IV. THREATS AND ANALYSIS

In this section we summarize the potential threats to current GUSG populations and/or habitats, and analyze (1) the risk of permanent GUSG habitat loss through urban development, (2) potential habitat linkages among populations, (3) population viability, (4) population augmentation options, (5) GUSG population size in relation to the amount of available habitat, and (6) population targets.

A. Threats Potentially Affecting GUSG

These topics are listed and discussed in alphabetical order. For the topics containing more detail, a short conclusion follows the general discussion.

Disease and Parasites

Nothing has been published about the types or pathology of diseases in GUSG; however, multiple bacterial and parasitic diseases have been documented in GRSG (Patterson 1952, Schroeder et al. 1999). Most infections reported produce no, or minor, ill effects in sage-grouse (Patterson 1952). Rangewide impacts of bacterial or parasitic diseases on sage-grouse have not been reported.

West Nile Virus

West Nile virus (WNV) is a relatively new and potentially important disease for sage-grouse (Naugle et al. 2004). The virus has rapidly spread through the country, occurring in all states (except Washington and Oregon) by December 2003. Transmission occurs when mosquitoes acquire the virus by biting an infected bird, and then transfer it by feeding on a new host (avian or mammalian). WNV causes illness and death in birds that have no natural resistance to the infection. Mortalities from the virus have been discovered in 234 bird species (Centers for Disease Control and Prevention 2004b). Most mortalities have occurred in the family Corvidae, which includes crows, ravens, and jays. The data are based on specimens brought to local health departments by the public for testing (Centers for Disease Control and Prevention 2002) and on laboratory tests (Komar et al. 2003). Six North American gallinaceous species, including the GRSG, are known to be susceptible to the virus (U. S. Geological Survey 2003).

In 2003, WNV was detected in all but 1 county in Colorado (San Miguel County) where GUSG are known to occur, but there have been no recorded cases of the virus in GUSG (Colorado Department of Public Health 2004). Although the virus was detected in wild bird, horse and/or mosquito samples, it does not appear the virus was widespread or that contact with sage-grouse was significant in 2003. There were no reports of the virus in San Juan County, Utah in 2003 (Centers for Disease Control and Prevention 2003). In 2004, WNV was detected in areas with GUSG in Gunnison, Mesa, and Montrose Counties in Colorado, and in San Juan County, Utah (Centers for Disease Control and Prevention 2004b, Colorado Department of Public Health 2004).

In early August 2004, WNV was confirmed in the remains of a radio-collared female GRSG in south Routt County, Colorado. Eight other radio-collared GRSG in the area continued to show normal activity. The virus was detected in areas with greater sage-grouse in Eagle, Garfield, Grand, Routt and Rio Blanco Counties in Colorado (Centers for Disease Control and Prevention 2004b, Colorado Department of Public Health 2004).

In 2004, the CDOW, in collaboration with the Colorado Mosquito Control Company (CMC) conducted a study to monitor both GUSG and mosquito populations in the Gunnison Basin. The CMC collected mosquitoes, determined species, and sent samples of the genus *Culex* to the Colorado Department of Health to test for the presences of WNV. *C. tarsalis* and *C. pipiens* are considered to be the most serious threat for transmitting the virus to avian species (Turell et al. 2001, Foster and Walker 2002, Centers for Disease Control and Prevention 2004a).

Simultaneously, the CDOW used radiotelemetry to monitor the survival and movement of GUSG in the Gunnison Basin. Monitoring movement will help assess the potential for GUSG to come in contact with *Culex* species. The majority of mosquito species (> 85%) were *Aedes* spp., *Ochlerotatus* spp. or *Culiseta* spp. These species are not considered to be effective in transmitting the virus to avian species (Turell et al. 2001, Foster and Walker 2002, Centers for Disease Control and Prevention 2004a). *C. tarsalis* has made up < 15% of the samples collected so far. No *C. pipiens* were found in any samples. Fourteen pooled *C. tarsalis* samples were tested and all were found to be negative for WNV. CDOW used radiotelemetry to track 16 GUSG (13 males, 3 females). Locations were recorded every 1-2 days. There were no mortalities among the marked grouse, nor were any observed among unmarked grouse.

Coccidiosis and Tularemia

Intestinal coccidiosis (caused by the protozoan *Eimeria* spp.) has occurred in outbreaks that seriously impact local sage-grouse populations (Carhart 1943, Grover 1944, Patterson 1952, Honess and Post 1968), but such outbreaks do not appear common. Typically, outbreaks have occurred in summer when grouse may concentrate around water sources (Carhart 1943, Wallestad 1975). Disease transmission occurs through ingestion of water contaminated by infected feces. Birds that recover from the infection carry some level of immunity (Friend and Franson 1999). A second disease reported to have local population effects in sage-grouse is tularemia, which is caused by the bacterium *Francisella tularensis* (Parker et al. 1932, Friend and Franson 1999). This disease is transmitted by ticks and is also uncommon (Friend and Franson 1999).

Avian Malaria

Avian malaria, caused by the protozoan *Plasmodium pediocetti*, does not decimate sage-grouse populations but may still have a negative effect on populations. The daily cycle of the disease causes infected birds to be less active in morning hours, thus affecting male sage-grouse courtship and reproductive success (Boyce 1990, Johnson and Boyce 1991). Avian malaria is transmitted by biting flies (Friend and Franson 1999).

Diseases of Captive Birds

The release of gallinaceous game birds, either native or exotic, in GUSG counties is a potential avenue for disease introduction. All game birds in the grouse (Tetraoninae) and wild turkey (Meleagridinae) subfamilies that are imported to Colorado must be certified as disease-free by an accredited veterinarian prior to importation. Disease testing occurs for *Salmonella pullorum* (Pullorum disease) and 3 species of *Mycoplasma* bacteria (*M. gallisepticum*, *M. meleagridis*, and *M. synoviae*). Pullorum disease and *Mycoplasma* spp. are of interest because of their potential impact if transmitted to captive poultry, not necessarily to wild galliforms. The Colorado Division of Animal Industry (state veterinarian) also requires testing for other diseases known to affect poultry, such as Newcastle disease. Disease testing is required if there has been a recent occurrence of Newcastle disease in the state/region exporting the birds. Birds such as pheasants, chukar, and quail may be imported into Colorado without disease screening of any kind. These are the 3 species most frequently found on game farms (Colorado Division of Wildlife 2004c). In 2003, there were commercial game bird farms licensed by the CDOW in 5 GUSG-inhabited counties (Delta, Dolores, Mesa, Montrose, and Saguache). An additional potential disease risk to sage-grouse could result from transportation of wild gallinaceous birds within the state, which also requires no disease testing.

In Utah, the state veterinarian requires that pheasants, chukar, and quail be tested for *S. pullorum*. Wild turkeys brought into Utah must be tested for *Mycoplasma* spp., as must all gallinaceous birds brought into Utah from other states by UDWR.

Pullorum disease is not known to have infected wild sage-grouse, but is found in gallinaceous birds raised in captivity (Friend and Franson 1999). This bacterial disease is septicemic and may cause high mortality in poultry (Western Wild Health Committee 2003). It is transmitted from bird to bird by inhalation of airborne bacteria or by ingestion (pecking at surfaces contaminated by infected feces) (Friend and Franson 1999). Pullorum disease may also be transmitted from parent to egg and/or chick.

Mycoplasmosis is generally respiratory in nature, though it may be subclinical, and often causes decreased egg production and/or chick survival in captive birds (Friend and Franson 1999, Western Wild Health Committee 2003). Mycoplasmosis transmission can be through direct contact with infected birds or contaminated equipment, airborne via dust or droplets, or through eggs (Friend and Franson 1999).

Newcastle disease is a very contagious disease caused by a virus, of which there are many strains. Some forms of Newcastle disease are very lethal, often causing 100% mortality in poultry flocks (Friend and Franson 1999). This disease has been found in captive gallinaceous birds, but not in wild sage-grouse (Friend and Franson 1999). The disease can cause a variety of effects, including paralysis (Friend and Franson 1999). Disease transmission can be airborne, by bird to bird contact, or through contaminated equipment, food or water (Friend and Franson 1999).

Conclusions

WNV currently poses the greatest disease threat to wild GUSG. Despite the fact that the most common game farm birds do not undergo disease testing when imported to Colorado, disease transmission from introduced gallinaceous birds to GUSG remains a low possibility.

The 2 diseases known to have caused local population problems for sage-grouse, coccidiosis and tularemia, are uncommon. The diseases tested for in imported grouse and turkeys (*Salmonella* and *Mycoplasma*) are not known to have an impact on wild sage-grouse. The possibility for diseases of introduced or captive birds to spread to GUSG would become more important if efforts to raise GUSG in captivity were initiated.

Fire and Fuels Management

Prior to European settlement, the sagebrush landscape was a mosaic of different sagebrush species, in varying seral stages, occupying areas with different soil, topographic, and moisture conditions (Miller and Eddleman 2000). Fires historically occurred in many sagebrush communities on a regular basis, ranging in frequency from 10 – 100 years, depending on the sagebrush species and local factors (Young et al. 1979, Wright and Bailey 1982, Howard 1999, Miller and Eddleman 2000). Fires spread in a patchy manner, especially in Wyoming big sagebrush, responding to the landscape mosaic and the amount and distribution of fuel in the understory (Howard 1999, Miller and Eddleman 2000). Natural fire regimes in sagebrush-dominated communities probably occurred on a variety of scales, from small to large.

How fire affects a sagebrush community depends on multiple local characteristics such as dominant sagebrush species, aridity, soils, topography, and disturbance (Bunting et al. 1987, Miller and Eddleman 2000). The 3 primary sagebrush species present in GUSG habitat are Wyoming big sagebrush, mountain big sagebrush, and black sagebrush, although basin big sagebrush also occurs at some sites. All 3 species are killed by fire, but can reestablish (McMurray 1986, Bunting et al. 1987, Howard 1999, Johnson 2000, Miller and Eddleman 2000), although recovery timeframes vary, especially depending on environmental conditions (Bunting et al. 1987). Wyoming big sagebrush can reestablish (often slowly) from the seedbank, or from seed produced by plants surviving the fire or from plants adjacent to the fire (Bunting et al. 1987, Howard 1999). However, adequate moisture for sagebrush seed germination is not present in all years or seasons, especially in the areas where Wyoming big sagebrush grows (Monsen 2005). Furthermore, the open aspect of many burned sites allows wind to move snow around, reducing moisture entrapment and further drying out the soil (Monsen 2005). Mountain big sagebrush can reseed from surviving plants or plants in adjacent habitat (Johnson 2000). Generally this species grows in sites with more reliable moisture (aiding in seedling establishment), but individual populations vary in their fire tolerance (Monsen 2005). Black sagebrush reseeds from off-site plants (McMurray 1986) and from the seedbank (Monsen 2005). Fire does not spread readily through black sagebrush because of its generally sparse vegetation (McMurray 1986), but in some cases cheatgrass has increased the fuel load and allowed fire to eliminate black sagebrush stands (Monsen 2005).

Many new disturbance factors have been introduced to the sagebrush landscape since European settlement, including livestock grazing, aggressive alien plant species, cultivation, and multiple factors associated with an increased modern human presence on the landscape (Young et al. 1979, Miller and Eddleman 2000). The resulting altered landscape has experienced significant changes in fire frequency, distribution, and intensity. Two new scenarios have emerged in some sagebrush habitats in the West. In sagebrush stands where aggressive alien weed species such as cheatgrass have become established, fire frequency may increase (Whisenant 1990, Billings 1994, Tirminstein 1999, Miller and Eddleman 2000), eventually changing the community to an annual grassland (Young et al. 1979, Connelly et al. 2000, Miller and Eddleman 2000). If fire suppression has occurred, sagebrush communities can advance successionally to piñon-juniper (Burkhardt and Tisdale 1969, Young and Evans 1981, Miller and Rose 1995, Miller et al. 2000). Fire suppression in some sagebrush areas may have contributed to expanses of monotypic late-seral stage habitat

(CACP 1998, DCCP 1998, PMCP 2000) that has been postulated to be vulnerable to widespread, intense fires (Young et al. 1979). In instances where sagebrush habitat has become fragmented and limited, there is potential for fire to eliminate the existing seed source, reducing the likelihood of natural regeneration.

In most GUSG population areas, cheatgrass is not currently a dominant problem and has not affected fire regimes. However, in many areas fire suppression has occurred, possibly reducing both the amount and quality of sage-grouse habitat (GBCP 1997, CACP 1998, DCCP 1998, SMBCP 1998, PMCP 2000). This creates a situation where small prescribed burns may be useful to open up large stands of late-seral stage sagebrush (Klebenow 1972), or to reduce advancing piñon-juniper in sagebrush habitat (Burkhardt and Tisdale 1969, Bunting et al. 1987, Miller et al. 2000). Extreme caution must be the dominant philosophy because uncontrolled wildfire or prescribed fire could be catastrophic for the existing sagebrush community. In areas where woody species (including piñon-juniper and Douglas fir) are encroaching on sagebrush habitat, mechanical treatments may be more effective than prescribed fire in keeping treatment areas small (see ”Habitat Enhancement” rangewide strategy, pg. 214, and “Fire and Fuels Management” rangewide strategy, pg. 206).

In addition to reducing the density of woody vegetation, prescribed fire can also improve native forb and grass understory growth and forb nutrition (Bunting et al. 1987, Miller and Eddleman 2000, Wirth and Pyke 2003). Thus, well-managed prescribed burning can be used as a tool to improve sage-grouse habitat, but great care must be taken to avoid exacerbating existing problems and to ensure weed invasion does not occur (Connelly et al. 2000, Nelle et al. 2000, Monsen 2005). Invasive weed management should be a part of any prescribed fire planning in GUSG range. The goal should be to re-introduce fire in a way that most closely reflects natural fire at the landscape scale and that meets the needs of GUSG. Sage-grouse use of burned habitat has been the subject of debate, but it appears that sage-grouse will use burned sites as long as the sites provide appropriate cover and food resources during the season of use (Slater 2003).

