, the number harvested decreases to about 500 per period and a new SS is reached when harvests equal births. The total decrease in the new SS elk population from the natural SS is about 1,800 (26%). Similar scenarios play out for deer and antelope hunting. Interactions between the forage consumption patterns of ungulates and their interactions with other key species, such as cattle, have been studied by authors such as Hobbs et al. (1996) who found that higher elk density lowers forage intake by cattle through competitive effects.

The impact on other species depends on their plant consumption and whether they have competitive or predator-prey relationships with other animals. For an example of the former relationship, removing elk benefits prairie dogs by lowering inter-specific competition, because removal lowers the energy price prairie dogs pay for grass. Examples of the latter relationship is seen in how the populations of herbivores track their forage plant populations, and predator



populations track their prey populations with slight lags. The effect on coyotes is particularly interesting since they prey upon both mule deer, a hunted shrub-eater, and jackrabbits that consume grass and benefit from elk hunting. After hunting, initially the coyote population increases because the increased jackrabbit abundance offsets the scarcity of mule deer; but when the growth of grass slows, jackrabbit density declines causing the population of coyotes to decline. The coyote and jackrabbit population changes are very small, but they are in the expected directions that reflect the complex inter- and intra-species interactions that are captured within GEEM.

### **3.2 Predicted Ecological Impacts of Energy Development**

ED is introduced into the ecosystem in period 100, after the post-hunting ecosystem attains a new SS. The loss of habitat due to the introduction of natural gas wells causes a decrease in both elk and sage grouse by about 14%, as these are the two species assumed to avoid the DA around natural gas wells (Powell 2003; Hebblewhite 2011). With respect to the natural SS, the total decline of the elk population is 37% when considering both hunting and energy development, and for sage grouse the total decline is 17%.

Plants are affected by ED because the mix of herbivores that consume them now differ between the UA and DA; in particular, elk and sage grouse are displaced from the DA and forage exclusively in the UA. Both elk and sage grouse populations eventually decline because they gather less energy in the relatively more congested UA. For grass, this yields a slightly larger population of smaller grass individuals in the UA. Other grass-eaters choose to forage relatively more heavily inside the DA after elk abandon it owing to less inter-specific competition. This result can be seen in the general equilibrium calculations pre and post ED. At the same time,

shrub-eaters forage relatively less heavily in the DA after ED because, with elk absent there, the shrub biomass decreases relative to the grass biomass. In Section 1 of the Appendix, we show that to attain an ecosystem general equilibrium in each period, an individual herbivores' marginal energy gains inside the DA and the UA must be equal. This follows because if an herbivore enjoyed greater energy gain in say the DA, it would switch to more grazing there which would increase the DA's energy price and lower the UA's energy price. This behavior would continue until the energy prices are equal and the incentive to switch vanishes. In effect, competition is equated between the DA and UA.

### 3.3 Impacts on Ecosystem Services and Biodiversity

Here we calculate the impact of ED on biodiversity and the value of ecosystem services (i.e., cattle ranching, hunting, and the existence value of sage grouse) in the ARNG region. Although sage grouse are only one of many species in the ARNG ecosystem, they are a high-profile species that receives significant media attention and have been repeatedly considered for listing under the U.S. Endangered Species Act (FWS 2015a). The total discounted sum of the values of the ecosystem services (TDVES) is defined as

$$\text{TDVES} = \lim_{T \rightarrow \infty} \sum_{t=t_{\text{ED}}}^T \frac{\pi_{\text{Ca}}^t + \pi_{\text{Hu}}^t + \pi_{\text{Sg}}^t}{(1+r)^{t-t_{\text{ED}}}} \quad (1)$$

where  $t_{\text{ED}}$  is the period when ED is introduced;  $\pi_{\text{Ca}}^t$  is the profit from cattle ranching;  $\pi_{\text{Hu}}^t$  is the net benefit from hunting elk, mule deer, and antelope;  $\pi_{\text{Sg}}^t$  is the existence value of sage grouse, and  $r$  is the discount rate (set at 3% in the simulations). Here we briefly describe how cattle profits, hunting net benefits, and the existence value of sage grouse are calculated with further details provided in Sections 2 and 3 of the Appendix.

Ranchers operate stocker programs whereby younger, smaller cows are purchased to be

grazed in the ARNG during the growing season and then sold for a profit in the fall. Data for cattle purchase price, purchase weight, and final price are from the stocker budget analysis (USDA online). Total cattle profits ( $\pi_{Ca}^t$ ) are given by the per animal gross revenue minus purchase price times the number of cattle in the ARNG. Gross revenues depend on the final weight of the animal (i.e., the purchase weight plus the weight gain over the growing season). The weight gain in turn depends on the abundance of grass, which in turn depends on the interspecies competition.

The hunting net benefits ( $\pi_{Hu}$ ) follow previous research (Hussain and Tschirhart 2010, 2013) and are given by:

$$\pi_{Hu} = \sum_{i \in SH} [(m_i + V(n_i))h_i + (\tau(h_i) - h_i)V(n_i)] - \sum_{i \in SH} C(\tau(h_i)). \quad (2)$$

In (2),  $h_i$  is the number of individual animals hunted from species  $i$ ,  $i \in SH \equiv \{\text{elk, mule deer and antelope}\}$ ,  $m_i$  is the meat value of an individual animal,  $\tau(h_i)$  is the number of hunters and is increasing in  $h_i$ ,  $V(n_i)$  is a hunter's willingness to pay for a hunting experience, and  $C(\cdot)$  is the total cost of hunting that is increasing in the number of hunters. The number of hunters,  $\tau(h_i)$ , and the harvest,  $h_i$ , are not equal because not all hunters are successful (i.e., to kill  $h_i$  animals requires issuing more than  $h_i$  hunting licenses). Table 1 provides data and the method used to calculate hunter success rates. The willingness to pay for a hunting experience,  $V(n_i)$ , is increasing in the population of the hunted species to capture the idea that hunters derive benefits from a hunting experience beyond those associated with a kill. The formulation in (2) shows that meat value is enjoyed only by a successful hunter, but the benefit of a hunting experience is enjoyed by all hunters.

Estimations of the functions  $V(\cdot)$  and  $C(\cdot)$  follow the previous research cited above. For example, the trip value function for elk is given by the concave function:

$$V(n_i) = \alpha n_i^\beta / (1 + n_i), \quad (3)$$

where the  $\alpha$  and  $\beta$  parameters were estimated using data from U.S. Fish and Wildlife Service (2003) on Wyoming residential and nonresidential elk hunters' consumers surplus from a hunting trip, and willingness to pay for viewing elk and wildlife from Shafer et al. (1993) and Rosenberger and Loomis (2001).

The discounted total existence value for sage grouse is calculated using benefit transfer methods and the meta-analysis of Richardson and Loomis (2009). The benefit transfer function specifies no change in the population of the bird species, non-use value only, a dichotomous-choice survey format with a one-time payment, and benefits per household that decrease with distance from the relevant habitat (Loomis 2000, p. 318). Since the baseline household existence value for endangered or threatened species are elicited from non-market household surveys that contain uncertainty due to information, preference, and strategic bias of the households surveyed, we perform a sensitivity analysis to investigate how sensitive the policy prescriptions are to the estimated sage grouse value. For further details regarding the benefit transfer calculations or the sensitivity analysis, see Sections 3 and 4 of the Appendix.

Biodiversity is measured using the divergence from natural diversity (DND) index. The DND index measures the deviation of the population of various species ( $n_i$ ) from the population at the natural SS ( $n_i^{SS}$ ):

$$DND = - \sum_{i \in SDND} \left( \frac{n_i - n_i^{SS}}{n_i^{SS}} \right)^2 \quad (4)$$

where SDND is the set of species used to calculate the DND index. The index considers all the animals except cattle, which were introduced by humans into the rangeland ecosystem. If the population of all the animal species remains at the natural SS, then the DND index is equal to zero; if the population of all the animal species is equal to zero, then the DND index is equal to

-10. Notice that the index falls even when the population of a species is larger than its natural SS population. Also the DND is the sum of quadratic terms so that large deviations from the baseline populations have an increasingly adverse effect on biodiversity. There are many alternative biodiversity measures that could be employed here, each having its own advantages and disadvantages. See Eichner and Tschirhart (2007) and Hussain and Tschirhart (2013) for a discussion of when using the DND is appropriate.

Figure 6 shows the annual value of the ecosystem services and the DND index over the same 200-year period as used in figure 5. The introduction of ED creates winners and losers in terms of ecosystem services. Total hunting net benefits decrease after ED, although this change is comprised of a decrease in elk hunting net benefits that more than offsets small increases in deer and antelope hunting net benefits. The reason for the larger decrease in elk hunting net benefits follows from the decline in the elk population because they avoid the DA. The existence value of sage grouse declines after the introduction of ED, again owing to their avoidance of the DA; but cattle profits increase due to the reduced competition for grass owing to lower elk population and the cattle willingness to graze in the DA. Cattle gain over 24 additional kilograms of weight per animal with respect to the natural SS and 9 additional kilograms with respect to the post-hunting SS. Monroe et al. (2017) found that cattle grazing can have both positive and negative effects on sage grouse populations, while our results show improved cattle grazing and a loss in sage grouse populations with ED. However, our results are not directly comparable to these authors who focused on cattle and sage grouse only, and who included great detail on the timing and level of cattle grazing which is not done here. Summing the annual gains and losses in ecosystem services yields a total discounted loss in the value of ecosystem services over time due to ED of approximately \$38.52 million with respect to the baseline scenario at the post-

hunting SS. This loss is accompanied by a sharp loss in biodiversity.