Genetics

There has been much concern about the genetic viability of GUSG populations (Oyler-McCance 1999, Oyler-McCance et al. 1999). The persistence of a population is typically influenced more by demographic processes than by environmental or genetic effects (Lande 1988, Caughley 1994, Soulé and Mills 1998). But when the number of individuals in a population declines to a very low level, genetic factors and their interaction with demographic and environmental factors (i.e., "extinction vortices") become increasingly important (Gilpin and Soulé, 1986, Lande 1988, Soulé and Mills 1998).

Small populations face 3 primary genetic risks: inbreeding depression, loss of genetic variation, and accumulation of new mutations. In this section we discuss each of these threats to population viability, and their relevance to GUSG populations (a discussion of genetic effective population size is found in Appendix E).

Inbreeding Depression

In geographically closed populations inbreeding is inevitable because individuals will become increasingly related. The genetic consequence of inbreeding is increased homozygosity (Falconer 1981). This increase in homozygosity can have individual and population consequences (Fig. 21), by either increasing the phenotypic expression of recessive, deleterious alleles (Charlesworth and Charlesworth 1987), or by a reduction in the overall fitness of individuals in the population, assuming there is increased fitness in being heterozygous (i.e., the heterozygote advantage; Wright 1977), or both (Kimura and Ohta 1971).

INBREEDING →	**GENETIC CONSEQUENCES→**	**INDIVIDUAL CONSEQUENCES→**	**POPULATION CONSEQUENCES**
	Increased homozygosity; increased potential for expression of recessive deleterious alleles, or loss of heterozygosity	Increased susceptibility to disease; physical deformity; reduced reproduction, increased mortality	Decreased recruitment; declining population growth rate; greater probability of extirpation or extinction

Fig. 21. Diagram of consequences of inbreeding.

Available evidence suggests that inbreeding is virtually universal (however, see Ralls et al. 1984), but inbreeding depression is rare and has highly variable effects (see Lynch and Walsh 1998, Crnokrak and Roff 1999, and Hedrick and Kalinowski 2000, for reviews). In a survey of 36 mammalian species, Ralls et al. (1988) estimated that a degree of inbreeding equivalent to parent-offspring mating reduced viability in captivity by 33%. Crnokrak and Roff (1999) reviewed 35 studies of inbreeding depression in the wild and found that 141 out of 157 populations showed reduced fitness in inbred individuals. In addition, Crnokrak and

Roff (1999) found that inbreeding depression in the wild was substantially stronger than in captivity. This agrees with experimental work showing inbreeding depression to be stronger in more stressful environments (Miller 1994). However, the effect of inbreeding on fitness differs widely among species (Price and Waser 1979, Ralls and Ballou 1983, Ralls et al. 1988, Laikre and Ryman 1991).

There is no evidence of inbreeding or inbreeding depression in sage-grouse. However, studies of greater prairie chickens in Illinois showed that fertility and hatching success of greater prairie chickens were correlated with a reduction in genetic variation due to a population bottleneck caused by habitat loss (Bouzat et al. 1998a, Bouzat et al. 1998b, Westemeier et al. 1998). However, there was no evidence that inbreeding depression was the mechanism creating the loss of genetic variation or the loss in fitness.

It is likely the deleterious effects of inbreeding will occur faster in small populations than in large ones (Frankham 1995). In a randomly-mating, geographically-closed population, with discrete generations and modest variation in reproductive success, the average inbreeding coefficient (F_t) increases according to

$$(1) \qquad F_t = 1 - \left(1 - \frac{1}{2N_e}\right)^t$$

where t is the number of generations and N_e is the genetic effective population size (discussed in detail in Appendix E) (Hedrick 2000). Figure 22 graphs equation (1) for populations of 20, 100, and 500. Inbreeding occurs much faster in a population of 20 than a population of 500 individuals (Fig. 22). More specifically, the initial rate of increase is 25 times faster in a population of 20 than 500. This illustrates that avoiding small population size (even for a few generations) is essential for avoiding inbreeding and reducing the potential for inbreeding depression.

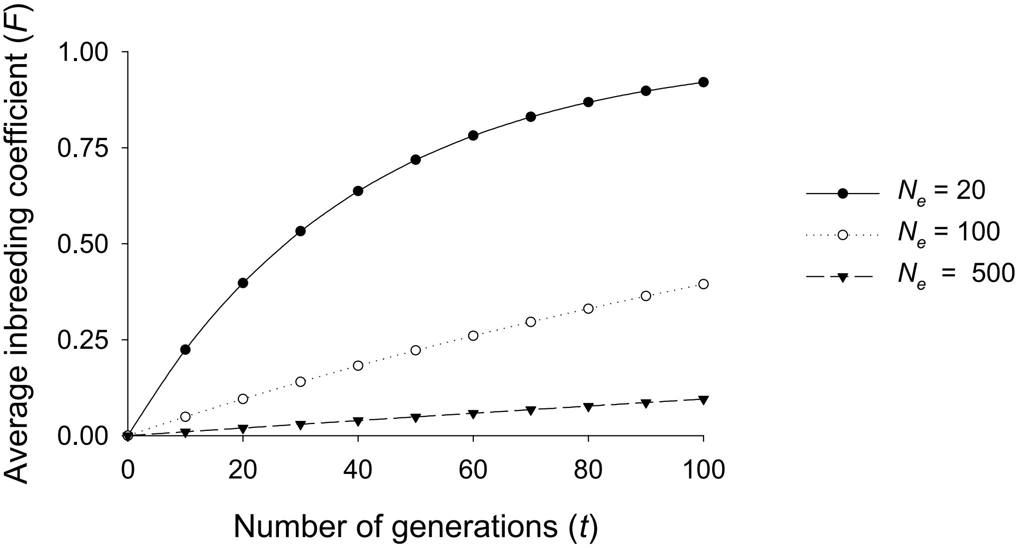

Fig. 22. The increase of average inbreeding coefficient as a function of genetic effective population size and the number of generations of breeding.

There is no consensus on how large a population must be to avoid biologically significant inbreeding depression, and there is little reason to believe that a single critical size or threshold exists. When inbreeding depression was first recognized as a threat to managed populations, Franklin (1980) and Soulé (1980) suggested that 50 individuals should be sufficient to avoid biologically significant inbreeding depression. This rule-of-thumb was based on anecdotal evidence that domesticated animals seemed to tolerate this level of inbreeding. Subsequent experimental inbreeding (in house and fruit flies), however, has shown that populations with a genetic effective size of 50 individuals often have substantial extinction rates (Latter et al. 1995, Bryant et al. 1999, Read and Bryant 2000). Although Franklin's (1980) and Soulé's (1980) guideline of 50 individuals has been shown to be too small, no larger size has emerged as a replacement guideline.

While inbreeding depression is considered a potential threat to small populations, we have no information to evaluate the relative threat of inbreeding to GUSG. We do not have adequate, long-term information on demographic rates (e.g., nest success, hatchability, juvenile or adult survival) to determine whether inbreeding depression is of concern. Inbreeding in small populations, such as sage-grouse, does not necessarily increase the likelihood of extinction (Caro and Laurenson 1994, Caughley 1994). Furthermore, it is possible that natural selection may purge deleterious alleles from the species, thereby eliminating the threat of inbreeding depression (Templeton and Read 1983, Lacy and Ballou 1998).

Loss of Genetic Variation

The loss of genetic variation, both within individuals and among populations, has the potential to reduce individual fitness and disrupt locally adapted populations (outbreeding depression). Adaptation to local changes in the environment is more likely to occur if there is large genetic variation among individuals in a population. In principle, populations with large amounts of genetic variation will have a greater chance of coping with climate change, exotic diseases, or other stresses. For example, O'Brien and Evermann (1988) found low variation in the major histocompatibility complex (an antigen-producing gene complex that plays a key role in the production of antibodies) in cheetahs, and documented a 50-60% mortality in cheetahs over a 3-year period due to a corona virus. They advocate that genetically depauperate populations face enhanced susceptibility to infectious disease or parasitic agents.

Genetic variation is introduced into populations by mutation. Natural and sexual selection work to eliminate deleterious alleles and retain favorable alleles. Genetic drift changes allele frequencies randomly, which leads to a net loss of genetic variation. For neutral loci, average heterozygosity (H) in a population declines according to

$$(2) \qquad H_{t+1} = H_t\left(1 - \frac{1}{2N_e}\right)$$

where t indicates the generation and N_e is the genetic effective size of the population (discussed in Appendix E). Note the similarity to Equation (1).

There is no consensus for how large populations must be in order to retain a level of genetic diversity that maximizes evolutionary potential. This question has been interpreted as how large a population must be in order for the processes of mutation and genetic drift to be balanced. Presumably, such a population would maintain its potential to adapt to local changes in the environment. Unfortunately, answering this question with confidence requires a more detailed understanding of mutation and heritability than is now available. Estimates currently range from 500 to 5000 individuals (Franklin 1980, Lande and Barrowclough 1987, Lande 1995), and these guidelines should be considered approximate.

GUSG have lower genetic diversity than GRSG (Oyler-McCance 1999, Oyler-McCance et al. 1999), but the consequences of this regarding threat of extinction are not well known. While genetic theory and empirical evidence suggest the loss of genetic diversity can have deleterious effects on reproductive fitness of individuals, the effect on the probability of extinction of a species can only be theoretically modeled (see "Population Viability Analysis", pg. 168). It has never been demonstrated that a population, much less a species, has gone extinct because of the loss of genetic diversity (Caro and Laurenson 1994).

Accumulation of New Mutations

Both genetic drift and natural selection change allele frequencies. The strength of natural selection is independent of population size, and the consequences of genetic drift are stronger in small populations than in large populations. One consequence of this is that if a population is small enough, slightly deleterious alleles behave as if they are neutral, and are almost as likely to increase as to decrease in frequency. When this is the case, slightly deleterious alleles can become fixed in the population. More specifically, alleles with selection coefficients less than $1/2N_e$ will respond to genetic drift in a manner similar to alleles that are selectively neutral (Kimura 1983).

Consider a population or species with a large number of individuals that then becomes reduced in size. Before population decline, deleterious alleles arise by mutation and are eliminated by selection. However, if the population declines in size enough, some deleterious mutations will become fixed. This accumulation of deleterious alleles may lead to extinction of the population, and this process is frequently called "mutational meltdown." The deleterious alleles responsible for mutational meltdown can be divided into 2 types: deleterious alleles existing at the time of population size reduction, and those that are new mutations. The negative impact of deleterious alleles existing at the time of population size reduction is essentially inbreeding depression. The mutational meltdown scenario predicts that in small populations the consequences of inbreeding depression will become magnified.

Mutational meltdown is probably the most controversial genetic threat to small populations. There is no doubt that genetic drift will cause mildly deleterious alleles to increase in frequency in small populations, but estimates for how large populations will have to be in order to prevent mutational meltdown vary dramatically. For example, Lande (1995b), Lynch et al. (1995), and Charlesworth et al. (1993) suggested that populations will need to have a genetic effective population size of 1000, 100, and 12 individuals, respectively, to avoid accumulating mutations. The wide discrepancy among these estimates is due to uncertainty regarding mutation rates. The process of mutation accumulation is slow when measured on a time scale relevant to most conservation applications. Even if mutational meltdown is a threat to small populations, it is expected to take hundreds to thousands of generations to occur.

Conclusions

Although there is no consensus for how large populations must be to avoid genetic problems associated with small population size, Shaffer (1987) states that populations smaller than a few hundred individuals warrant careful scrutiny in this regard. As noted above, it is highly debated whether reduced genetic variation reduces the viability of a population.

Small populations, (regardless of the amount of genetic variation) are at risk of extinction because of demographic fluctuations (Gilpin and Soulé 1986, Caughley 1994). Because of such factors, Lande (1988) and Caughley (1994) argued that, for conservation plans, demographic and behavioral concerns should be a higher priority than genetic concerns.

GUSG face many threats to survival, and these risks may interact. For example, climate change and exotic diseases may stress GUSG populations in the future, and populations with more genetic variation should be able to deal with these stresses better than populations with less genetic variation (e.g., Keller and Waller 2002 and references therein). The low levels of genetic diversity found in GUSG, particularly when compared to GRSG, may be of concern for the conservation of GUSG. However, even though research is needed to evaluate the impact of low genetic diversity on population viability (see "Research" rangewide strategy, pg. 247, Objective 5, Strategy 3), there is currently no direct evidence of inbreeding depression in GUSG. The maintenance of current genetic diversity in GUSG populations is addressed in "Population Viability Analysis" (pg. 168).

Grazing

Grazing is a major use of sagebrush rangelands in the West. Although it is likely that livestock and wild ungulate grazing (and associated land treatments), particularly historical over-grazing, have altered plant composition, increased topsoil loss, and increased spread of exotic plants (particularly cheatgrass), the impacts on sage-grouse are much less clear. It is recognized that current livestock stocking rates are substantially lower than historic levels, when GUSG numbers were presumed to be high. However, it is impossible to identify and/or quantify all other factors related to GUSG populations and habitat during the same time period. Thus, we are unable to derive a direct correlation between causative historic conditions. Because there are several recent thorough reviews of this topic we chose to primarily present quotations directly from these articles. Specifically, Rowland (2004) recently reviewed the literature on the effects of grazing on GRSG. Other recent reviews of the impact of grazing on sagebrush steppe habitats and bird communities were presented by Entwistle et al. (2000), Wambolt et al. (2002), Knick et al. (2003), Connelly et al. (2004), and Crawford et al. (2004).