### **3.4 Optimal Mitigation of Ecosystem Externalities**

In this section, we discuss possible mitigation strategies for the U.S. BLM and WGFD to counteract some of the adverse impacts associated with ED in the ARNG. We begin by noting that the spatial impact of the gas wells and other anthropogenic features are not fully incorporated into the model (Leu and Knick 2008). For example, while we account for overlap in the DAs (see figure 3), the simulations do not consider other spatial aspects such as the impact of having eight separated clusters with different degrees of energy (see figure 4). In future research, we intend to consider a spatial model that respects well spacing, impacts on migration patterns of species, effects of roads, and possible optimization of well location to maximize the ecosystem service benefits.

Can government agencies coordinate to vary cattle stocking and hunting rates in the ARNG region to increase the benefits that society derives from the rangeland ecosystem? The question is complicated because of all the intra- and inter-species competition for resources. In terms of grazing, adding more cattle to the ecosystem may increase rancher profits because there are more cattle to sell, but it also adds more competition for grass so cattle gain less weight that reduces profit per head. Moreover, if the BLM reduces the cattle density, it may decrease cattle profits, but there will be more grass available for other grass-eating species, such as elk. In terms of WGFD management, changing the number of hunting licenses impacts rancher profits. Issuing fewer elk (antelope and deer) licenses and increasing the herds imposes a negative (positive) ecosystem externality on rancher profits. This follows because more elk makes grass less abundant for cattle, and more antelope and mule deer makes grass more abundant for cattle

because it decreases shrub density, which in turn decreases the grasses' loss of energy from shading and increase its biomass.

In a first attempt to address the coordination question, we perform a grid search to find the optimal combination of cattle density and elk hunting rates with GEEM, assuming, for simplicity and tractability, that the antelope and mule deer hunting rates remain fixed. The elk hunting rate and cattle density are selected in each simulation's first period, remain constant at those levels, and then the simulation runs until the ecosystem reaches a SS. The objective is to maximize the TDVES in equation (1) and reach the social optimum.

We consider different optimization problems based on two constraints: (1) a grid search subject to the restriction that no animal species can fall below 50% of the original natural SS, and (2) a grid search subject to the restriction that the DND index cannot fall below -0.5. The DND index value at the SS that arises from the optimum of constraint 1 when there is no ED is approximately -0.5. Therefore, -0.5 is used for the second constraint to show that the DND index allows more flexibility than the first constraint. If either of the restrictions binds in constraint 1 or 2, the simulation stops and the value of ecosystem services is set to zero for the rest of the planning horizon. This is important because otherwise it may be optimal to drive the populations of species to zero which violates the U.S. Endangered Species Act and international treaties such as the 1940 Convention on Nature Protection and Wildlife Preservation in the Western Hemisphere which states that the Governments of the American Republics commit "to protect and preserve in their natural habitat representatives of all species and genera of their native flora and fauna".

Figure 7 depicts the welfare surfaces in the presence of ED. The goal is to identify management strategies that WGFD and BLM officials could make to ameliorate the negative

impacts of ED and maximize TDVES. The height of the surface represents the TDVES for different combinations of elk hunting rates and cattle grazing density ranging from 0.0 to 0.1. The welfare surfaces (i.e., TDVES plotted against hunting rates and cattle density) assume that the ecosystem is initially at the SS that results from choosing the social optimum without ED.

The unconstrained optimization (not shown in figure 7) produces a cattle density of 0.0362 and elk hunting rates from of 0.6097 to reach a maximum TDVES of \$509.46 million. However, a hunting rate of that magnitude is so high that it yields an immediate sharp drop in the elk population, and has the potential of irreversibly disrupting the ecosystem and driving the populations of shrubs and elk to zero. Also, with sufficient nitrogen another extreme scenario arises where grass completely crowds out shrubs and the shrub-eaters (i.e., sage grouse, antelope and deer) also would go extinct.

### **3.4.1 Constraint 1: 50% Population Restriction**

First, we restrict the grid search so that the population of each animal species is always above 50% of the natural SS. Whenever the population of a species is below the restricted level, the simulation stops and the TDVES is set to zero for the remainder of the planning horizon for that combination of cattle grazing density and elk hunting. The optimal management strategy is, therefore, to find a balanced combination of elk hunting and cattle density that can be maintained before the restriction binds.

If WGFD and BLM officials coordinate and respond immediately after the introduction of ED, they can mitigate the negative impacts of ED on the ecosystem. As shown in the first panel of figure 7, the socially optimal combination after ED is an elk hunting rate of 0.2271 and a cattle stocking density of 0.0360 per ha, which yields a TDVES of \$362.33 million (i.e.,



constant annual flow of \$10.55 million). The optimum is achieved by considering that the ecosystem is initially at the post-hunting SS and that the social planner can modify the cattle density and elk hunting at the same time that ED is introduced to the ecosystem. To calculate the loss of ecosystem services from ED, we contrast the optimized TDVES where ED is never introduced to the ecosystem with the TDVES from the top of figure 7. If ED is never introduced, then society would benefit from the ecosystem services with a TDVES of \$423.68 million which also is obtained with a grid search but not shown in figure 7. Therefore, the discounted loss of ecosystem services value from ED is \$61.35 million (\$423.68 minus \$362.32) with coordination.

As an aside, to keep the social planning problem tractable we restrict policy to only consider time-invariant hunting and cattle stocking rates. This lack of flexibility imposed on the WGFD and BLM may result in second-best outcomes. We hypothesize that the first-best strategy might specify a lower cattle density in the short run while the elk population is relatively high, and then slowly increase cattle density, once hunting has reduced the elk population and the competition for grass. We hope to consider time-varying strategies for future research.

### **3.4.2 Constraint 2: Biodiversity Restriction**

The second constraint imposes the restriction that the DND index cannot be lower than -0.5, which is the value of the DND index at the optimum from constraint 1 before ED. Unlike constraint 1, this restriction allows a degree of substitution between the populations of the species because a change in one species' population can compensate for a change in the population of another species. Whenever the DND index is below -0.5, the simulation stops and the TDVES is set to zero for the remainder of the planning horizon for that combination of cattle density and elk hunting.

As shown in the bottom of figure 7, the optimal combination after ED is an elk hunting rate of 0.4488 and a cattle stocking density of 0.0362, which amounts to a TDVES of \$408.51 million (i.e., constant annual flow of \$11.88 million). If ED does not occur, the TDVES at the hunting SS is \$497.91 million. Therefore, the discounted loss of ecosystem services value from ED is \$89.4 million with coordination (497.91 minus 408.51). The superiority of the biodiversity constraint can also be seen by comparing the optimized TDVES with the DND index (\$408.51 million) to the optimized TDVES with the 50% population constraint (\$362.32 million). However, the main take-home message is that with either the 50% constraint or the biodiversity constraint, the negative effects of ED on the ecosystem can be mitigated by coordination between management agencies and selection of appropriate levels of other human disturbances.

### 3.5 Coordination Failure

In this section, we explore what would happen if BLM and WGFD officials, instead of maximizing societal welfare (as defined by the TDVES) after the introduction of ED, maximize the net benefits of the actors that they represent. If the BLM represents the interests of ranchers and the WGFD represents the interests of hunters, then a **coordination failure** could arise since cattle ranching and elk hunting impose ecosystem externalities on one another. The aforementioned ecosystem externalities are present even before the introduction of ED, but are likely to be exacerbated after the introduction of such a large disturbance to the ecosystem.

Table 2 shows the outcomes of a non-cooperative game between the BLM and the WGFD. The rows represent the possible elk hunting rates that the WGFD indirectly chooses by issuing hunting permits. The columns represent the possible cattle densities that are implied by the public grazing permits issued by the BLM. Inside each cell there are four values, all measured in

millions of dollars: the top left indicates the net benefits from hunting for elk, antelope, and mule deer if the antelope and mule deer hunting rates remain fixed according to the 2014 data (WGFD 2014); the top right indicates the profits of cattle ranching; the lower left is the existence value of sage grouse; and the lower right is the TDVES. The bold values are the best response of the WGFD to each action of the BLM and vice versa. The red shaded cells are Nash equilibria (NE), whereby the strategies of both the BLM and WGFD are best responses. Blank cells represent combinations of elk hunting rates and cattle densities that violate constraint 1.

As a point of clarification, consider the possibility where the WGFD issues enough elk hunting permits to allow a 12% annual reduction in the elk population and the BLM issues enough grazing permits to allow a seasonal cattle density of 0.04 per ha. In this case, the benefits would be: \$51.88 million for hunting net benefits, \$48.24 million for cattle ranching profits, \$219.06 for the existence value of sage grouse, and \$319.17 million for TDVES.

Some important remarks on table 2: (1) when cattle density is between 0.005 and 0.035, issuing more hunting licenses does not increase the ranching profits because the density is so low that cattle are already satiated and the increase in grass has no impact on the accumulated energy of cattle; (2) when the elk hunting rate is between 0 and 0.06, increasing the cattle density puts extra foraging pressure on grass which leads to a decrease of grass biomass (negatively affecting grass-eaters) and an increase of shrub biomass (positively affecting shrub-eaters). This increases the net benefits of antelope and mule deer hunting but decreases the benefits of elk hunting. The former impact is greater than the latter impact and the total net benefits increase with cattle density; (3) when the elk hunting rate is between 0.09 and 0.232, increasing the cattle density has negative effect on the net benefits of hunting because the decrease from elk-hunting benefits shadows the increase of benefits from antelope and mule deer.

The main message from table 2 is that public policy without coordination between the BLM and WGFD could lead to suboptimal outcomes. Notice there are multiple NE where each agency is choosing the best strategy given the choice of the other agency. In general, the most favorable NE for the WGFD are also the least favorable for the BLM and vice versa. Only one of the NE is the social optimum but there is no guarantee it will be selected if the two agencies do not coordinate.

### **3.6 Discussion**

Previous studies have tended to focus on how ED affects a narrow set of species, such as caribou and elk, without analyzing the feedbacks from the rest of the species in the food web (Sorensen et al. 2008; Wasser et al. 2011; Dzialak et al. 2011). The advantage of using GEEM is that a multi-species food web can be used to capture the feedbacks. To illustrate the advantage, we ran two different simulations using GEEM: the first uses a simpler version of the food web that includes only grass, cattle, and elk; the second version uses the entire food web. The two versions are labelled the naïve and complex scenarios. In both scenarios, the ecosystem is initialized at the natural SS. Elk hunting is then introduced in period 3 according to the 2014 WGFD harvest reports, and ED is introduced in period 100. Figure 8 shows the results.

In figure 8, the impact of elk hunting and ED on grass is overstated in the naïve scenario because the competition for sunlight between grass and shrubs is not modeled and other grass-eaters are not considered. In the naïve scenario, the forage pressure on grass is underestimated, because the model only accounts for cattle and elk and omits other grass eaters. In addition, the complex scenario includes other predator-prey relationships and foraging such as jackrabbits, which forage on both grass and shrubs. Another advantage of the complex model is that it more

accurately reflects the gradual recovery for elk after hunting is introduced. In the naïve scenario, the population of elk recovers in less than 10 periods before reaching the hunting SS, while elk recovery takes over 80 periods in the complex scenario. Finally, under a naïve scenario cattle ranching profits increase when elk are pushed out of the DA, and the increase is more than enough to compensate for the net benefit losses from elk hunting. This leads to the conclusion that ED has little impact on TDVES, while GEEM with the complete food web and more complex interactions shows a significant reduction in TDVES.

We highlight two main findings of our research. First, although ED provides substantial economic benefits to society, it has the potential to create winners and losers in terms of species, and generally harms biodiversity. On the one hand, species such as sage grouse and elk that avoid the area around the natural gas wells will lose habitat and experience population declines. On the other hand, the populations of herbivores that are willing to graze in the DA may increase due to reduced forage competition, as will the populations of predators of these herbivores.

Second, there are important ecosystem externalities that hunting and cattle ranching, and the agencies that represent them, inflict on each other. Given this context, it is necessary to promote coordination between these economic agents to reach the social optimum, particularly after the introduction of ED. For example, a higher elk hunting rate will cause cattle to experience less competition for grass, gain more weight, and increases the overall value of ecosystem services. In terms of biodiversity, decreasing the density of cattle will cause grass to become more abundant, ameliorating the reduction in the elk population caused by ED. This implies that the energy companies, WGFD officials, and BLM officials can coordinate to generate a management strategy that balances energy production, changes in ecosystem services, and potential losses in biodiversity. Crawford (2004) shows the linkage of habitat and population

dynamics of sage grouse by considering factors such as the effects of fire-ecology, livestock grazing, and herbicide-based control of big sagebrush; they conclude that solving management issues associated with the decline of sage grouse will require cooperation among wildlife biology, range science, and other professional abilities. Our research supports this conclusion.

We close by mentioning a couple of caveats for our work. First, there is substantial uncertainty in some of the calibrated parameters and modeling assumptions. We vary parameters related to (1) the subset of species affected by ED, (2) the availability and plant uptake of nitrogen in the soil layers, and (3) the existence value for sage grouse per household. In all but one case, the impact of introducing ED is negative and measured in terms of the total discounted value of ecosystem services (TDVES), the loss varies between \$5.62 and \$76.96 million in the baseline, non-optimized scenario. As a reference point, recall that the negative impact estimated in our baseline scenario is \$38.52 million. See Section 4 of the Appendix for further details.

Second, plants and animals may be impacted by ED for various reasons. For example, studies show that sage grouse populations may decline if established leks are abandoned due to increased human or predator activity from the introduction of roads and power lines (Ellis 1984; Lyon 2003; Holloran 2005; Kaiser 2006; Doherty 2008). For these and other reasons, our estimates on species populations and the associated loss in ecosystem services might be considered a lower bound on the true effects. In future research, we intend to further explore the other ways in which ED may impact the ecosystem.

## References

- Abrams, P.A. 1999. The adaptive dynamics of consumer choice. *American Naturalist* 153, 83-97.
- Bagheera. 2016. *Black-footed ferret an endangered species*. Retrieved 2016-06-23, from <http://www.bagheera.com/inthewild/vananimferret.htm>
- Chapin III, F. Stuart, Vitousek, Peter M., Van Cleve, Keith. 1986. The Nature of Nutrient Limitation in Plant Communities. *The American Naturalist*, 127(1), 48-58.
- Convention on Nature Protection Wild Life Preservation in the Western Hemisphere. 1940. *Treaty TRE-000085*. Retrieved 2017-05-19, from <http://www2.ecolex.org/server2neu.php/libcat/docs/TRE/Full/En/TRE-000085.txt>
- Crawford, J. A., Olson, R. A., West, N. E., Mosley, J. C., Schroeder, M. A., Whitson, T. D., and Boyd, C. S. 2004. Ecology and management of sage-grouse and sage-grouse habitat. *Journal of Range Management*, 57(1), 2-19.
- Crocker, T. D., and Tschirhart, J. 1992. Ecosystems, externalities, and economies. *Environmental and Resource Economics*, 2(6), 551-567.
- Daily, G. C. 1997. Introduction: What are ecosystem services? In G. Daily (Ed.), *Nature's services: Societal dependence on natural ecosystems*. (pp. 1-10). Washington, D.C.: Island 697 Press.
- Davies, K. W., Bates, J. D., and Nafus, A. M. 2010. Vegetation characteristics of mountain and Wyoming big sagebrush plant communities in the Northern Great Basin. *Rangeland Ecology Management*, 63, 461-466.
- Derner, J. D., and Hart, R. H. 2007. Grazing-induced modifications to peak standing crop in northern mixed-grass prairie. *Rangeland Ecology and Management*, 60(3), 270-276.
- Doherty, K. E., Naugle, D. E., Walker, B. L., and Graham, J. M. 2008. Greater sage-grouse winter habitat selection and ED. *Journal of Wildlife Management*, 72, 2644-2654.
- Dzialak, M. R., Webb, S. L., Harju, S. M., Winstead, J. B., Wondzell, J. J., Mudd, J. P., and Hayden-Wing, L. D. 2011. The spatial pattern of demographic performance as a component of sustainable landscape management and planning. *Landscape Ecology*, 26, 775-790.
- Eichner, T., and Tschirhart, J. 2007. Efficient ecosystem services and naturalness in an ecological/economic model. *Environmental and Resource Economics*, 37, 733-755.
- Ellis, K. L. 1984. Behavior of lekking sage grouse in response to a perched golden eagle. 715 *Western Birds*, 15, 37-38.
- Finnoff, D., and Tschirhart, J. 2008. Linking dynamic economic and ecological general equilibrium models. *Resource and Energy Economics*, 30, 91-114.