General Debate

Wambolt et al. (2002:24) framed the debate over grazing in a perceptive manner:

> "Livestock grazing is possibly the most contentious, polarizing, politically charged and complex issue facing those who make and implement public land policy. Advocates for removing livestock argue that their "evidence" of ecological damage is incontrovertible, and their opponents argue that grazing can be managed in a sustainable and ecologically friendly manner (Clifford 2002). Attempts to integrate empirical results have not quelled the argument that "the science is out there" to bolster the argument of any of the various interests in this contentious debate (Vavra et al. 1994). In the middle are land managers, mostly from federal agencies. On one hand, anti-grazing interests accuse land managers of not making the difficult decisions necessary to get livestock off of public land. At the same time, grazing interests accuse land managers of making decisions based on weak or nonexistent science and/or data. The key policy issue before us is this: to restore grouse populations, sagebrush systems will have to be managed for the benefit of the bird. How this affects livestock grazing is a complex question. Overall, most of the research on sage-grouse habitat needs took place, and continues to take place, on habitats that are grazed. We can see from the range of data that grouse and grazing coexist in many, if not most, areas so we know with reasonable certainty that grouse and livestock are not mutually exclusive."

Rowland (2004:17-19) noted the difficulty in finding studies with direct evidence of grazing impacts on sage-grouse:

> "Effects of livestock grazing on vegetation species composition and structure in the sagebrush community have been well documented (Vale 1974, Owens and Norton 1992, Fleischner 1994, West 1999, Belsky and Gelbard 2000, Jones 2000, Anderson and Inouye 2001). However, few empirical studies report the responses of sage-grouse to grazing, and experimental research on effects of livestock on sage-grouse is lacking (noted by Braun 1987, Guthrey 1996, Beck and Mitchell 2000, Connelly et al. 2000b, Rowland and Wisdom 2002). No published studies on the effects of livestock grazing on sage-grouse were manipulative experiments in which cause-effect relationships could be measured. Instead, many studies imply negative effects of livestock grazing on sage-grouse by noting that grazing systems must be designed such that adequate herbaceous and shrub cover for nesting or brood rearing are maintained (e.g., Gregg et al. 1994, DeLong et al. 1995, Sveum et al. 1998). For example, DeLong et al. (1995) found that predation rates on sage-grouse nests in Oregon were negatively related to percent cover of tall grass and medium-height shrubs, and suggested that practices, such as livestock grazing, that remove grass cover may negatively affect nesting sage-grouse. Interactions of livestock grazing with other factors, such as wildfire, are complex and not widely studied."

Impacts on Sage-grouse Habitat

Rowland (2004:17-19) summarized studies that suggested negative impacts to sage-grouse habitat:

> "Beck and Mitchell (2000) summarized potential effects of livestock grazing on sage-grouse habitats, and cited only four references that provide empirical evidence of direct negative effects of livestock grazing on sage-grouse, as follows. Of 161 nests examined in Utah, two were trampled by livestock (one sheep, one cattle) and five were deserted due to disturbance by livestock (Rasmussen and Griner 1938). In Nevada, sage-grouse habitat in wet meadows was degraded through overgrazing by domestic livestock and altered system hydrology (Oakleaf 1971, Klebenow 1985; as reported by Beck and Mitchell 2000). Klebenow (1982) examined sage-grouse habitat use in relation to grazing at the Sheldon NWR in Nevada, where sheep and cattle had grazed for >130 yr. Dominant sagebrush species at the refuge were low sagebrush, mountain big sagebrush, and Wyoming big sagebrush. Grasses included Sandberg and Cusick's bluegrass (Poa secunda and P. cusickii, respectively) in wet meadows, and Sandberg bluegrass and mat muhly (Muhlenbergia richardsonis) in dry meadows. A rest-rotation system was implemented for cattle grazing in 1980 over the majority of the refuge, where season-long grazing had occurred historically; a smaller portion had previously been managed under deferred rotation. Meadows heavily grazed by livestock (e.g., with few forbs and grasses and dense shrubs present) were avoided by sage-grouse, with the exception of use for

> free water when available (Klebenow 1982). (No explicit definitions were provided for light versus moderate or heavy grazing.)"

In some cases, there were positive impacts on sage-grouse habitat (Rowland 2004:17-19):

> "Some positive effects of livestock grazing were noted. When cattle were introduced into a meadow with residual grass, sage-grouse initially preferred the grazed openings, which had an effective cover height (sensu Robel et al. 1970) of 5 to 15 cm, compared to 30 to 50 cm in the lightly grazed surrounding areas. Grouse avoided dense, ungrazed basin wildrye meadows but were observed in adjacent wildrye that was grazed. One 40-ha meadow that was lightly grazed by cattle (41 yearling heifers, 60 days in June-August) was used throughout the summer by sage-grouse and had more sage-grouse (100) than any other meadow on the refuge. Effective cover height in the meadow did not decrease below 5 cm during the summer."

Impacts on Sage-grouse Behavior and Demographics

Studies that focused on sage-grouse behavior and demographic parameter response to grazing reported mixed impacts (Rowland 2004:17-19):

> "Danvir (2002) reported two instances of nest abandonment related to livestock grazing in northern Utah during 7 yr of observations; one was caused by cattle, the other by sheep. Sage-grouse behavior on leks did not appear to be altered by the presence of cattle grazing (Danvir 2002). Sheep grazing in Idaho did not appear to disrupt use of leks by sage-grouse (Hulet 1983). Autenrieth (1981), however, cautioned against grazing sheep in sage-grouse winter habitat. He also suggested that livestock use of meadows occupied by sage-grouse, as well as livestock drives in sage-grouse habitat, could be detrimental to sage-grouse. In Wyoming, nesting densities of sage-grouse were considerably lower (10 nests/100 ha) in areas heavily grazed by domestic sheep compared to adjacent sites with moderate grazing (28 nests/100 ha) (Patterson 1952). Nest desertion caused by migrant bands of sheep also was documented (Patterson 1952).
>
> Heath et al. (1998) compared sage-grouse nesting and breeding success at three ranches with different grazing operations and levels of predator control in Wyoming. They found that, despite heavier livestock use (removal of >50% of annual herbaceous production, and grazing by both sheep and cattle) and long-term predator control on one ranch, nesting and breeding success of sage-grouse did not differ substantially among the three sites. Chick survival to 21 days was, however, greater on the ranch with lighter grazing, suggesting that predator control did not fully compensate for the greater reductions in herbaceous production (Heath et al. 1998). Further, hens were documented leaving the more heavily grazed ranch to nest elsewhere but returning to that ranch to rear broods (Heath et al. 1998). In a

> similar study, Holloran (1999) examined sage-grouse habitat use and productivity in relation to grazing management strategies at four ranches in southeastern Wyoming. He found no differences in nest success, brood survival, or numbers of chicks fledged among the ranches. Some differences in habitat use by sage-grouse were found among the ranches; however, these could not be ascribed to differences in grazing pressure, but were ascribed to differences in soil types and precipitation patterns (Holloran 1999). Above-average precipitation during the study, however, may have obscured any potential differences in habitat suitability for sage-grouse among sites. Neither of these studies employed control sites or replication."

Grazing Rotation, Intensity, and Timing

Rowland (2004:17-19) noted research specifically investigating different grazing methods:

> "Research on upland meadows in Nevada showed that pastures under a rest-rotation system provided better production of those forb species eaten by sage-grouse than did pastures that were not rested, but sage-grouse also used a pasture not grazed by cattle for 10 yr (Neel 1980). The author concluded that light grazing in meadows might enhance habitat for sage-grouse. Evans (1986, as reported in Beck and Mitchell 2000) also found that grazing by cattle stimulated production of forb species used by sage-grouse in upland meadows in Nevada".

Crawford et al. (2004:10) described the results of grazing on sagebrush plant communities as follows:

> "Research suggests that moderate livestock grazing or less in mid to late summer, fall, or winter is generally compatible with the maintenance of perennial grasses and forbs in sagebrush habitat (Pechanec and Stewart 1949; Mueggler 1950; Laycock and Conrad 1967, 1981; Gibbens and Fisser 1975; Miller et al. 1994; Bork et al. 1998). Herbaceous species in sagebrush plant communities are predominantly cool-season (C-3) plants that are vulnerable to defoliation during late spring and early summer. Heavy grazing (approximately 60% or greater utilization by weight) during this time has predictable results: 1) the vigor, yield, and cover of late-seral grasses and forbs decrease; 2) early-seral species (including annual grasses) may increase; 3) sagebrush density and canopy cover may increase (Craddock and Forsling 1938, Pechanec and Stewart 1949, Mueggler 1950, Laycock 1967, Bork et al. 1998); and 4) transition of sagebrush uplands to higher ecological status is inhibited (Mueggler 1950, Eckert and Spencer 1986, Laycock 1987)".

Recommendations

Crawford et al. (2004:11) discussed recommendations for timing of grazing to reduce impacts in riparian areas important for brood-rearing:

> "Timing of grazing greatly influences the effects of livestock grazing in meadows and riparian areas. These sites are particularly vulnerable in late summer when excessive grazing and browsing may damage riparian shrubs, reduce the yield and availability of succulent herbs (Kovalchik and Elmore 1992), and cause deterioration of riparian function over time (Klebenow 1985). However, moderate utilization by livestock in spring, early summer, or winter is sustainable in non-degraded meadow and riparian areas within sagebrush habitat (Shaw 1992, Clary et al. 1996, Mosley et al. 1997). Moderate use equates to a 10-cm residual stubble height for most grasses and sedges and 5-cm for Kentucky bluegrass (Mosley et al. 1997, Clary and Leininger 2000). Shrub utilization should not exceed 50-60% during the growing season, and at least 50% protective ground cover (i.e., plant basal area + mulch + rocks + gravel) should remain after grazing (Mosley et al. 1997). While hydrophytic shrubs may not directly serve as sage-grouse habitat, they do impact the stability of riparian and meadow habitats important to sage-grouse (Winward 2000). The length of time livestock have access to meadows may be more important than the level of utilization; it has been suggested that livestock access be limited to 3 weeks (Myers 1989, Mosley et al. 1997). In riparian and meadow habitat degraded by heavy livestock utilization, rest from grazing may be necessary for recovery (Clary and Webster 1989)."

Based on her literature review, Rowland (2004:24) recommended the following:

> "Manage livestock grazing through stocking rates and season of use on all seasonal ranges of sage-grouse to avoid habitat degradation (Paige and Ritter 1999, Beck and Mitchell 2000, Wisdom et al. 2000), especially on recently disturbed sites, such as those sprayed or burned (Braun et al. 1977). In nesting and brood-rearing habitats, ensure that grazing does not reduce herbaceous understory cover below levels that serve as a deterrent to potential predators of eggs and chicks (Connelly et al. 2000b, Hockett 2002). Healthy native understories also support insects and forbs that are important in diets of pre-laying hens and chicks (Johnson and Boyce 1990, Barnett and Crawford 1994, Drut et al. 1994b). Riparian areas and wet meadows used for brood rearing are especially sensitive to grazing by livestock; in these habitats, removal of livestock before the nesting season may be prudent (Beck and Mitchell 2000, Hockett 2002)."

Grazing Management: Related Structures and Activities

Structures and activities associated with grazing management can have multiple and variable affects on sage-grouse and sage-grouse habitat. Fences, corrals, windmills, and other structures related to livestock grazing can cause mortality of grouse by collisions, and provide perches that raptors may use, which could increase avian predation on grouse (Call and Maser 1985). Grazing structures, such as fences or stock tanks, also influence livestock distribution, which may have a positive or negative effect on GUSG and their habitat, depending on the resulting distribution. Livestock may trample grouse nests, and tightly herded livestock near nesting areas may cause nest abandonment. Conversely, salting locations or historic sheep bed grounds have been used as lek sites, as long as adjacent habitat continues to provide adequate hiding cover. Water developments may alter existing GUSG habitat by congregating livestock use in previously unused upland habitat, or by lowering water tables associated with riparian areas. However, water developments can also be used to improve overall riparian habitat condition by drawing livestock and wild ungulates away from previously degraded areas.

If the livestock reduce and degrade the understory significantly, hiding cover is reduced, potentially increasing predation on grouse. Vegetation manipulations to improve livestock forage can impact prairie grouse in different ways, depending on pre-existing and resulting habitat conditions. Removal of brush essential for grouse nesting or wintering cover can adversely impact grouse reproduction and survival, but brush treatments in less critical or degraded grouse habitat may increase habitat capability (Giesen and Connelly 1993, Giesen 1998, Connelly et al. 2000). In any case, vegetation treatments should be planned and implemented to maintain adequate suitable habitat for GUSG while other areas are recovering. Potential impacts, both positive and negative, from grazing related structures are usually localized in nature, and should be considered and addressed.

Wild Ungulate Effects on Sage-grouse Habitat

The effect of wild ungulates on GUSG and their habitat has been raised as an issue that requires greater understanding. Direct physical confrontation between GUSG and pronghorn antelope, elk, or mule deer is probably not a major concern, although an instance has been observed of an elk consuming sage-grouse eggs in Wyoming (Holloran and Anderson 2003). Others have observed mule deer and GRSG in Middle Park, Colorado using sagebrush areas for forage that otherwise would have been inaccessible if not for elk breaking trails and exposing sagebrush during deep snow conditions (D. Freddy, CDOW, personal communication). Hobbs et al. (1996) documented a decline in available dead perennial grasses and early spring live perennial grasses as elk densities increased. They further noted a small increase in quality of the forage as elk densities were increased, due to the increased digestibility and nitrogen content of new forage. They suggested that competition for forage between elk and domestic livestock will primarily only be a concern during heavy snowfall years, when wild ungulates are concentrated in large densities on lower elevation winter ranges (Hobbs et al. 1996). These conditions could negatively impact nesting cover for sage-grouse in extreme situations. Ultimately, further research needs to be conducted to fully understand the effects of wild ungulate grazing on sage-grouse.