- Finnoff, D., and Tschirhart, J. 2009. Plant competition and exclusion with optimizing individuals. *Journal of Theoretical Biology*, 261, 227–237.
- Finnoff, D., and Tschirhart, J. 2011. Inserting ecological detail into economic analysis: Agricultural nutrient loading of an estuary fishery. *Sustainability*, 3, 1688–1722.
- Gude, J. A., Garrott, R. A., Borkowski, J. J., and King, F. 2006. Prey risk allocation in a grazing ecosystem. *Ecological Applications*, 16(1), 285–298.
- Hanser, S. E., M. Leu, S. T. Knick, and C. L. Aldridge (editors). 2011. Sagebrush ecosystem conservation and management: ecoregional assessment tools and models for the Wyoming Basins. Allen Press, Lawrence, KS.
- Hart, R. H., and Ashby, M. M. 1998. Grazing intensities, vegetation, and heifer gains: 55 years on shortgrass. *Journal of Range Management*, 51, 392–398.
- Hebblewhite, M. 2011. Effects of ED on ungulates. In D. E. Naugle (Ed.), *ED and wildlife conservation in western North America* (pp. 71–94). Washington, D.C.: Island Press.
- Hobbs, N. T., Baker, D. L., Bear, G. D., and Bowden, D. C. 1996. Ungulate grazing in sagebrush grassland: Mechanisms of resource competition. *Ecological Applications*, 6(1), 200–217.
- Holloran, M. J. 2005. *Greater sage-grouse (Centrocercus urophasianus) population response to natural gas field development in western Wyoming* (Dissertation Thesis). University of Wyoming.
- Hussain, A. M. T. and Tschirhart, J. 2010. Optimal harvest licensing when harvest success is uncertain. *American Journal of Agricultural Economics*, 92(1), 125-140.
- Hussain, A. M. T., and Tschirhart, J. 2013. Economic/ecological tradeoffs among ecosystem services and biodiversity conservation. *Ecological Economics*, 93, 116–127.
- Jachowski, David S. and Lockhart, J. Michael 2009. Reintroducing the black footed ferret *Mustela nigripes* to the Great Plains of North America. *Small Carnivore Conservation* 41, 58-64.
- Kaiser, R. C. 2006. *Recruitment by greater sage-grouse in association with natural gas development in western Wyoming* (Dissertation Thesis). University of Wyoming.
- Kirol, C. P., Beck, J. L., Uzurbazar, S. V., Holloran, M. J., and Miller, S. N. 2015. Identifying Greater Sage-Grouse source and sink habitats for conservation planning in an ED landscape. *Ecological Applications*, 25(4), 968–990.
- Köchy, M. and Wilson, S.D. 2000. Competitive effects of shrubs and grasses in prairie. *Oikos* 91, 385-395.
- Leu, M., Hanser, S. E., and Knick, S. T. 2008. The human footprint in the west: A large-scale analysis of anthropogenic impacts. *Ecological Applications*, 18(5), 1119–1139.



Lyon, A. G., and Anderson, S. H. 2003. Potential gas development impacts on sage-grouse nest initiation and movement. *Wildlife Society Bulletin*, 31, 486–491.

Miller, T. E., Burns, J. H., Munguia, P., Walters, E. L., Kneitel, J. M., Richards, P. M., ... Buckley, H. L. 2005. A critical review of twenty years' use of the resource ratio theory. *The American Naturalist*, 165(4), 439–448. Retrieved from <http://www.journals.uchicago.edu/doi/10.1086/428681>

Minnick, T. J., and Alward, R. D. 2015. Plant-soil feedbacks and the partial recovery of soil spatial patterns on abandoned well pads in a sagebrush shrubland. *Ecological Applications*, 25(1), 3–10.

Monroe, A. P., Aldridge, C. L., Assal, T. J., Veblen, K. E., Pyke, D. A., and Casazza, M. L. 2017. Patterns in Greater Sage-grouse population dynamics correspond with public grazing records at broad scales. *Ecological Applications*, 27(4), 1096–1107. Retrieved from <http://doi.wiley.com/10.1002/eap.1512> doi: 10.1002/eap.1512

National Wildlife Federation. 2014. *Global Warming and Sagebrush Steppe Habitat*. Retrieved 2014-10-16, from <http://www.nwf.org/Wildlife/Threats-to-Wildlife/Global-Warming/Effects-on-Wildlife-and-Habitat/Sagebrush-Steppe.aspx>

National Wildlife Federation, and Wyoming Wildlife Federation. 2007. *Appeal presented to the Interior Board of Land appeals by the National Wildlife Federation and the Wyoming Wildlife Federation who appeal to the Bureau of Land Management regarding the Atlantic Rim Natural Gas Field Development Project record of decision*. Retrieved 2015-03-26, from [http://www.ourpubliclands.org/files/upload/Final SOR 7-20-07 NWF \\_WWF.pdf](http://www.ourpubliclands.org/files/upload/Final%20SOR%207-20-07%20NWF%20_WWF.pdf)

Northrup, J. M., and Wittemyer, G. 2013. Characterising the impacts of emerging ED on wildlife, with an eye towards mitigation. *Ecology Letters*, 16, 112–125.

Onsager, J. A. 1984. A method for estimating economic injury levels for control of rangeland grasshoppers with malathion and carbaryl. *Journal of Range Management*, 37(3), 200–203.

Peden, D. G., Van Dyne, G. M., Rice, R. W., and Hansen, R. M. 1974. The trophic ecology of Bison bison L. on shortgrass plains. *Journal of Applied Ecology*, 11(2), 489–497.

Plumb, G. E., and Dodd, J. L. 1993. Foraging ecology of bison and cattle on a mixed prairie: Implications for natural area management. *Ecological Applications*, 3(4), 631–643.

Powell, J. 2003. *Distribution, habitat use patterns, and elk response to human disturbance in the Jack Marrow Hills, Wyoming* (M.S. thesis). University of Wyoming.

Rowland, Mary M., Suring, Lowell H. and Wisdom, Michael J. 2010. Assessment of habitat threats to shrublands in the Great Basin: A case study. General Technical Report - Pacific Northwest Research Station, USDA Forest Service 2010 No.PNW-GTR-802(2) pp.673-685 ref.33.

Sage Grouse Initiative. 2016. *Conserve our Western roots*. Retrieved 2016-06-17, from

<http://www.sagegrouseinitiative.com/roots/>

Shafer, E.L., Carline, R., Guldin, R.W. and Cordell, H.K. 1993. Wildlife management, illegal hunting and conflicts: a bioeconomic approach. *Environmental Management*, 17(5), 669-82.

Smika, D. E., Haas, H. J., Rogler, G. A., and Lorenz, R. J. 1961. Chemical properties and moisture extraction in rangeland soils as influenced by nitrogen fertilization. *Journal of Range Management*, 14(4), 213-216.

Solow, A. R., and Beet, A. R. 1998. On lumping species in food webs. *Ecology*, 79(6), 2013-2018.

Sorensen, T. C., Mcloughlin, P. D., Hervieux, D., Dzus, E., Nolan, J., Wynes, B., and Boutin, S. 2008. Determining sustainable levels of cumulative effects for boreal caribou. *The Journal of Wildlife Management*, 72(4), 900-905.

Tilman, D. 1985. The resource-ratio hypothesis of plant succession. *The American Naturalist*, 125(6), 827-852.

Tschirhart, J. 2000. General equilibrium of an ecosystem. *Journal of Theoretical Biology*, 203(1), 13-32.

Tschirhart, J. 2002. Resource competition among plants: From maximizing individuals to community structure. *Ecological Modelling*, 148, 191-212.

Tschirhart, J. 2009. Integrated ecological-economic models. *Annual Review of Resource Economics*, 1, 381-407. Retrieved from <http://www.annualreviews.org/doi/10.1146/annurev.resource.050708.144113>

U.S. Bureau of Land Management. 2007. *Record of decision: Environmental impact statement for the Atlantic Rim Natural Gas Field Development Project, Carbon County, WY.* (Tech. Rep.).

U.S. Department of Agriculture Natural Resource Conservation Service. Technical resources: Economics. Retrieved 2018-08-20, from <http://www.economics.nrcs.usda.gov/technical/tools/index.html>

U.S. Energy Information Administration. Natural gas. Retrieved 2018-08-20, from <https://www.eia.gov/dnav/ng/hist/n9140us2A.htm>

U.S. Fish and Wildlife Service. 2003 Net Economic Values for Wildlife-Related Recreation in 2001. Addendum to the 2001 National Survey of Fishing, Hunting and Wildlife-Associated Recreation. Report 2001-3.

U.S. Fish and Wildlife Service. 2015a. *2015 endangered species act finding*. Retrieved 2016-02-16, from <http://www.fws.gov/greatersagegrouse/findings.php>

U.S. Fish and Wildlife Service. 2015b. *Greater sage-grouse*. Retrieved 2016-03-23, from <http://www.fws.gov/greatersagegrouse/>

Van Dyke, F., Fox, A., Harju, S. M., Dzialak, M. R., Hayden-Wing, L. D., and Winstead, J. B. (2012). Response of elk to habitat modification near natural gas development. *Environmental Management*, 50, 942–955.

Walker, B. L., Naugle, D. E., and Doherty, K. E. (2007). Greater sage-grouse population response to ED and habitat loss. *The Journal of Wildlife Management*, 71(8), 2644–2654.