Conclusions

Grazing by domestic and wild ungulates has played an important role in shaping the current vegetation communities in GUSG range. Rowland (2004) provided a literature review addressing the issue of the impacts of grazing on sage-grouse. This review suggested there has been no experimental research that demonstrates grazing alone is responsible for the reduction in sage-grouse numbers. However, several studies have been conducted that show grazing may be one factor that has contributed toward the condition of sage-grouse habitat and use of it by sage-grouse, and many studies have demonstrated the effect of various grazing practices on vegetation. In fact, grazing should be used as a tool to maintain and improve seasonal habitats for sage-grouse. Enough is known about GUSG habitat requirements to make reasonable recommendations to design management practices to maintain and improve GUSG habitat (see Appendix H, "GUSG Structural Habitat Guidelines"). Developing grazing systems and management plans that would achieve desired vegetation composition and structure, including shrubs, forbs, and grasses, should benefit both GUSG and domestic and wild ungulates.

Habitat Quality

Quality of GUSG habitat is not addressed in the RCP as a separate threat or issue because it is taken into account under the individual activities or factors that impact habitat quality. These include “Fire and Fuels Management” (pg. 107), “Grazing” (pg. 114), “Mining, Energy Development, and Human Community Infrastructure” (pg. 127), “Noxious and Invasive Weeds” (pg. 131), “Predation” (pg. 134), and “Weather/Drought” (pg. 143).

Hunting

GUSG are still classified as a game species in Colorado, but they have not been hunted in the Gunnison Basin since 2000, when the Colorado Wildlife Commission eliminated the season. The Wildlife Commission took this action when lek count population indices failed to meet the 5% per year increase (calculated as a change in a 3-year moving average), as suggested in the Gunnison Basin Conservation Plan. Hunting has not occurred in other Colorado populations of GUSG since at least 1995, when the Piñon Mesa area was closed, although a clerical error created an inadvertent season at Poncha Pass in 1992. Utah has not allowed hunting within the current range of GUSG since 1989.

Prior to 2000, hunting had occurred in the Gunnison Basin continuously since 1953, with variable season lengths and bag limits over the years. Season lengths and bag limits were generally restrictive (3-day season and bag and possession limit of 2 birds) from 1953-1978, but were then gradually liberalized. Season length increased to 7 days in 1978, 9 days in 1979, 16 days from 1980-88, 30 days in 1989-91, 34 days in 1992, and 33 days in 1993, then decreased to 16 or 17 days from 1994-1999.

Harvest estimates from the Gunnison Basin are available, but there is no information on fall population size, so a harvest rate cannot be determined. Harvest estimates for 1995-1999 (Table 19) should be unbiased since a permit or prior registration was required (which provided an accurate sampling frame) and several attempts were made to contact all permit holders by phone (which largely eliminated non-response bias). Prior to that time, estimates were generated from mail-in surveys, but numerous studies have shown that this type of survey tends to overestimate harvest because successful hunters are far more likely to respond than unsuccessful hunters.

Table 19. Hunter numbers and harvest (estimate ± 95% confidence limit) of GUSG in the Gunnison Basin, estimated from a telephone survey of permit holders (1995-1997) or Hunter Information Program registrants (1998-1999).

Year	Estimated Hunters	Estimated Harvest
1995	229 ± 15	298 ± 34
1996	197 ± 16	269 ± 37
1997	154 ± 8	191 ± 16
1998	187 ± 64	278 ± 4
1999	95 ± 40	127 ± 1

Whether hunting impacts the rate of population growth of sage-grouse remains a subject of some debate, since experimental research on the topic has not been conducted. It is not known to what extent fall hunting is compensatory or additive to natural mortality. Hunting is not likely to be completely compensatory or completely additive. Relative to other gallinaceous birds, sage-grouse are fairly long-lived, lay moderate sized clutches, and are relatively poor re-nesters, all of which suggest hunting may be more additive than for shorter-lived, more productive galliforms. Harvest oriented towards juveniles and/or males will be more compensatory than harvest of adult females. A reported direct recovery rate of 7-10% of banded birds in North Park, Colorado occurred from 1973 to 1990 (Zablan et al. 2003), a period when the number of displaying males counted increased from about 580 to over 1,500. That is not to suggest hunting caused the observed increase in displaying males, but it demonstrates that this increase was not prevented under significant harvest.

Based on telemetry studies in Idaho, Connelly et al. (2000) suggested that successful adult hens were disproportionately vulnerable to harvest. Johnson and Braun (1999) conducted a population viability analysis based on the North Park GRSG population and concluded hunting could reduce growth rates.

Lek Viewing

It has been postulated that human activities associated with the viewing of strutting sage-grouse on leks may have impacts on the grouse. The Gunnison local work group suggested that a decline in lek counts at one of the South Parlin area leks (in the early 1990's) may have been due to unmanaged lek viewing (GBCP 1997). However, there has been very little research on this topic. Profera (1985) conducted an experiment evaluating the distance at which GRSG responded to various disturbances, but the findings were inconclusive. She suggested that females flushed at larger approach distances to disturbance than males. She also found that male response to disturbance was related to the number of females present (Profera 1985). Baydack and Hein (1987) evaluated the response of sharp-tailed grouse to experimental lek disturbances in Manitoba, but the results are of questionable utility regarding lek viewing because all disturbances were placed on the center of the lek.

In the absence of controlled experiments, some information may be obtained from case studies of individual leks used for viewing activities in the past. For the past 25 years a GRSG lek in north-central Colorado (Coalmont lek) has been used for lek viewing. Birdwatchers are referred to this site by both the CDOW and the BLM. Since 1987, a detailed description of how to access this lek has been included in "A Birder's Guide to Colorado" (Holt and Lane 1987) and the location has been common knowledge among birders. Although GRSG populations have fluctuated over this period, the long-term trends appear stable. Counts on the Coalmont lek (Fig. 23) reflect both this variability and a similar long-term stable trend, suggesting there may be no impact to grouse due to lek viewing. The local Chamber of Commerce has conducted organized tours to view sage-grouse displays on an additional lek (Boetcher lek) since 1999 (2-3, 2-day tours per year in late April and early May). Lek counts have increased during this period (Fig. 23).

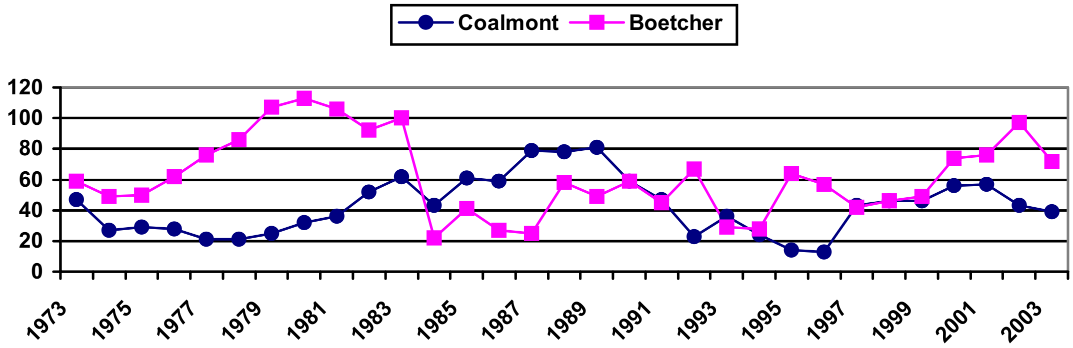

Fig. 23. Lek counts at 2 leks where public lek viewing is allowed in northern Colorado.

The CDOW established a Watchable Wildlife site at the Waunita lek in the Gunnison Basin. Recently, the site has been developed to include a rockwall and screen behind which a viewing trailer is hidden. The site also includes an outhouse, information panels, and parking area. In 2003, Sisk-a-dee, a non-profit conservation group, attempted to mitigate and document potential impacts of this site and corresponding GUSG viewing protocol on GUSG (Sisk-a-dee, unpublished report, 2003). Volunteers from Western State College (WSC) manned the site each day to help enforce compliance with viewing protocols. Viewers were asked to arrive in the dark at least 1 hour before sunrise and remain quietly in the trailer until all birds had departed the lek. Observers mostly followed the protocols, with 88% of vehicles complying, although there were a "few" violations of protocol. Human-related disturbances were credited with flushing grouse from the lek on only 2 of 34 days (although no human-related reasons for birds flushing are listed in the detailed table). Waunita has been a dedicated viewing lek since 1999, without obvious impacts to the number of males counted, in comparison to other leks in the same lek area (Doyleville, Fig. 24).

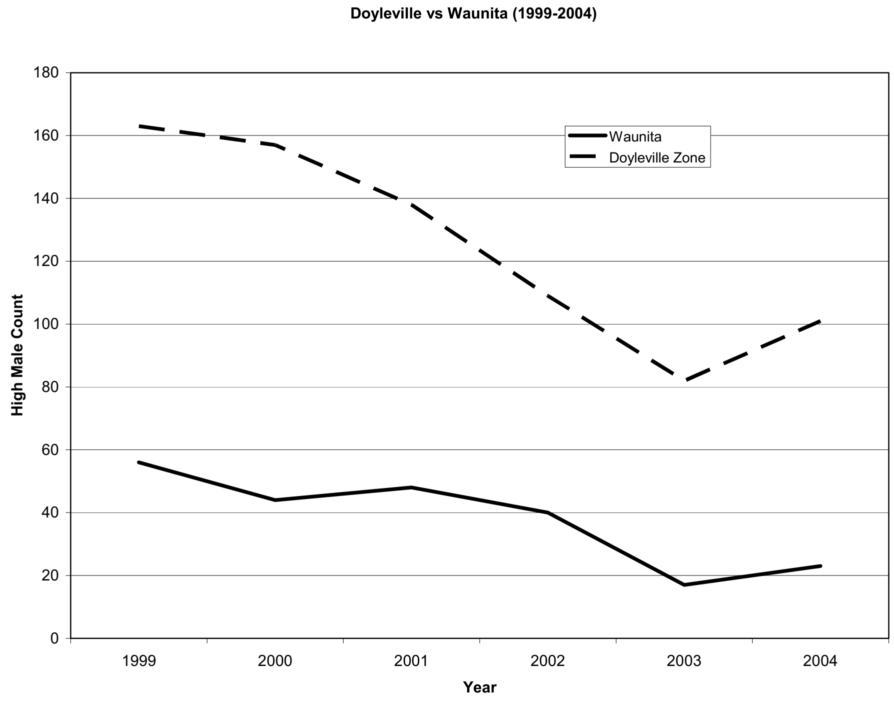

Fig. 24. Lek count data from Waunita Watchable Wildlife lek and other leks in the same (Doyleville) lek area.

Critical experiments to assess the impacts of lek viewing on sage-grouse (or other prairie grouse) have not been conducted and are needed. A first step is to critically evaluate sage-grouse biology and behavior and propose hypothetical impacts. An obvious disturbance would be to flush birds from the lek, which could hypothetically affect individuals and/or a population.

Sage-grouse are frequently flushed off leks by predators and respond to this disturbance in various ways. Boyko et al. (2004) described how GRSG react to predators visiting lek sites. It is likely that sage-grouse perceive humans on and near leks as predators and respond to them as they do to predators. Human disturbance, particularly if it is additive to disturbance by predators, could reduce the time the lek is active and reduce its size (by lowering attendance by "subordinate" males). Lek size has been postulated to affect the appeal of the group to females. Hence, a conceivable secondary effect might be to reduce the numbers of females mating there. Mating skew (the degree of domination by 1 male) tends to go up on small leks, so fewer males may participate in mating if disturbance reduces attendance and recruitment. However, this could be offset by lower skew on other leks attended by displaced birds. Disturbance during the peak of mating could result in some proportion of females not breeding, and females might shift breeding activities off-lek if disturbances were chronic.

Perhaps a more interesting possibility (from a population ecology perspective) is that disturbance might affect nesting habitat selection by females. Even though some females nest far from their lek of capture (and presumed mating), leks are typically close to areas in which females nest. If females move to poorer quality habitat farther away from disturbed leks, nest success could decline. If chronic disturbance caused sage-grouse to recruit to a new lek site away from preferred and presumably higher quality areas, both survival and nest success could decline. Whether any or all of these are real issues would presumably depend on timing and degree of disturbance.

Although impacts of lek viewing have not been demonstrated, studying the hypothesized impacts listed above would be challenging. In any case, it is prudent to take measures to avoid potential negative impacts to GUSG through careful management of lek viewing.

Mining, Energy Development, and Human Community Infrastructure

This section summarizes the effects on GUSG of mining, energy development, powerlines, pipelines, cables, wind turbines, communication towers, and roads. Although the activities and structures covered in this section are not all related, their potential risks to GUSG are similar: (1) habitat loss, fragmentation, and degradation; (2) direct disturbance and mortality of grouse; and (3) increases in predation pressure. Some of these activities have an obvious effect on habitat (e.g., habitat loss resulting from construction of utility or mining structures), and for others there is evidence of impacts on individual grouse (e.g., mortality from collisions with powerlines), but not necessarily on populations. Nevertheless, most of the threats in this section have not been rigorously studied in any prairie grouse species. The possible risks to GUSG of each type of activity or structure are discussed.

Habitat Loss, Fragmentation, and Degradation

Construction of any structure in sagebrush habitat (including overhead or underground powerlines, pipelines, and cables, wind turbines, communication towers, and fences) will inevitably result in some habitat loss. If construction is extensive, it could fragment habitat in a way that might negatively affect GUSG. Habitat fragmentation could force grouse to move across more open areas (less optimal habitat), potentially exposing them to predators more frequently than in contiguous habitat. Oil and gas drilling may decrease the amount of contiguous GUSG habitat through construction of wells, well pads, access roads, compressor stations, pipelines, power lines, produced water containment pits, and sludge pits (Massey 2001). Sand, gravel, and other mineral extraction may result in abandoned mining pits, mining infrastructure, access roads, and overburden placement in sagebrush habitat.