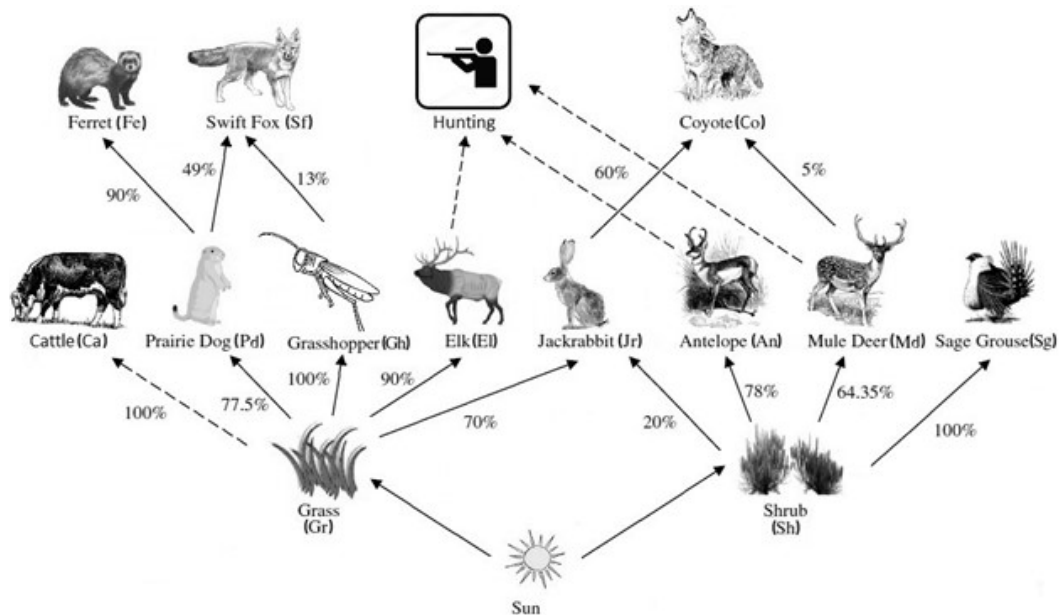
Walston, L. J., Cantwell, B. L., and Krummel, J. R. 2009. Quantifying spatiotemporal changes in a sagebrush ecosystem in relation to ED. *Ecography*, 32, 943–952.

Wasser, S. K., Keim, J. L., Taper, M. L., and Lele, S. R. 2011. The influences of wolf predation, habitat loss, and human activity on caribou and moose in the Alberta oil sands. *Frontiers in Ecology and the Environment*, 9(10), 546–551.

Weaver, J. E. 1926. Root habits of native plants and how they indicate crop behavior. In J. E. Weaver (Ed.), *Root development of field crops* (1st ed., chap. IV). New York, NY: McGraw-Hill Book Company. Retrieved from <http://soilandhealth.org/wp-content/uploads/01aglibrary/010139fieldcroproots/010139toc.html>

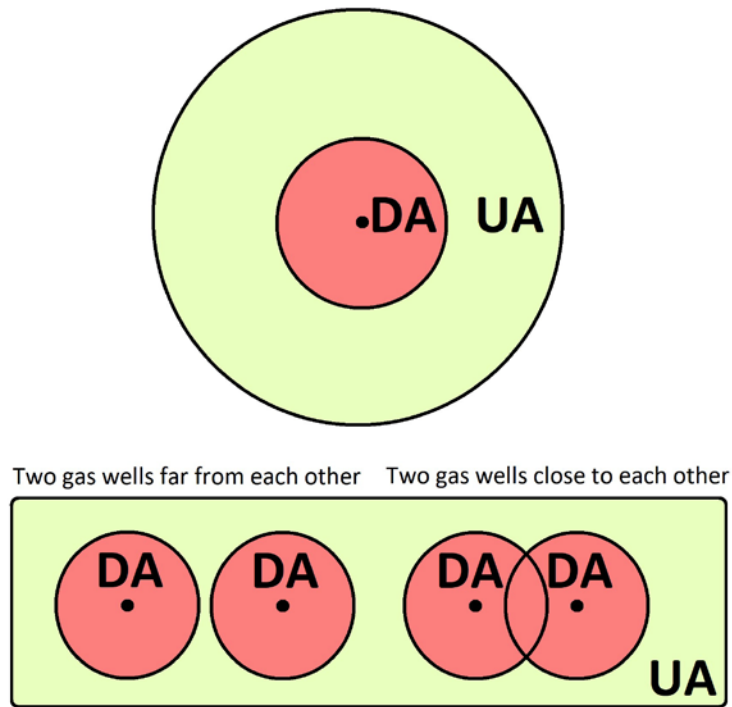
Wyoming Game and Fish Department. 2014. *Annual reports of big and trophy game harvest and annual reports of small and upland game harvest*. Retrieved 2014-10-16, from <https://wgfd.wyo.gov/Hunting/Harvest-Reports/>

**Figure 1. Sample Rangeland Ecosystem and Ecosystem Services**



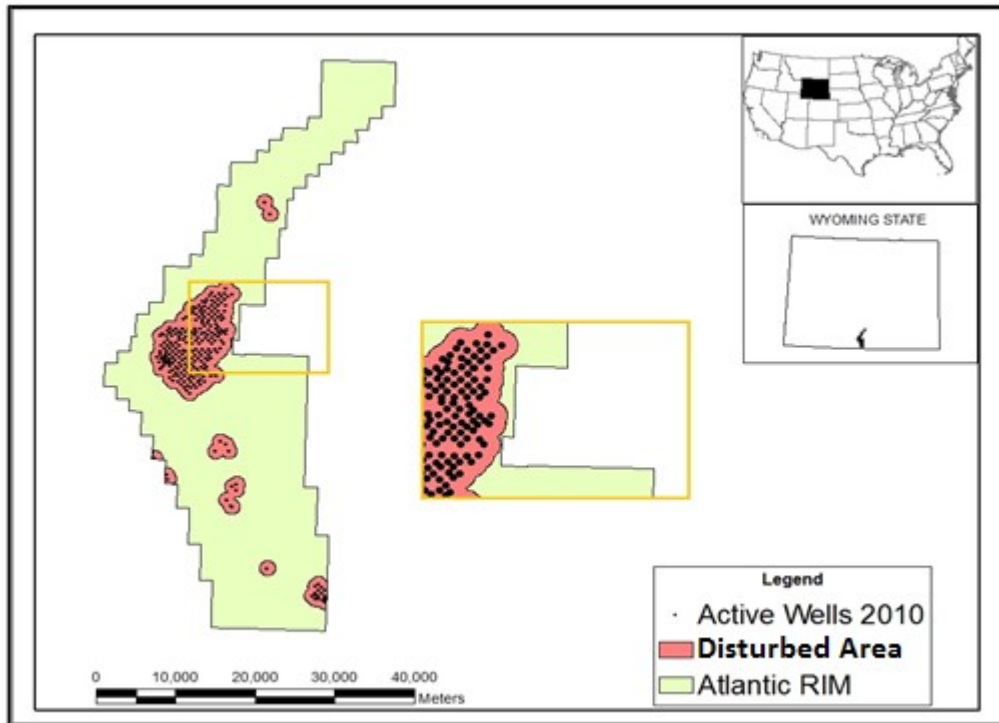
**Figure 1:** Grass (Gr) and shrubs (Sh) are the two plants and are composites of various species. (See Solow and Beet (1998) on lumping species in food webs.) Grass is comprised of Idaho fescue (*Festuca idahoensis*), prairie junegrass (*Koeleria macrantha*), bluebunch wheatgrass (*Pseudoroegneria spicata*), Thurber’s needlegrass (*Achnatherum thurberianum*), needle and thread (*Hesperostipa comata*), squirreltail (*Elymus elymoides*), and Sandberg bluegrass (*Poa secunda*). Shrubs are comprised of *Artemisia tridentata* from the Wyoming big sagebrush and the mountain big sagebrush ecosystems (Davies, 2010, p. 462-4). Grass-eating herbivores are elk (*Cervus elaphus*: El), black-tailed prairie dogs (*Cynomys ludovicianus*: Pd), black-tailed jackrabbits (*Lepus californicus*: Jr), cattle (Ca) and grasshoppers (Gh). The latter are a composite of *Orthoptera acrididae*, *Ageneotettix deorum*, *Melanoplus infantilis* Scudder, *Aulocara elliotti*, *Melanoplus sanguinipes*, and *Melanoplus bivittatus* (Onsager, 1984, p. 200). Shrub-eating herbivores are mule deer (*Odocoileus hemionus*: Md), pronghorn antelope (*Antilocapra americana*: An), black-tailed jackrabbits (Jr), and sage grouse (*Centrocercus uophasianus* Bonaparte: Sg). Carnivores include black-footed ferrets (*Mustela nigripes*: Fe), swift fox (*Vulpes velox*: Sf), and coyotes (*Canis latrans*: Co). Ferrets are endangered owing to massive eradication campaigns against their main prey prairie dogs (Bagheera 2016). Although they may not currently be present in ARNG, following decades of captive breeding 518 individuals were released into nearby Shirley Basin, Wyoming starting in 1991 (Jachowski and Lockhart, 2009).