Lack of coal reserves in GUSG range negates any potential threat from coal mining (Bureau of Land Management 1998), but the distribution of other energy reserves is germane to several of the GUSG populations (Bureau of Land Management 1999, Fig. 25). The greatest possibility for oil and gas drilling in GUSG habitat is in all of the Dove Creek, San Miguel Basin, and Monticello, Utah areas, the northern quarter of the Crawford population, and the western half of the Cerro Summit – Cimarron - Sims Mesa population (Fig. 25). The eastern half of the Cerro Summit – Cimarron - Sims Mesa population, a small eastern portion of the Piñon Mesa population, and the northern half and southeast corner of the Gunnison Basin population, are all within areas with low potential for oil and gas drilling. The only areas that have no potential for oil and gas drilling are approximately the southern half of the Gunnison Basin population, the entire Poncha Pass population, and almost all of the Piñon Mesa population. The southern 3/4 of the Crawford population has medium potential for oil and gas development.

Sand, gravel, and other mineral mining activities frequently occur adjacent to existing river and stream channels and in old river channel meanders, which, in GUSG range, may represent brood-rearing habitat. This type of mining may also be located close to towns or areas of impending development, potentially affecting other GUSG seasonal habitats.

Construction of any substantial structure or road, as well as use of access roads can cause increased deposition of dust on plants and invasion of non-native plants, potentially degrading GUSG habitat. Soil erosion, disruption and/or contamination of water sources, and lowering of water tables may also result from oil and gas drilling related activities (Wyoming Game and Fish Department 2003, Bureau of Land Management 2004a), and, although unlikely, could affect sage-grouse habitat. Reclamation efforts may also inadvertently introduce noxious or invasive weeds, altering the sagebrush community.

Disturbance to and Collision Mortality of Grouse

It has been proposed that disruption of normal grouse behavior and productivity can result from various disturbances associated with oil and gas development (Braun et al. 2002, Lyon and Anderson 2003) and powerlines (Robel, in press).

Aspects of oil and gas activities that could disturb prairie grouse include exploratory shot wells, Vibroseis trucks, other exploration vehicles, drill rigs, construction vehicles, venting, flaring, compressor station noise, and human presence. GUSG were observed flushing from a lek when a compressor station switched on (J. Garner, CDOW, personal communication). Alternatively, Attwater's prairie-chickens and lesser prairie-chickens may use well pads or other physically disturbed areas related to oil and gas activities for lek sites (Lutz 1979, Hagen et al. 2004), although further disturbance to the leks and other seasonal habitats may occur (Hagen et al. 2004). In a secondary effect, roads created for oil and gas exploration and drilling could lead to increased recreational use of an area and associated human disturbances (Massey 2001, Wyoming Game and Fish Department 2003). Sand, gravel, and other mineral extraction could cause disturbance to grouse during extraction, sorting and crushing operations.

Grouse mortality may be caused by collisions with wind turbines, communication towers (and associated guy wires), fences, and structures in various utility corridors. The USFWS has proposed a set of guidelines to minimize the danger of collision with wind turbines (Manville 2004). GRSG in Utah have been observed flying into telephone lines (Borell 1939), and Ligon (1951) reported that lesser prairie-chickens were killed by power lines and telephone wires in New Mexico. GUSG collisions with powerlines have also been reported in Colorado (J. Stiver, University of Nebraska, personal communication). Braun (1998) suggested that collision with fences (especially woven wire fences) was a potential factor in sage-grouse decline. Connelly et al. (2000) noted that grouse have been observed hitting or narrowly missing fences or grouse remains have been found next to fences. The impact of collisions on populations of grouse has not been investigated.

Increased Predation Pressure

Elevated structures of various types may provide perch sites for raptors that can prey on grouse. Ellis (1984) described an instance of a golden eagle that changed the strutting behavior of GRSG on a lek in Utah. Ellis (1987) attributed changes in sage-grouse movements on a lek and a shift in lek location in northeastern Utah to construction of a 345-KV transmission line within 660 ft of the lek. Braun et al. (2002) reported that a sage-grouse population in Wyoming inhabiting 40 lek areas within 0.25 miles of power lines had a significantly lower growth rate than a sage-grouse population using lek areas farther away. Increased avian predation was the suggested cause of the lower growth rate. A recent study in Nevada regarding the effects of new transmission lines on sage-grouse in relation to avian predators, concludes in a preliminary progress report (Collopy and Lammers 2004) that the numbers of avian predators documented during surveys did not change significantly after construction of the new powerline. They also documented that the perch deterrents used on the new, as well as the existing powerlines, did not prevent perching, but there was evidence that they did reduce the amount of time spent perching for all species. Apart from actual predation impacts, if grouse perceive a greater threat of harassment and/or predation, they might avoid areas with overhead structures.

Spread of West Nile Virus

Oil and gas drilling may produce water containment pits that could conceivably become mosquito breeding habitat in areas where water was not previously present. Sand and gravel pits are often left as open water ponds after completion of mining, also potentially enhancing mosquito habitat. Any increase in the distribution and numbers of mosquitoes could pose a risk to GUSG because these insects spread WNV (see "Disease and Parasites", pg. 103).

Fig. 25. Potential oil and gas reserves in current GUSG range. Data sources include Bureau of Land Management (1999) for the potential oil and gas resource and lease data and Colorado Oil and Gas Conservation Commission (2005) and the Utah Division of Oil (2004), Gas and Mining for the well location data.

Noxious and Invasive Weeds

A noxious weed is legally defined in Colorado as being non-native and having 1 or more of the following characteristics: (1) aggressively invades or is detrimental to economic crops or native plant communities; (2) is poisonous to livestock; (3) carries detrimental insects, diseases, or parasites; or (4) the presence of the plant is detrimental to environmentally sound management of natural or agricultural ecosystems (Code Title 35 (Agriculture), Article 5.5 (Noxious Weed Act), 103 (Definitions); Colorado Department of Agriculture 2003). There are also plant species, both native and exotic, that are not designated as noxious but are aggressive in growth habit and are considered invasive.

Noxious and invasive weeds have been identified as an important issue in the Crawford, Gunnison Basin, and Piñon Mesa GUSG populations. Stakeholders are concerned with cheatgrass invasions, and the Crawford area local work group has also mentioned knapweed and thistle. Cheatgrass in the Gunnison Basin is receiving research attention from CSU and WSC, as well as treatment with herbicides (Colorado Division of Wildlife 2003).

Cheatgrass and several species of knapweed and thistle are on the Colorado Noxious Weed List (Colorado Department of Agriculture 2003). All of the Colorado GUSG counties have county noxious weed programs, most of which identify knapweed and thistle species, but not cheatgrass, as noxious weeds listed for county control purposes. Noxious and invasive weeds are not known to directly threaten the physical health of GUSG. However, the invasive characteristics of these weedy plants could cause a decline in quality and/or quantity of sage-grouse habitat, thus affecting population parameters.

Cheatgrass is a species that thrives in disturbed, and especially burned, areas (Vallentine 1989, Whisenant 1990). It can even increase fire frequency (Whisenant 1990, Billings 1994, Miller and Eddleman 2000), favoring itself and potentially inhibiting perennial seedling establishment (Wright and Bailey 1982, Whisenant 1990, Grahame and Sisk 2002). A cheatgrass invasion into sagebrush habitat can lead to an eventual conversion of sagebrush/grass (perennial) community to sagebrush/grass (annual) or annual grass rangeland (Connelly et al. 2000, Miller and Eddleman 2000). Sage-grouse food sources vary through the year and include primarily sagebrush, forbs, and insects, but not grasses (Schroeder et al. 1999). In some cases, cheatgrass invasion encourages other exotic species such as knapweed and thistle (Grahame and Sisk 2002)

Pesticides

Insecticides

The pesticides used to control insects (insecticides) are those most likely to affect sage-grouse. Insects are generally a minor diet item for adult sage-grouse but the importance to chicks has been well documented (Patterson 1952, Klebenow and Gray 1968, Johnson and Boyce 1990, Fischer et al. 1996a). Insects, especially ants (Hymenoptera) and beetles (Coleoptera), can comprise a major proportion of the diet of juvenile sage-grouse (Patterson 1952) and are important components of early brood-rearing habitats (Drut et al. 1994a). Fischer et al. (1996a) found that insect abundance was greater at brood-rearing areas than at non-brood sites. Johnson and Boyce (1990) reported that survival and growth rates of sage-grouse chicks were proportional to the amount of insect material in the diet. Early brood-rearing habitats are generally close to nesting habitat and are often relatively open areas with abundant herbaceous cover (Sveum et al. 1998a). These areas may include farmlands and irrigated croplands adjacent to sagebrush habitats.

Impacts of insecticide spraying to sage-grouse may be direct or indirect and are dependent on type of insecticide used, timing of insecticide spraying, and site-specific factors affecting use by sage-grouse, such as crop types and proximity to sagebrush cover. Direct (acute) toxicity of insecticides to sage-grouse occurs through consumption of animal or plant materials with sufficiently high amounts of residue to kill them, dermal absorption, or vapor inhalation through the mucosa of the respiratory tract (Smith 1987). Indirect (subacute) impacts are the disruption of neuronal and endocrinological systems affecting immune function, development and behavior. Another important indirect impact is the reduction of an important food supply for chicks.

Insecticides are used primarily to control insects causing damage to cultivated crops on private lands. The application of insecticides, primarily to control grasshoppers and Mormon crickets, has also occurred on public lands. Infestations of Russian wheat aphids have occurred in GUSG occupied range in Colorado and Utah. Disulfoton, a systemic organophosphate that is extremely toxic to wildlife, was routinely applied to over a million acres of winter wheat crops to control the aphids during the late 1980's. More recently, an infestation of army cutworms occurred in sage-grouse habitat along the Utah-Colorado state line. Thousands of acres of winter wheat and alfalfa fields were sprayed with insecticides such as permethrin by private landowners to control army cutworms. There has been 1 reported instance of sage-grouse mortality following application of organophosphate and carbamate pesticides to cultivated crops in Idaho (Blus et al. 1989).

The arrival of WNV in GUSG range presents an additional potential problem with insecticides. Infection with WNV could threaten GUSG populations, but use of insecticides to control mosquitoes which transmit the virus could have detrimental effects on sage-grouse. Use of larvicides such as Bti (*Bacillus thuringiensis israelensis*), which have extremely low toxicities to vertebrates, can greatly mitigate risks (Rose 2004). Available adulticides include synthetic pyrethroids such as permethrin, which are applied at very low concentrations and have very low vertebrate toxicity (Rose 2004). Organophosphates such as malathion have been used at very low rates to kill adult mosquitoes in and near urban areas for decades, and are judged relatively safe for vertebrates (Rose 2004).

Herbicides

Historically, different combinations of herbicides (pesticides applied to plants) and seasons of applications were developed to remove sagebrush, other unwanted woody shrubs, and weedy annuals from western rangelands (Tueller and Evans 1969, Evans and Young 1975, Evans and Young 1977). The use of herbicides has the potential to directly and indirectly impact GUSG. The impacts can be through direct contact (Ward et al. 1942, Post 1951, Blus et al. 1989) or indirectly through modification of components of the habitat. These modifications can include the removal of sagebrush (Carr and Glover 1970, Klebenow 1970) and the reduction of forbs or insects (Eng 1952).

Herbicide applications of 2,4-D (2,4-dichlorophenoxy acetic acid) or tebuthiuron (N-[5-(1,1-dimethylethyl)-1,3,4-thiadiazol-2-yl]-N,N'-dimethylurea) were commonly used to kill large expanses of sagebrush, leaving the standing dead skeletons of the shrubs with low risk of soil erosion. However, herbicides, if used full strength during the growing season, have killed or injured many forbs (Crawford et al. 2004). More recently, thinning of sagebrush density by Tebuthrion, rather than sagebrush removal from large areas, has been the focus of some treatments (Emmerich 1985, Olson and Whitson 2002).

Predation

Sage-grouse and Predators

Predation is frequently cited as a major cause of mortality in sage-grouse (Bergerud 1988, Schroeder et al. 1999, Connelly et al. 2000). Predation rates vary seasonally. The period of highest mortality for yearling and adult males occurs during the lekking (breeding) season, for yearling and adult females during nesting and brood-rearing, and for juveniles during the first few weeks after hatch (Patterson 1952, Schroeder et al. 1999, Schroeder and Baydack 2001). However, the effect of predation on the fluctuations and viability of sage-grouse populations has never been investigated (Connelly and Braun 1997, Connelly et al 2000, Schroeder and Baydack 2001). Schroeder and Baydack (2001) suggest that nest predators may have an important impact on sage-grouse population dynamics given the high variation in nest success. Nest predation may be higher, more variable, and have a greater impact on small, fragmented populations. The population viability analysis of GUSG presented in this plan (see pg. 168) suggests that juvenile survival may be a limiting factor for population growth. Predation may be an important factor in juvenile mortality, but nutrition, habitat quality, and environmental conditions also affect juvenile mortality (Pyle and Crawford 1996, Sveum et al. 1998a).

Sage-grouse have evolved with native predators, and consequently have developed traits to survive with high predation pressures. For example, both yearling and adult females attempt to nest, lay moderately large clutches, and attempt to renest if nests are destroyed by predators (Svedarsky 1988, Schroeder 1997). Grouse have also adapted anti-predator behaviors such as crouching low or seeking cover under vegetation in the presence of predators, or flying in the opposite direction of attack from avian predators (Hartzler 1974, Ellis 1984, Schroeder et al. 1999). Females perform displays (e.g., erratic movements or dragging their wings on the ground) to distract predators from nests (Schroeder et al. 1999). GRSG females have also been documented defending their nests from ground squirrels (Schroeder 1997), and Girard (1937) observed females attacking predators in the defense of their broods.