**Figure 2. Undisturbed and Disturbed Areas**



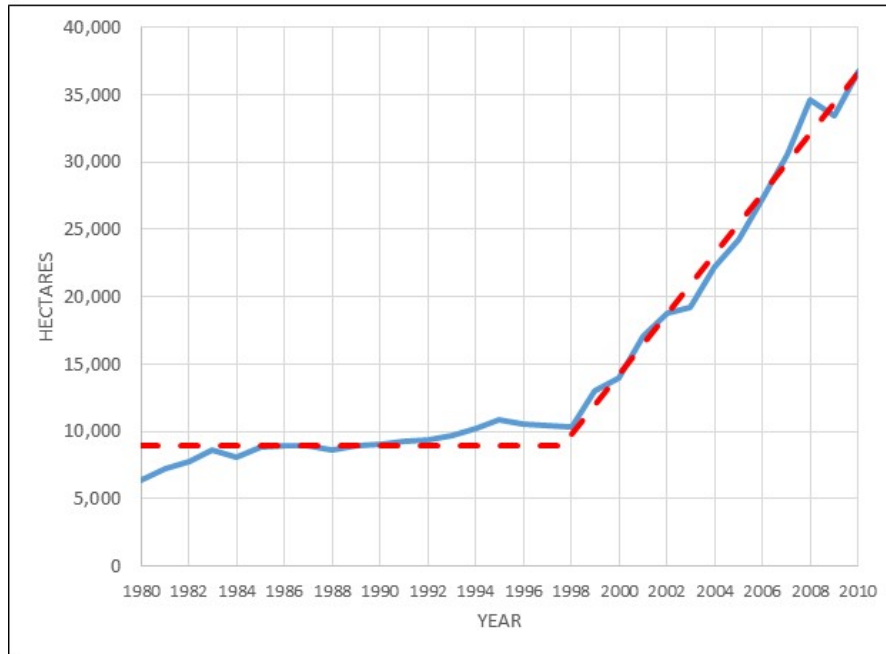
**Figure 2:** The top schematic represents the elk and sage grouse habitat loss or disturbed area (DA) around a well pad which is represented by a point. The DA has a radius of 1000 m and any area not within a DA is the undisturbed area (UA). Other herbivores and all carnivores forage and prey, respectively, in both DAs and UAs. The bottom schematic shows that the total DA is the union of all DAs, and the total UA is the area of ARNG minus that union.

**Figure 3. ARNG Development**



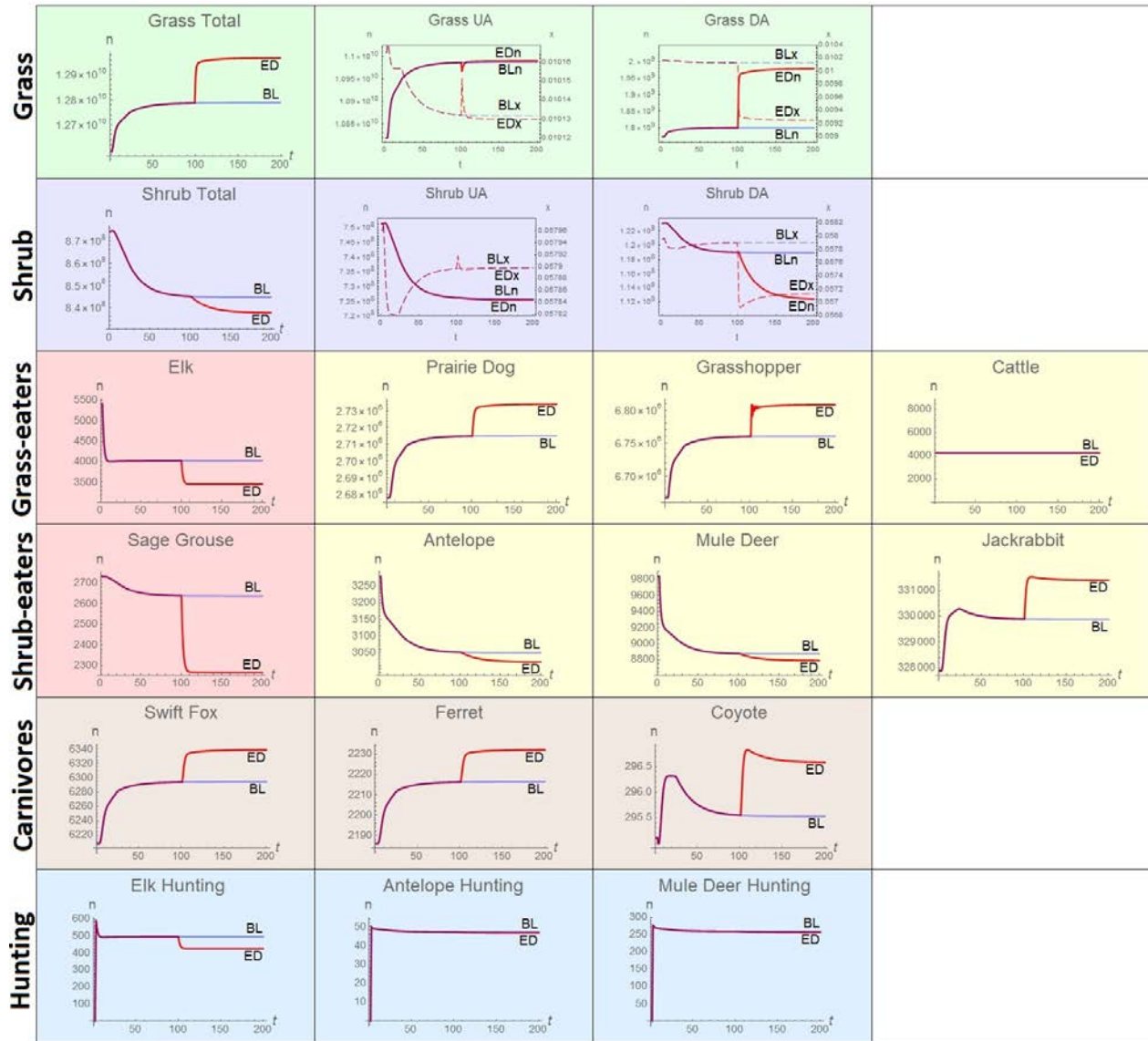
**Figure 3:** The upper right two maps show the location of Wyoming and the ARNG inside Wyoming. The yellow area represents the ARNG, the black dots represent the location of active natural gas wells in 2010, and the peach-colored area represents the total DA when a radius of 1 km around each well is used.

**Figure 4. Disturbed Area in the Atlantic Rim Area (1980-2010)**



**Figure 4:** The solid line shows the total DA in the ARNG region from 1980 to 2010 assuming a 1,000 m radius around all types of natural gas wells, active or non-active. Importantly, even abandoned wells are relevant when measuring the impacts of ED. For example, Minnick and Alward (2015) find evidence that it is necessary to consider feedbacks between soil and vegetation for the recovery of ecosystem functions in the DAs. The dashed line shows the piecewise linear trend of the total DA. Although it has generally been growing, the total DA started increasing sharply in 1998. The first segment of the solid line shows a practically zero growth between 1980 and 1998, while the second section of the solid line shows fast growth after 1998. Given this pattern, there have been approximately 15 years of major ED in the region to date.

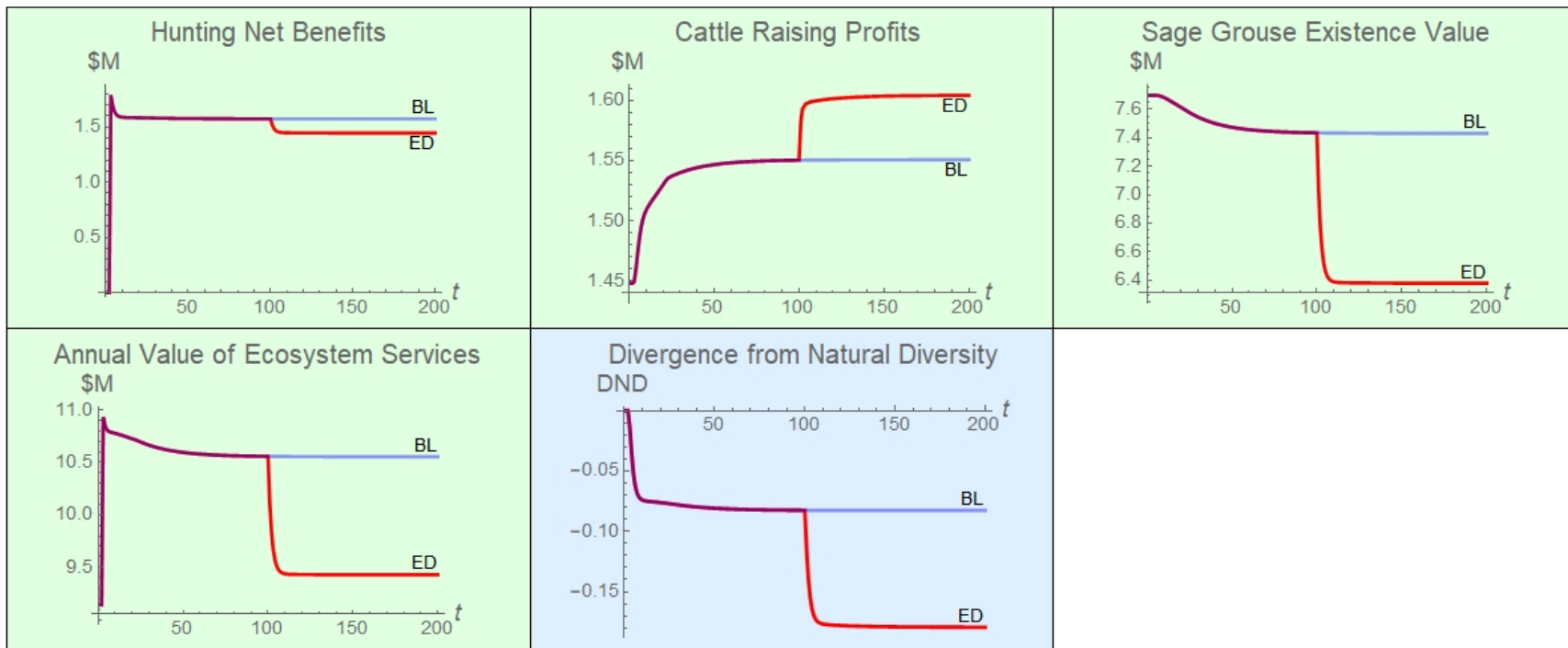
**Figure 5. Plant and Animal Populations after the Introduction of Hunting and ED**