Predator Community and Interactions

If there is an effect of predation on sage-grouse populations, it will depend on the species composition of the predator community. There is no published information on the species of predators of GUSG, but predators of GRSG have been well documented (Schroeder et al. 1999). Predators that depredate juvenile and adult GRSG include avian predators such as golden eagles, red-tailed hawks, ferruginous hawks, Swainson's hawks, northern harriers, gyrfalcons, northern goshawks, Cooper's hawks, American kestrels, merlins, and great-horned owls; and mammalian predators such as coyotes, red foxes, weasels, and bobcats. Predators that mainly depredate eggs include avian predators such as common ravens, American crows, and black-billed magpies; and mammalian predators such as badgers, ground squirrels, raccoons, and striped skunks.

The composition and density of predator communities can vary greatly across space and time (Greenwood 1986, Johnson et al. 1989, Sargeant et al. 1993, Sovada et al. 1995). The effect of predation on the demographic structure and population fluctuations of GUSG is unknown, but will likely depend on the composition of the predator community. Avian predators, primarily corvids, were major predators of GRSG nests in Idaho (Autenrieth 1981) and Washington (Vander Haegen 2002), while ground squirrels and badgers were major nest predators in Colorado (Gill 1965) and Wyoming (Patterson 1952). Giesen (1995) documented very poor nesting success in North Park, Colorado, in 1993 and 1994 (22% of 42 nests and 27% of 20 nests, respectively). Most nest loss (87%) was due to depredation, primarily by Richardson's ground squirrels. It is possible that most mammalian predation will be on eggs. Only coyotes and red foxes are likely to prey on all grouse life stages. Most raptor predation will be on juveniles and older age classes, while other avian predators (e.g., common ravens, American crows and black-billed magpies) will mainly affect clutches.

Increasing residential development has been identified in most GUSG local conservation plans as a risk to GUSG. Development not only contributes to the loss of sage-grouse habitat, but it also increases the likelihood that non-native predators (e.g., feral cats and dogs) will be introduced into local GUSG populations. Development can also contribute to increased populations of predators (e.g., red foxes, American crows, and common ravens) that are frequently associated with altered landscapes that provide additional denning or nesting sites, or additional food resources from agricultural waste grain or landfills.

There are other complex ecological consequences associated with predation that must also be addressed before specific management strategies can be recommended. This includes the behavioral and spatial interactions of predators with GUSG and with other predator species. Removing predators from a specific area can lead to a functional and/or numerical response by other predators. Predators compensate for predator removal by either moving into vacated areas (functional response) (Sargeant 1972, Gese et al. 1989) or by producing larger litters that typically have higher survival rates (numerical response) (Knowlton 1972). The reproductive and movement characteristics of predators such as red foxes (Allen 1983), raccoons (Fritzell 1978) and striped skunks (Greenwood and Sargeant 1994) make it possible for these species to respond quickly to predator removal programs.

Furthermore, it has been argued that removing dominant predators from an ecosystem can result in increased populations of lower trophic-level predators - i.e., "mesopredators" such as red foxes, raccoons, ground squirrels and feral pets (Soulé et al. 1988, Rogers and Caro 1998, Crooks and Soulé 1999). The increased population densities of lower trophic-level predators may compensate for the removal of dominant predators such that overall predation rates are not affected (Parker 1984, Greenwood 1986). Predator control programs that focus on removing coyotes can lead to increased populations of red foxes (Sargeant et al. 1987, Voigt and Earle 1983). Red foxes may have a more profound effect on sage-grouse populations than coyotes. In prairie ecosystems, red foxes are a major predator of grassland birds (Sargeant et al. 1984, Greenwood et al. 1987, Johnson et al. 1989) and have a greater impact on nest success of grassland birds than do coyotes (Johnson et al. 1989). Both coyotes and red foxes are territorial and red foxes avoid areas with coyotes (Voigt and Earle 1983). Areas with high densities of coyotes have low densities of red foxes and higher overall nest success (Sovada et al. 1995). Therefore, behavioral and spatial interactions between predator species are complex, and compensatory predation may undermine predator control programs that focus on a single predator species. Attempts at controlling multiple mammalian predators may lead to increased predation rates by avian predators.

Predator Control - Background

Numerous predator control studies in prairie ecosystems have had variable success in increasing waterfowl nest success or productivity (Greenwood 1986, Sargeant et al. 1995). The variability may be partly due to restrictions on the methods allowed (Sargeant et al. 1995), but may also be due to compensatory predation from predator species not included in the control program, or by a numerical and/or functional response by predators included in the program. Predator removal was most successful in small (< 1,236 acres), intensively managed waterfowl nesting areas (Balser et al. 1968, Chesness et al. 1968, Duebbert and Lokemoen 1980, Greenwood 1986, Sargeant et al. 1995). However, moderate improvements in nest success and brood production have been documented for predator removal programs that used multiple methods over relatively larger (<64,247 acres) areas (Balser et al. 1968, Schranck 1972, Duebbert and Kantrud 1974, Duebbert and Lokomoen 1980, Garrettson et al. 1996). However, increases in nest success as a result of predator removal programs tend not to last beyond the duration of active predator removal (Chesness et al. 1968, Duebbert and Kantrud 1974) and generally have not resulted in significant recruitment or population growth in prey populations over time (Cote and Sutherland 1997).

Legal restrictions on some predator control techniques (e.g., trapping and poisoning) may influence a predator control program. In Colorado, it is unlawful to kill wildlife by trapping or poison unless a landowner can provide evidence of ongoing damage to livestock or crops and that other methods not prohibited by law have failed (Colorado Constitution, Title 33: Article 6). Even then, trapping is allowed only for a 30-day period each year. Utah allows predator control (Utah Code sections 23-14-18 and 23-14-19) only if predator management plans specifically define the predation problem and prey species, identify the strategies and methods to be used, and establish measurable objectives for the predator control. Some poisons such as sodium monofluoroacetate (1080) have been banned on federal lands since 1972. The ban was put in place due to a lack of evidence that poisons such as 1080 effectively controlled predator populations (particularly coyote populations), and because nontarget animals (e.g., badgers, eagles, livestock and pets) were often unintentional victims.

Predator Control - Methods

A variety of lethal and nonlethal predator control methods have been developed (Lokemoen 1984, U. S. Department of Agriculture 1994). Occasionally, multiple methods are used to increase the effectiveness of predator control programs, but typically methods are designed for specific predator species in localized areas and are limited by budget and personnel constraints (U. S. Department of Agriculture 1994). Most methods focus on controlling mammalian nest predators, but some target avian nest predators.

Lethal predator control methods are the most traditional and controversial of predator management programs (U. S. Department of Agriculture 1994). These methods include both species-specific chemical toxicants (e.g., zinc phosphide for rodents, sodium cyanide for canids, and DRC-1339 for blackbirds) and non-target strychnine or arsenic based toxicants (U. S. Department of Agriculture 1994). They also include methods such as shooting (e.g., aerial gunning of coyotes), kill-traps, catch and kill techniques (i.e., euthanizing predators after capturing them in leg-hold traps, snares or box traps), killing offspring in dens (used mostly for coyotes and red foxes), or destruction of nests/eggs/hatchlings of avian predators (U. S. Department of Agriculture 1994).

Non-lethal predator control strategies can be divided into small-scale (intensive) methods or relatively large-scale (extensive) methods. Small-scale predator control methods are typically designed to repel predators from well-defined important areas (e.g., a small block of dense nesting habitat). One type of small-scale method involves building fences (predator exclosures) around small blocks of nesting habitat (Lokomoen et al. 1982, Lokomoen 1984, Greenwood et al. 1990) or around individual nests (Sargeant et al. 1974). These barriers can be effective, but are often expensive.

Scare tactics are another type of small-scale method that attempt to disrupt predators from their normal hunting behavior and potentially repel them in important areas. Scare tactics can include distress calls (or calls from avian predators that are designed to ward off other avian species, such as common ravens and American crows), strips of flagging attached to fence lines, bright lights (spotlights) or loud noises (e.g., propane exploders, gunfire, pyrotechnics, or ultrasonic devices) that are triggered by a predator, or scarecrows. Scare tactics are relatively inexpensive; however, many predators (particularly canids) are quick to adapt to the tactics. Some tactics such as bright lights and loud noises may be more annoying to people than to predators.

Another small-scale nonlethal predator control strategy involves altering predator behavior through aversion techniques (Nicolaus et al. 1982, 1983, Nicolaus 1987, Conover 1989, 1990). The techniques attempt to train individual predators to either avoid prey items such as eggs or avoid important areas. Chemically treated eggs are placed where they will be commonly encountered by a predator. The method works only if the predator associates the eggs with the chemical's taste, otherwise predators will continue to disturb nests and destroy eggs to determine if they contain the chemicals (Conover 1989). Other aversion techniques include repellents broadcast over an important area (U. S. Department of Agriculture 1994). Repellants are typically nontoxic, aversive chemicals applied to trees or fence posts. Scent stations are also used to repel predators, but are used only for territorial predators such as canids. Chemical repellants are regulated by the Federal Insecticide, Fungicide and Rodencide Act as administered by the Environmental Protection Agency. Aversion techniques have not been demonstrated to be consistently successful and are relatively expensive and labor-intensive (Greenwood and Sovada 1996).

Another nonlethal approach is to inhibit reproduction of predators through sterilization (U. S. Department of Agriculture 1994). It is argued that inhibiting reproduction will reduce predation rates since parents will have fewer offspring to feed and ultimately, the predator population size will decline as a result of lower recruitment. However, any gains from the approach are likely to be offset by compensatory predation from other species and by a functional response by predators (i.e., untreated predators from adjacent areas move into the treated area in response to the decreased population density).

Habitat Management as Predator Control

Habitat management, as a nonlethal approach to predator control, is receiving increasing attention. A variety of habitat related techniques have been suggested for predator control, including: 1) managing the composition and configuration of habitats at landscape scales, 2) small-scale restoration and management of vegetation structure for cover from predation, 3) managing habitats to enhance (or diminish) the presence of alternative prey, and 4) removing den or nesting sites, and perching sites from important habitats.

The quantity of nesting habitat in the landscape has been correlated to the nesting success of grassland birds and has often been linked to the rate of predation (Kirsch, 1974, Greenwood et al. 1987, 1995, Connelly et al. 1991, Andren 1992, Ball 1996). Furthermore, the composition and configuration of habitats in the landscape can influence the movement patterns and ability of predators to find nests of grassland birds (Kuehl and Clark 2002, Phillips et al. 2003, 2004). Large blocks of nesting habitat in landscapes with alternative habitat types, such as pastures that have food resources attractive to predators (Greenwood et al. 1999), decrease the foraging efficiency of mammalian predators in grassland ecosystems (Phillips et al. 2003, 2004). The fragmentation of important habitat is considered an important mechanism in the decline of many avian populations (Wilcove 1985, Johnson and Temple 1986, 1990) and has been correlated to the type and density of the predator community (Robinson et al. 1995, Yahner 1996, Vander Haegen et al. 2002). It is argued that habitat fragmentation increases predation by decreasing the amount of cover habitat for birds while increasing the amount of habitat easily traveled and searched by predators (e.g., edge habitat). Studies have indicated that the rate of predation is highest in small, linear patches of nesting habitat (Chesness et al. 1968, Haensly et al. 1987, Mankin and Warner 1992). Management of sagebrush habitat at the landscape scale may be a cost effective way to reduce the effect of mammalian predation on GUSG.

Habitat with adequate shrub and grass structure may provide sage-grouse and sage-grouse nests some protection from predators (DeLong et al. 1995, Sveum et al. 1998b). It is suggested that dense vegetation structure will prevent predators from detecting nests. Several studies in prairie ecosystems have reported high nest success for grassland birds in areas with dense vegetation (Schranck 1972, Duebbert and Lokemoen 1976, Livezy 1981, Cowardin et al. 1985, Sugden and Beyersbergen 1986, 1987). The success of the approach may depend on patch size as well as the predator community. Mammalian predators that use olfactory cues to search for prey may not be affected as much by vegetation structure as avian predators that rely more on visual cues.

One possible management tool that has been suggested for controlling predators is managing habitat (or supplementing food resources) so that there is greater abundance of alternative prey (or food resources) either in, or adjacent to, areas of important nesting or brood-rearing habitat. The assumption is that predators will alter their behavior and search for prey items (i.e., alternate prey such as rodents and lagomorphs) that are more abundant or require less energy to find and consume than nests or broods. Therefore, predation rates may be greater for grouse if alternate prey are scarce. However, the few studies that have addressed the question have not been conclusive. Nest success of grassland birds has either improved (Angelstam et al. 1984, Crabtree and Wolfe 1988), shown no response (Greenwood et al. 1998), or declined in the presence of alternative prey (Vickery et al. 1992). Conflicting results may be due to complex predator-prey population dynamics such that temporal or spatial population fluctuations of alternative prey may be too erratic for a predictable predator response.

It has been suggested that predator populations (both the species and population abundance) may be controlled by removing den sites, such as abandoned farmsteads, and nesting or perching structures, such as powerlines and fences (Fleskes and Klaas 1991, Herkert 1994, Greenwood et al. 1995, Larivierre et al. 1999). However, there has been no research on the influence of these structures on predator or sage-grouse populations.

Manipulating habitat to influence predator communities may be the most cost-effective long-term predator control method. However, habitat manipulation will take time and it may not be feasible to reverse the trends in habitat loss and fragmentation for some populations (e.g., in areas of residential development). Because some GUSG populations are so small and are embedded in highly fragmented and developed landscapes, intensive predator control should be considered as a short-term management tool where legally feasible. An integrated program that includes both intensive and extensive predator control methods may be the most effective but will likely be costly. Any predator control program must include long-term monitoring of both predator and GUSG populations in order to evaluate the effectiveness and validity of the program.

Conclusions

Before a predator control program is implemented, research is necessary to: (1) evaluate the demographic status of GUSG populations; (2) eliminate other contributing factors to population fluctuations (e.g., drought or disease); (3) address the behavioral and spatial interactions of predators and sage-grouse; (4) identify the extent of predation pressures and contributing predator community; and (5) evaluate the role of predation on the long-term viability of sage-grouse populations.