**Figure 5:** The graph shows the simulated time paths of plant biomasses and the populations of all species for two scenarios: (1) where hunting and ED are introduced and (2) where only hunting is introduced, the base line (BL). The horizontal axes show the time periods ( $t$ ), the vertical axis shows either the species' populations ( $n$ ) with a solid line or an individual plant's biomass ( $x_i$ ) with a dashed line. The plant populations in the first two rows are divided into three categories: the total ARNG population, the population in the UA, and the population in the DA. The total population equals the sum of the UA and DA populations. The DA/UA distinction becomes important after period 100 when ED is introduced. Even though ED is not introduced until period 100, all the plants are divided into UA and DA starting in period 1. Prior to ED introduction, plants in UA and DA are subject to the same set of foragers.

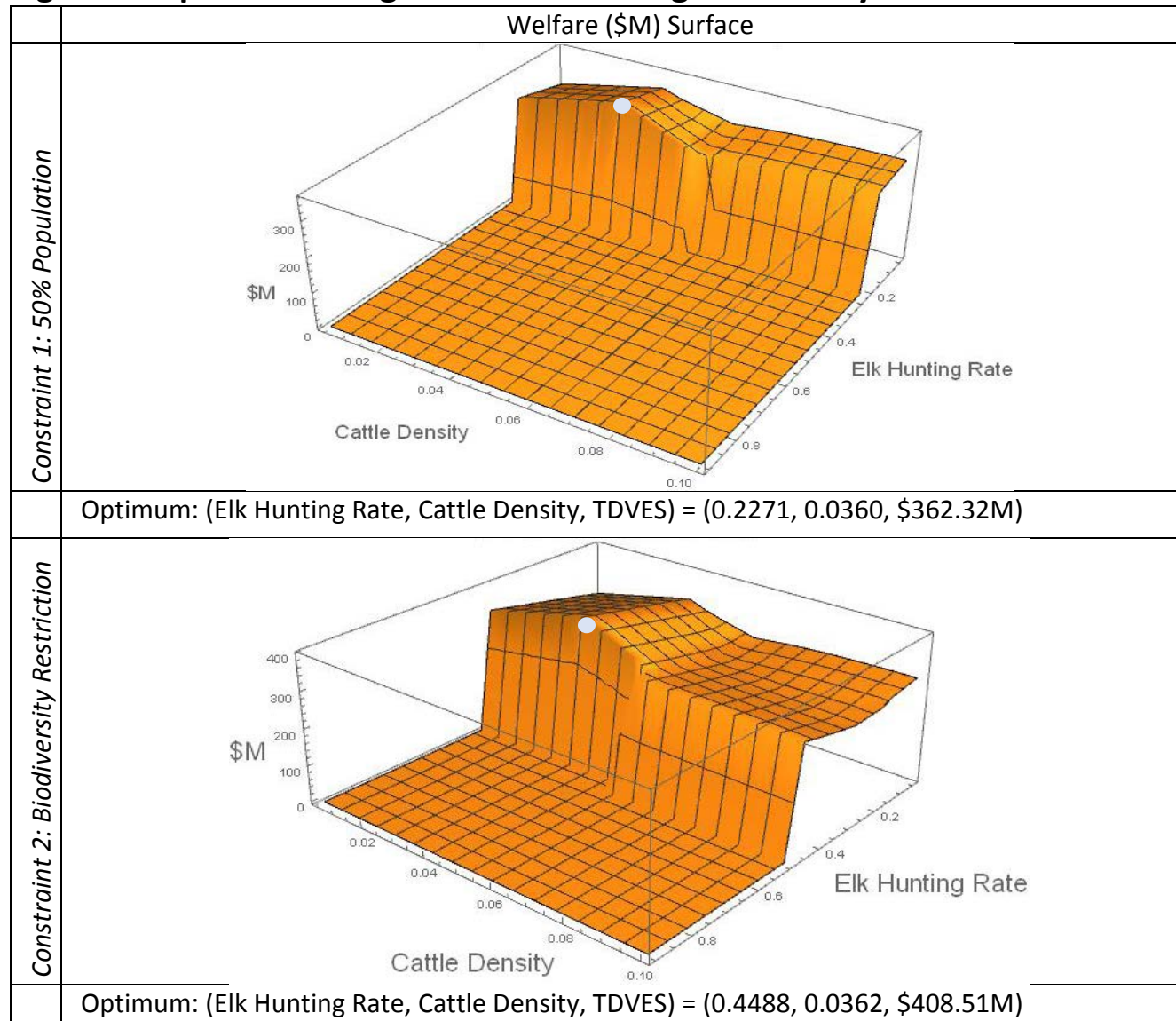


**Figure 6. Ecosystem Services and Biodiversity**



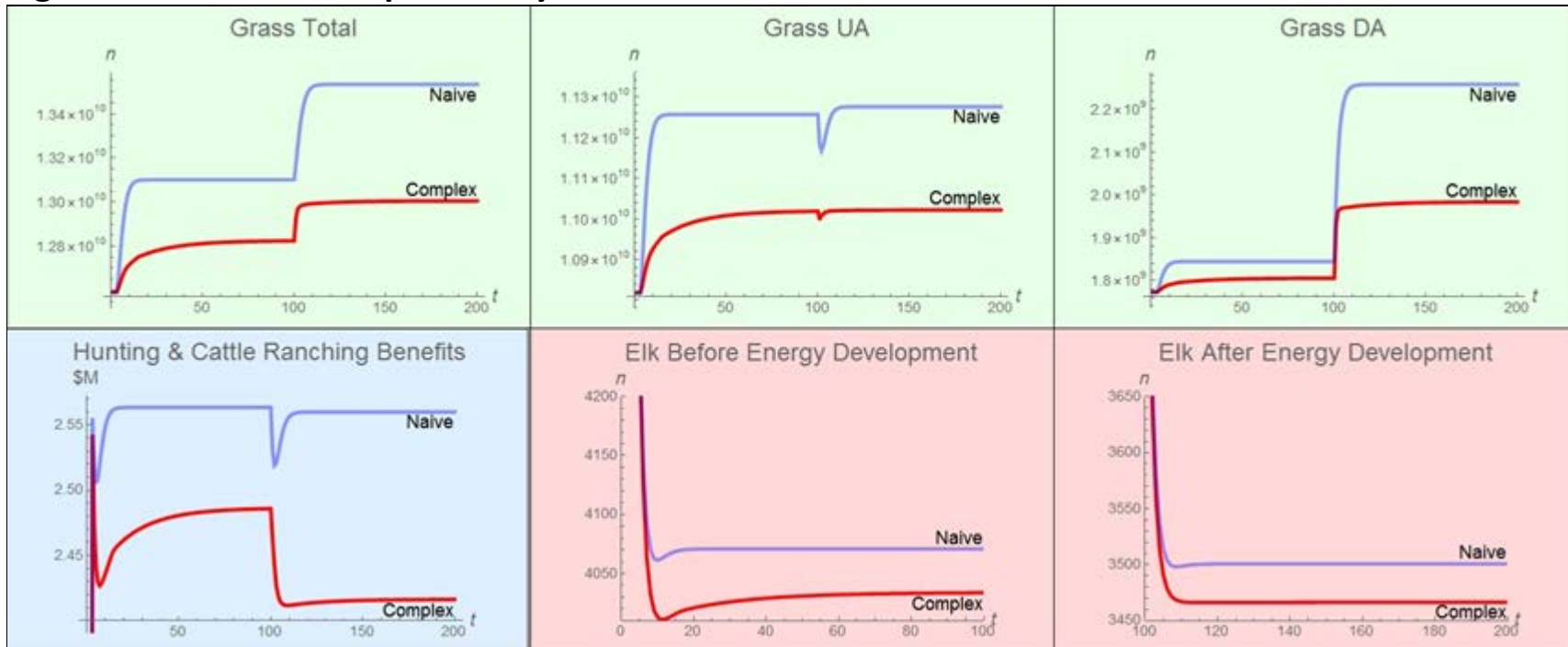
**Figure 6:** The first four graphs show the value of ecosystem services divided into the three types of service and the their total value, and the fifth graph shows the DND index. The simulation starts at the natural SS; hunting is introduced in period 3; and ED is introduced in period 100. The vertical axis on the four graphs is in millions of dollars. BL and ED have the same meaning as in figure 5. The existence value of sage grouse plus the annual profits of cattle ranching are approximately \$9.14 million at the natural SS. There is a sharp increase in the annual value of the ecosystem services when hunting is introduced, but it is accompanied by a loss of biodiversity. After introduction, hunting net benefits are initially high given the large populations of the hunted species, but they decrease slightly when the populations of the hunted species decline. Furthermore, the introduction of elk hunting increases the relative abundance of grass, which in turn increases cattle profits.