The development of an effective predator management program is problematic given the complexity of the ecological and legal consequences, lack of reliable information, and public resistance to lethal predator control (Messmer et al. 1999). However, predator control may be necessary under some circumstances for GUSG populations with small numbers of grouse in isolated and fragmented populations. In these cases, a predator control program should be designed for a specific GUSG population, since the relevant predator community will likely vary for each population. An integrated program that includes both intensive and extensive (lethal and nonlethal) predator control methods may be the most effective, but will likely be costly. Predator control may be valid only if nest success and/or female (or brood) survival is exceptionally low. The population viability analysis of GUSG that we present ("Population Viability Analysis", pg. 168) indicates a higher extinction probability for populations with < 25 breeding individuals.

Therefore, predator control measures should be considered if a local GUSG population is, (1) below 25 breeding individuals or 25% of the long-term population goal (especially, if it is a declining or recently augmented population); (2) nest success is < 25%, and/or female (or brood) survival is < 45% (Connelly et al. 2000); and (3) the population is assessed as a high conservation priority (see Table 41, pg. 303). Quantifiable objectives within a specific time-frame must be specified, and long-term monitoring of both predator and prey communities (sage-grouse as well as other prey species), are necessary in order to objectively evaluate the success of the program. All predator management plans in Colorado will follow directives of the Colorado Wildlife Commission Mammalian Predator Management Policy and be submitted to the Wildlife Commission and the Director of the Division of Wildlife for review and approval or rejection. In Utah, the UDWR and the Utah Wildlife Board regulate predator management by establishing rules and policies, and by developing and implementing predator management plans. Through an MOU with UDWR, Wildlife Services, a branch of the Utah Department of Agriculture, is responsible for handling livestock depredation problems.

Recreational Activity

It has been postulated that recreational activity might have a negative effect on GUSG. Although there has been no research into or evidence to support this possibility, a review of the potential recreational effects on wildlife in general is instructive. Recreation on lands managed by the BLM is a significant land use (Connelly et al. 2004) and recreational use of national forests has increased 76% since 1977 (Rosenberg et al. 2004). Human activities such as recreation can impact wildlife through four primary routes: (1) exploitation; (2) disturbance; (3) habitat modification; and (4) pollution (Knight and Gutzwiller 1995). Exploitation refers to immediate death from hunting, trapping, or scientific collection (see "Hunting", pg. 122). Unintentional disturbance to individual animals may result from activities such as birdwatching, wildlife photography, hiking, biking, or motorized use through habitat. Recreationists may inadvertently modify vegetation, soil, water, and even microclimates, which in turn can impact species associated with these affected habitats. Some wildlife species are indirectly affected by pollution, such as human trash, including food and plastic objects.

If recreation does impact GUSG, disturbance is the most likely factor to affect the species. Most studies on wildlife species have documented immediate, rather than long-term responses to disturbance (Knight and Gutzwiller 1995). Some of these potential responses are behavioral changes, including nest abandonment and a change in food habits, as well as physiological changes, such as elevated heart rates.

Wildlife viewing has the potential to negatively affect wildlife. Avid birders sometimes intentionally seek out rare or spectacular species, such as GUSG. Because viewing activities sometimes occur during sensitive times of year (e.g., strutting/nesting), they have the potential to negatively affect wildlife behavior, if not managed properly (see "Lek Viewing", pg. 124). Of five different recreation-user groups at a wildlife refuge in Florida, photographers were the most disruptive, since they were most likely to stop, leave their vehicles and approach wildlife (Klein 1993, as cited in Knight and Gutzwiller 1995).

Dispersed recreational activities, such as off-road vehicle use, backpacking, hiking, cross-country skiing, and horseback riding, have increased dramatically in recent years. These activities are geographically extensive in nature and have the ability to disrupt wildlife in many ways, particularly by displacing animals from an area. Most documented responses have been behavioral and short-lived (Knight and Gutzwiller 1995).

Disturbance during a species' breeding season may affect individual productivity. Wildlife may respond to disturbance during the breeding season by abandoning their nests or young, leading to reproductive failure. Human activity can also alter parental attentiveness, increasing the vulnerability of the young to predators, disrupting feeding patterns, or exposing the young or eggs to adverse environmental stress.

One extension of human recreation in wildlife habitats is the effect on wildlife of domestic dogs that might accompany recreationists. Dogs can cause disturbance, harassment, displacement, and/or direct mortality of wildlife. Authors of many wildlife disturbance studies concluded that dogs with people, dogs on-leash, or loose dogs provoked the most pronounced disturbance reactions from their study animals (Sime 1999). Dogs extend the zone of human influence when off-leash. Potential consequences of dogs off-leash are primarily harassment, due to the predator instinct of dogs to chase/hunt animals. Harassment by dogs can lead to physiological stress, as well as the separation of adult and young, or flushing incubating birds from their nest. Displacement, whether caused by dogs or humans, also has the potential to increase predation by the natural predators, as well, by increasing the vulnerability of adults and young.

Weather/Drought

Colorado and Utah can experience extreme climatic conditions during all seasons. Long periods of below average precipitation, above average summer temperatures, above average snowfall, or below average winter temperatures may have adverse effects on sage-grouse reproductive success and survival. In fact, prolonged drought during the 1930's and in the latter part of the 20th century coincided with declines in grouse populations throughout their range (Patterson 1952, Fischer 1994, Hanf et al. 1994). Extreme climatic conditions that occur during critical life cycle sequences have the potential to adversely affect the abundance and quality of food resources and hiding cover (Hanf et al. 1994, Fischer et al. 1996b).

Severe winter conditions may reduce grouse survival, although evidence to support this is not conclusive (Wallestad 1975, Beck 1977, Robertson 1991). Winter snow accumulation forces birds to move to areas blown free of snow or areas with sagebrush which extends above the snow (Eng and Schladweiler 1972, Wallestad 1975, Beck 1977, Hupp and Braun 1989b, Robertson 1991). A severe winter in 1983-84, with a long period of extreme cold and heavy snow, is believed to have been a factor in decline in GRSG population in northwestern Colorado and GUSG populations in the Gunnison Basin.

Poor weather conditions in the spring may influence sage-grouse production (Connelly et al. 2000). Good winters followed by relatively wet springs can increase grouse production (Wallestad 1975, Autenrieth 1981) by resulting in good insect and forb production. In contrast, severe spring weather (cold temperature combined with rain and wind) that coincides with hatching can decrease production (Wallestad 1975).

GRSG can be very sensitive to fluctuations in annual moisture (Patterson 1952, Fischer 1994, Hanf et al. 1994). Sage-grouse summer diet, especially that of chicks, is heavily dependent on insects and succulent plant growth. GRSG populations decline in years of low precipitation, most likely due to low nest success and/or poor chick survival (Hanf et al. 1994; Fischer et al. 1996b). Quality nesting cover requires sagebrush canopy as well as forb and grass cover to hide hens and their nests. Severe drought conditions, such as those much of the western states experienced in 2002-2004, may have hindered the production of grasses, forbs, and sagebrush.

Some sagebrush communities across the range of the GUSG experienced defoliation, die-off, and loss of understories in 2003 due to lack of water (2002 drought), insects, and possibly pathogens (Wenger et al. 2003). Conversely, some stands of sagebrush that appeared to be in poor condition also experienced high seed production in the fall of 2003, as well as a release of young sagebrush, other shrubs, grasses, and forbs. Consequently, some disturbance to the vegetation community may help to set back succession of sagebrush within otherwise undisturbed communities, thus possibly improving sagebrush and understory quality and quantity.

Threats Summarized by ESA Listing Factor

The ESA listing factors evaluate threats to a species and are used to determine whether a species is threatened or endangered, thereby warranting listing under the ESA. Here we give a general summary of the threats to GUSG, grouped by ESA listing factors. In addition, the potential issues affecting each GUSG population have been identified separately by the USFWS, CDOW, UDWR, The Nature Conservancy, and in GUSG local conservation plans (Table 20). Some of the threats listed in Table 20 are specific local issues that we discuss under a more general topic (e.g., "Urban Development" and "Agricultural Conversion" are discussed in the RCP under "Habitat – Risk of Permanent Loss", pg. 149). For further background on a given topic, see appropriate topics earlier in this section, "Threats and Analysis" (beginning pg. 103).

Listing Factor A: The present or threatened destruction, modification, or curtailment of the species' habitat or range.

Size of GUSG range and quality of GUSG habitat have been reduced by direct habitat loss, fragmentation, and/or degradation from building development, road and utility corridors, fences, energy development, conversion of native habitat to hay or other crop fields, alteration or destruction of wetland and riparian areas, incompatible livestock management, competition for winter range by big game, and creation of large reservoirs.

Listing Factor B: Overutilization of the species for commercial, recreational, scientific, or educational purposes.

GUSG populations have not been overused for commercial, scientific, or educational purposes, but some of the smaller populations may have been impacted by legal and illegal hunting. The Gunnison Basin population had a hunting season through 1999; whether or not hunting impacted the Gunnison Basin population is debatable. None of the other populations has been included in a hunting season for many years, and it is unlikely that any of the populations (including Gunnison Basin) will have a hunting season in the foreseeable future. With increased awareness of the plight of sage-grouse by the public and increased attention by state wildlife law enforcement personnel, it is believed that little illegal hunting currently occurs and may be limited to incidental shootings.

Because the GUSG is a newly designated species, many bird-watchers wish to add GUSG to their "life lists". Increased numbers of observers could cause disturbance to sage-grouse at commonly known or newly discovered lek sites. Concern over disturbance by birdwatching has been specifically mentioned for the Gunnison Basin and Crawford Area populations.

Current research efforts that include trapping, banding, and radio-marking GUSG are not believed to adversely affect populations, although 1 local plan expressed concern over potential impacts from increased research levels.

Listing Factor C: Disease or predation affecting the species.

No disease problems have been detected in GUSG, but the recent appearance of West Nile virus and its known impact on some bird species could be a threat to GUSG, especially in the smaller populations, in the near future. It is also possible that other game birds, such as turkeys, pheasants, and chukars could transmit diseases to sage-grouse.

Predation on sage-grouse by many mammalian and avian predators has been observed (Schroeder et al. 1999). Most loss of potential productivity is through nest failure, which is often caused by ground or avian predators (Schroeder et al. 1999). Structures such as fences, buildings, and utility poles provide hunting perches for raptors, and if placed near lek sites they might be detrimental to sage-grouse.

Listing Factor D: The inadequacy of existing regulatory mechanisms to protect the species.

For a detailed description of existing management and legal authorities for the protection of GUSG see "Management and Legal Authorities" (pg. 14). The GUSG is a sensitive species in Colorado and Utah. The CDOW and UDWR have authority for setting hunting seasons and possession limits and for enforcement against poaching and harassment. However, the state wildlife agencies do not have authority for protecting against habitat loss. Furthermore, federal land management agencies do not have authority to protect against habitat loss on private land.

There have been many actions taken on private land to conserve GUSG and the willingness of landowners to carry out these actions offers great potential for conservation of the species on private land. The RCP is needed to direct rangewide population goals, transplant/genetic needs, and reestablishment of habitat linkages between populations and subpopulations. Furthermore, participation in federal programs directed towards private land management is voluntary and dependent on program funding. Actions carried out by federal land management agencies on federal lands are also dependent on funding. Wildlife programs of the BLM and USFS have received funding specifically for sage-grouse in recent years, but further habitat improvements are needed, especially on BLM land, which represents 42% of the currently occupied GUSG habitat in Colorado and Utah (Appendix D).

Listing Factor E: Other natural or manmade factors affecting the species' continued existence.

Other factors that may affect continued existence of GUSG include fire suppression (allowing encroachment of sagebrush habitat by piñon and juniper or old and even-aged stands of sagebrush), overgrazing by elk, deer and domestic livestock, drought, disturbance or mortality caused by off-highway-vehicles, disturbance by construction projects or oil and gas development, harassment from people and pets, continuous noise that impairs acoustical quality of leks, inbreeding depression, herbicides, insecticides, pollution, and competition for habitat from other species.

Table 20. Current and potential issues affecting GUSG populations. Issues have been identified in the following documents: C = Local Conservation Plan; D = Colorado Division of Wildlife, Annual Candidate Status Review Summaries (Colorado Division of Wildlife 2002, 2003); F = U. S. Fish and Wildlife Service, 2003 Candidate Review Form; U = Utah Division of Wildlife, Strategic Management Plan for Sage-grouse; N = The Nature Conservancy, (The Nature Conservancy 2002).