**Figure 7. Optimal Management of the Rangeland Ecosystem**



**Figure 7:** The heights of the surfaces represent TDVES after ED is introduced for different combinations of elk hunting and cattle densities, and for two different constraints on species' populations in the rangeland ecosystem. The dot indicates the location of the optimum.

**Figure 8. Naïve vs. Complex Ecosystem**



**Figure 8:** The graphs compare the results of considering all species in a complex food web versus considering only grass, elk and cattle in a naïve food web. Elk hunting and cattle ranching profits are measured in millions of dollars per year.

**Table 1. Elk, Antelope, and Mule Deer Harvested in the ARNG in 2014**

Concept	Elk	Antelope	Mule Deer
Total harvest in hunting units	1,818	819	280
Total unit area inside ARNG	32.7%	34.1%	17.9%
Total harvested in ARNG	595	279	50
Equilibrium population	5,421	9,837	3,279
Percent harvested	11.0%	2.8%	1.5%
Average success rate	47.7%	53.1%	9.1%

**Table 1:** Estimation using the data from harvest reports (WGFD 2014). The hunting areas (or units) intersect with the ARNG region but none falls entirely within it. We calculated the percentage of each unit that falls inside the ARNG region, and then assumed that the same percentage of elk, antelope, and mule deer from the total animals hunted were harvested inside the ARNG region.

**Table 2. Payouts and Nash Equilibria for Non-Cooperative Game between the BLM and WGF**

BLM		CATTLE GRAZING DENSITY																							
WGF		0.005	0.01	0.015	0.02	0.025	0.03	0.035	0.04	0.045	0.05	0.055	0.06												
ELK HUNTING RATE	0	21.88	9.78	21.91	19.55	21.94	29.33	21.97	39.10	22.00	48.88	22.04	58.66	22.08	<b>68.43</b>	22.36	46.22	22.69	19.82	22.96	-4.85	23.18	-26.75	23.36	-29.81
		216.11	247.77	216.71	258.18	217.31	268.58	217.90	278.98	218.54	289.42	219.16	299.85	221.69	312.21	237.40	305.99	254.27	296.78	267.17	285.28	277.15	273.57	285.32	278.86
	0.03	30.34	9.78	30.35	19.55	30.37	29.33	30.39	39.10	30.40	48.88	30.42	58.66	30.44	<b>68.43</b>	30.52	47.42	30.76	21.08	30.97	-3.57	31.15	-26.54	31.30	-29.81
		215.80	255.91	216.39	266.30	217.00	276.69	217.60	287.09	218.22	297.50	218.84	307.91	219.46	318.33	225.75	303.68	241.35	293.18	254.94	282.34	265.39	270.00	273.72	275.20
	0.06	38.03	9.78	38.03	19.55	38.04	29.33	38.04	39.10	38.05	48.88	38.05	58.66	38.06	<b>68.43</b>	38.07	47.96	38.17	22.25	38.33	-2.37	38.47	-26.19	38.58	-29.81
		215.49	263.30	216.09	273.67	216.69	284.05	217.30	294.44	217.84	304.77	218.53	315.23	219.15	325.64	219.67	305.70	229.53	289.95	242.35	278.32	253.41	265.68	262.24	271.01
	0.09	45.19	9.78	45.19	19.55	45.18	29.33	45.17	39.10	45.17	48.88	45.16	58.66	45.16	<b>68.43</b>	45.16	48.10	45.16	23.05	45.24	-1.21	45.34	-25.63	45.43	-29.81
		215.19	270.16	215.79	280.52	216.39	290.89	216.99	301.27	217.60	311.65	218.22	322.04	218.84	332.43	219.36	312.62	220.30	288.52	230.63	274.66	241.15	260.87	250.23	265.85
	0.12	51.99	9.78	51.97	19.55	51.95	29.33	51.93	39.10	51.92	48.88	51.90	58.66	51.88	<b>68.43</b>	51.88	48.24	51.88	23.20	51.88	-0.38	51.92	-24.72	51.98	-29.81
		214.90	276.66	215.49	287.01	216.09	297.37	216.69	307.73	217.30	318.10	217.87	328.43	218.54	338.85	219.06	319.17	219.48	294.56	221.03	272.54	229.60	256.80	238.27	260.43
	0.15	58.52	9.78	58.49	19.55	58.47	29.33	58.44	39.10	58.41	48.88	58.38	58.66	58.36	<b>68.43</b>	58.34	48.38	58.34	23.33	58.33	-0.19	58.33	-23.82	58.34	-29.81
		214.60	282.90	215.19	293.24	215.79	303.58	216.39	313.93	217.00	324.29	217.62	334.66	218.24	345.02	218.76	325.48	219.18	300.85	219.52	277.65	220.44	254.95	227.02	255.54
	0.18	64.88	9.78	64.85	19.55	64.81	29.33	64.77	39.10	64.73	48.88	64.70	58.66	64.66	<b>68.43</b>	64.64	48.52	64.63	23.47	64.62	-0.07	64.61	-23.65	64.60	-29.81
		214.30	288.96	214.89	299.29	215.49	309.63	216.09	319.97	216.70	330.31	217.31	340.67	217.92	351.01	218.45	331.61	218.88	306.97	219.21	283.76	219.49	260.45	219.72	254.51
	0.21	71.15	9.78	71.10	19.55	71.05	29.33	71.00	39.10	70.96	48.88	70.91	58.66	70.86	<b>68.43</b>	70.84	48.66	70.82	23.60	70.80	0.06	70.79	-23.51	<b>70.78</b>	-29.81
		214.00	294.92	214.59	305.24	215.19	315.57	215.79	325.90	216.39	336.23	217.01	346.57	217.62	356.92	218.14	337.64	218.57	312.99	218.91	289.77	219.18	266.47	219.41	260.38
	0.225	74.26	9.78	74.21	19.55	74.15	29.33	74.10	39.10	74.05	48.88	74.00	58.66	73.94	<b>68.43</b>	73.91	48.73	73.89	23.67	<b>73.88</b>	0.13	<b>73.86</b>	-23.44		
		213.85	297.88	214.44	308.20	215.03	318.51	215.63	328.84	216.24	339.17	216.85	349.50	217.47	359.84	217.99	340.64	218.41	315.97	218.75	292.76	219.03	269.45		
	0.226	74.47	9.78	74.41	19.55	74.36	29.33	74.31	39.10	74.25	48.88	74.20	58.66	74.15	<b>68.43</b>	<b>74.12</b>	48.74	<b>74.10</b>	23.67						
		213.84	298.08	214.43	308.39	215.02	318.71	215.62	329.03	216.23	339.36	216.84	349.70	217.46	360.04	217.98	340.84	218.40	316.17						
0.227	74.67	9.78	74.62	19.55	74.57	29.33	74.51	39.10	74.46	48.88	74.41	58.66	<b>74.35</b>	<b>68.43</b>											
	213.83	298.28	214.42	308.59	215.01	318.91	215.61	329.23	216.22	339.56	216.83	349.89	217.45	360.23											
0.228	74.88	9.78	74.83	19.55	74.77	29.33	74.72	39.10	74.67	48.88	<b>74.61</b>	<b>58.66</b>													
	213.82	298.47	214.41	308.79	215.00	319.10	215.60	329.43	216.21	339.75	216.82	350.09													
0.229	75.09	9.78	75.03	19.55	74.98	29.33	74.93	39.10	<b>74.87</b>	<b>48.88</b>															
	213.81	298.67	214.40	308.98	214.99	319.30	215.59	329.62	216.20	339.95															
0.230	75.29	9.78	75.24	19.55	75.19	29.33	<b>75.13</b>	<b>39.10</b>																	
	213.80	298.87	214.39	309.18	214.98	319.50	215.58	329.82																	
0.231	75.50	9.78	75.45	19.55	<b>75.39</b>	<b>29.33</b>																			
	213.79	299.06	214.38	309.38	214.97	319.69																			
0.232	<b>75.71</b>	<b>9.78</b>																							
	213.78	299.26																							

**Table 2:** The table represent the TDVES for different combinations of elk hunting and cattle grazing densities in the context of a non-cooperative game between the BLM and the WGF. Rows (columns) represent elk hunting rates (cattle grazing densities) indirectly chosen by the WGF (BLM). Each cell contains the values (million dollars) of the net benefits hunting (top left), cattle ranching profits (top right); existence value of sage grouse (lower left); and TDVES (lower right). Bold values indicate the best responses of WGF (BLM) to the actions of the BLM (WGF), shaded cells are NE, and blank cells represent combinations that violate constraint 1.