USFWS Listing Factor	**Issue Affecting GUSG**	**GUNNISON SAGE-GROUSE POPULATION**							
		Cerro – Cimarron - Sims Mesa	**Crawford**	**Dove Creek**	**Gunnison Basin**	**Monticello Utah**	**Piñon Mesa**	**Poncha Pass**	**San Miguel Basin**
Factor A: Present or Threatened Destruction, Modification, or Curtailment of its Habitat or Range	**Urban Development**	**D, F, N**	**C, D, F, N**	**C, D, F, N**	**C, D, F, N**	**C, F**	**C, D, F, N**	**C, D, F**	**C, D, F, N**
	Roads	**D, F, N**	**C, D, F**	**C, D, F**	**C, D, F**	**C, F**	**C, D, F, N**	**C, D, F**	**C, D, F, N**
	Utility Corridors	**D, F, N**	**C, D, F**	**C, D, F**	**C, D, F**	**C, F**	**C, D, F, N**	**C, D, F**	**C, D, F, N**
	Fences	**D, F**	**C, D, F**	**C, D, F**	**C, D, F**	**C, F**	**C, D, F**	**C, D, F**	**C, D, F**
	Oil and Gas Wells					**U**			**D, F, N**
	Oil and Gas Pipelines	**N**	**C**	**C, F**	**C**	**C, U**	**C, N**	**C**	**C, F, N**
	Mining				**C, F**			**C, D**	**C, F**
	Agricultural Conversion	**D, F**	**C, F**	**C, D, F, N**	**C, F**	**C, F, U**	**C, D, F**	**C, F**	**C, D, F**
	Wetland/Riparian Alteration	**D, F**	**C, F**	**C, F**	**C, F, N**	**C, F**	**F**	**F**	**C, F**
	Livestock Management	**D, F, N**	**C, F, N**	**C, F, N**	**C, F, N**	**C, F, U**	**C, F, N**	**C, F**	**C, F, N**
	Big Game Grazing/Browsing	**F, N**	**C, F**	**C, F**	**C, F**	**F**	**C, F**	**C, F**	**C, F, N**
	Reservoirs		**C**		**C, F**				**C, F**
	Poor Habitat Quality/Quantity	**D, F, N**	**C, D, F, N**	**C, D, F, N**	**C, D, F, N**	**C, F, U**	**C, D, F, N**	**C, F**	**C, D, F, N**
	Habitat Fragmentation	**D**	**C**	**C, D**	**C**	**C**	**C**	**C**	**C, D**

USFWS Listing Factor	**Issue Affecting GUSG**	**GUNNISON SAGE-GROUSE POPULATION**							
		Cerro – Cimarron - Sims Mesa	**Crawford**	**Dove Creek**	**Gunnison Basin**	**Monticello Utah**	**Piñon Mesa**	**Poncha Pass**	**San Miguel Basin**
Factor B: Overutilization for Recreational, Commercial, Scientific, or Educational Purposes	**Lek Viewing**		**D, F**		**C, D, F, N**	**U**		**C**	
	Hunting		**C**		**C, F**		**C**	**C, F**	
	Poaching			**C**	**C**	**U**			**C**
	Recreation	**D, F, N**	**C**		**C, F, N**	**C**	**C**	**C**	**C**
	Lek Counts/Monitoring		**C**	**C**	**C**		**C**	**C**	**C, F**
	Sage-grouse Research	**F**	**C, F**	**C, F**	**C**	**F**	**C, F**	**C, F**	**D**
	Vegetation Experiments								

Table 20 (con't). Current and potential issues affecting GUSG populations. Issues have been identified in the following documents: C = Local Conservation Plan; D = Colorado Division of Wildlife, Annual Candidate Status Review Summaries (Colorado Division of Wildlife 2002, 2003); F = U. S. Fish and Wildlife Service, 2003 Candidate Review Form; U = Utah Division of Wildlife, Strategic Management Plan for Sage-grouse; N = The Nature Conservancy, (The Nature Conservancy 2002).

USFWS Listing Factor	Issue Affecting GUSG	GUNNISON SAGE-GROUSE POPULATION							
		Cerro - Cimarron - Sims Mesa	Crawford	Dove Creek	Gunnison Basin	Monticello Utah	Piñon Mesa	Poncha Pass	San Miguel Basin
C. Disease or Predation.	Disease	D, F	D, F	C, F	C, F, N	C, F	C, F	C, F	C, D, F
	Predation	D, F	D, F, N	F	D, F	F	F	C, F	D, F

USFWS Listing Factor	Issue Affecting GUSG	GUNNISON SAGE-GROUSE POPULATION							
		Cerro – Cimarron - Sims Mesa	Crawford	Dove Creek	Gunnison Basin	Monticello Utah	Piñon Mesa	Poncha Pass	San Miguel Basin
Factor D: Inadequate Regulatory Mechanisms.	Conflicting Land Zoning	D, F	F	F	C, F	F	F	F	F
	Conflicting Regulations and Policies*		C	C	C				C, F
	Inadequate Conservation Funding	F	F	F	F	F	F	F	F
	Inadequate Habitat Protection Regulations	F	F	F	F	F	F	F	F
	Inadequate Regulations for Native/Exotic Releases					U			

Table 20 (con't). Current and potential issues affecting GUSG populations. Issues have been identified in the following documents: C = Local Conservation Plan; D = Colorado Division of Wildlife, Annual Candidate Status Review Summaries (Colorado Division of Wildlife 2002, 2003); F = U. S. Fish and Wildlife Service, 2003 Candidate Review Form; U = Utah Division of Wildlife, Strategic Management Plan for Sage-grouse; N = The Nature Conservancy, (The Nature Conservancy 2002).

USFWS Listing Factor	Issue Affecting GUSG	GUNNISON SAGE-GROUSE POPULATION							
		Cerro – Cimarron - Sims Mesa	Crawford	Dove Creek	Gunnison Basin	Monticello Utah	Piñon Mesa	Poncha Pass	San Miguel Basin
Factor E. Other Natural or Manmade Factors Affecting Species' Existence	Piñon-Juniper Encroachment	F	C, D, F	C, F	F	F, U	C, D, F	F	D, F
	Oakbrush Encroachment	D, F		C, F			C, F		D, F
	Fire Suppression	D	C, N	C	C, N		C, D, N		C
	Sagebrush Community Changes	D	C	C	C	C	C, D	D	C, D
	Exotic Weed Invasion	D, N	D		C, D, F	C	D, F, N		D F, N
	Drought Impacts to Sagebrush Habitat	D, F	C, D, F	C, F	C, D, F	C, F	C, D, F	C, F	C, D, F
	Motorized Vehicles	N	C	C	C, F, N	C	C	C	C
	Disturbance from Construction	F	F	F	C, F	F	F	F	F
	Harassment from People/Pets	D, F, N	C, F	C, F	C, F, N	F	C, F	C, F	C, F
	Noise Impacts to Leks	F	C, F	C, F	C, F	F	C, F	C, F	C, F
	Geographic Isolation	D, F	F	F		F, U	F	C, F	F
	Herbicide Use	D, F	C, F	F	C, F	C, F	F	F	C, F
	Insecticide Use	F	F	F	F	C, F, U	F	F	F
	Pollution	F	F	F	C, F	F	F	F	F
	Competition from Other Species	D, F	C, F	C, F	C, F	C, F	C, F	F	C, F
	Inappropriate Vegetation Treatments	F, N	C, F	C, F, N	C, F, N	F	C, F, N	C, F	C, F, N
	Soil Erosion		C	C	C			C	C, D

B. Habitat – Risk of Permanent Loss

Problem Definition

There is no other issue more fundamental to the long-term preservation of GUSG than protection of the sagebrush and other habitats on which they depend. The decline in distribution and abundance of GUSG is due largely to conversion of native habitats to crop production, reservoirs, or developments. Oyler-McCance et al. (2001) compared low-level aerial photographs from the 1950's and 1990's and concluded 20% of sagebrush-dominated areas in southwestern Colorado had been lost in that time frame. They noted much of the sagebrush habitat outside the Gunnison Basin had been converted to other uses before their earliest photos, and hence was not captured in the analysis. They also found 37% of their study plots had experienced substantial fragmentation of sagebrush, and concluded if current trends in habitat loss continued, GUSG could become extinct. Type conversion or development results directly in loss of habitat, degradation of remaining habitat from fragmentation, as well as indirect impacts from associated factors (e.g., roads, fencing, powerlines, increased human activity), and may facilitate introduction of novel predators and noxious weeds.

Riebsame et al. (1996) described a changing pattern in residential development in Colorado which began in the 1970's and continues today: a significant amount of home building now occurs in subdivisions and large lots far from existing townsites. Exurban development for primary population growth and for second homes has been a significant cause of loss of sagebrush habitats. Theobald et al. (1996) described trends in exurban development in the East River Valley in Gunnison County from 1964 to 1994. They documented an increase in total road length from 114 to 183 miles over this period, and an increase in building numbers (216 to 552). Nearly 90% of the buildings were located in low-elevation meadow, grassland, and sagebrush cover types.

Regulatory and Other Relief

Protection of habitats from permanent loss on publicly owned and managed lands seems straightforward, but protection of habitat quality from other land uses such as grazing, energy development, and recreation may be needed. Protection of habitat from permanent loss on private land is much more problematic. Authority for regulating land use on non-federal lands was delegated to the 63 counties in Colorado in 1974. All units of local governments including counties, cities, and towns were given authority to regulate land use within their jurisdictions (C.R.S. 29-20-101).

In Colorado, the CDOW is required by statute (C.R.S. 106-7-104) to provide counties with information on "significant wildlife habitat", and provide technical assistance in establishing guidelines for designating and administering such areas, if asked. Counties may, but are not required to, protect land from activities that would cause immediate or foreseeable material danger to significant wildlife habitat, or endanger a wildlife species.

Normally conversion of land zoned as agricultural from one agricultural use, such as native pasture containing sagebrush to another use, such as cropland, would not come before a county zoning commission, so typically habitat loss of that nature is not regulated. State statute exempts parcels of land of 35 acres or more per home from regulation, so county zoning laws can only restrict developments with housing densities greater than 1 per 35 acres (C.R.S. 30-28-101).

A recent change in Gunnison County's land use code exempts from regulation as subdivisions parcels with 2 houses per 35 acres or larger. This right to build a house or cabin (or 2 in Gunnison County) on parcels of 35 acres or larger means regulatory measures alone will never guarantee protection of important habitat from development. Where such development is a likely threat, other protections such as easements or fee-title acquisition of important habitats will be necessary. Maintaining sustainable rural economies (where traditional land uses compatible with sage-grouse are profitable) can significantly reduce threats associated with subdivisions.

Private property owners have a right to develop their land. Those that develop early may gain little reward compared to those who wait for land values to inflate. Ironically, those who are most reluctant to develop may suffer the most harm if a species becomes listed and the regulatory provisions of the ESA are enforced. Long-term and community-based planning to direct growth and development to appropriate areas, along with compensations for restrictions on developments in important areas are the most efficient way to accomplish conservation.

Risk Assessment of Habitat Loss Among Populations

We employed a variety of methods to assess the relative extent of risk of permanent habitat loss to aid in identifying the need for, and to aid in prioritizing, conservation measures among populations. As discussed above, major factors causing permanent habitat loss within GUSG range include human population increases (and resultant housing developments and associated infrastructure such as roads, fences, and powerlines), conversion to agriculture, and development of energy resources such as oil and gas extraction of mining. We used U.S. Census Bureau data to examine projected human population increases. We also identified acres enrolled in the CRP as potentially at risk. Sage-grouse in some populations use these fields seasonally to some extent, but if fields are not re-enrolled, or the program does not continue, this would very likely represent a permanent habitat loss to these populations. Risk of habitat loss from extraction of oil and gas resources was evaluated using BLM relative rankings of potential oil and gas reserves (none, low, medium, or high potential; Fig. 25, pg. 130).

The United States Census Bureau projected population growth between 2000 and 2020 for each county in the United States, accessible through the Colorado Department of Local Affairs website (Colorado Department of Local Affairs 2004). They also projected the increase in housing units that would be expected from this population increase based on a 10-year average of residents per housing unit. These increases are shown in Table 21 for each population of GUSG, based on the data for the county in which the population resides. It should be noted that county-wide projections may only serve as a crude index to permanent habitat loss for GUSG, since growth may be concentrated in urban areas away from currently occupied habitat. The current density of people is also provided, to scale the threat; i.e., a 50% increase in population may be more significant from a baseline of 50 people/mi^2 (rising to 75) than it is for a population of 2 people/mi^2 (rising to 3).

Table 21. Summary of threats that may cause permanent habitat loss, by population.

Population	Population Growth[1]	Growth in Housing Units[2]	People/ mi^2 (County)	Agricultural Conversion[3]	Energy Development Potential[4]	Acres in CRP[5]	Known Development Threats
Cerro Summit – Cimarron - Sims Mesa	68%	68%	25	-17%	Low - High	0	-
Crawford	51%	58%	24	-3%	Medium High	0	Elk Ranch subdivision
Dove Creek	32%	30%	2	-2%	High	26,485	2,700 acres (Secret Canyon subdivision – 9%)
Monticello, Utah	30%	54%	2	no data	High[6]	36,800	-
Gunnison Basin	25%	30%	5	-22%	None - Low	60	-
Piñon Mesa	40%	56%	55	-6%	None - Low	0	-
Poncha Pass	34%	31%	3	-10%	None	0	Some parcels for sale
San Miguel Basin	71%	62%	9	28%	High	3,358	-

[1] Based on Census Bureau projections for county population resides in, 2000-2020.

[2] Calculated by dividing population projections by the 10-year average of residents per housing unit.

[3] Indexed as percent change in total acres in cropland from 1987-1997.

[4] Based on BLM rankings of potential oil and gas reserves (see Fig. 25, pg. 130).

[5] Grassland habitats enrolled in the Conservation Reserve Program for 10-year periods, which are subject to conversion back to cropland.

[6] Monticello subpopulation received a "High" rating for potential energy development based on its proximity to the Dove Creek subpopulation, and because of the existence of active and inactive wells in or near the currently occupied population area (Fig. 25, pg. 130).

Montrose County was identified as one of the fastest growing counties in the country, with human population and associated housing units expected to increase 68% from 2000 to 2020 (Table 21). Although the greatest current density, and presumably future development, is in and near Montrose, growth is likely to impact private property currently used by GUSG in Cerro Summit - Cimarron - Sims Mesa, and potential linkages connecting the San Miguel population to Crawford (Table 21, and Fig. 29 on pg. 167).

Nearby Delta County (Crawford population) was also projected to increase greatly in population and housing density, with increases of 51 and 58%, respectively (Table 21). This may impact the relatively small amount of private land not already protected by easement in the currently occupied habitat within the Crawford population (~15% of currently occupied habitat), although most of this growth is likely to occur in and near the towns of Delta and Crawford.

The Dove Creek sub-population is in Dolores County, which had a low density of only 2 people/mi^2 in 2000 and is expected to grow by 32% by 2020 (Table 21). Presumably much of this growth will occur in Dove Creek and thus away from GUSG habitat, but "full build-out" of the Secret Canyon subdivision will result in significant permanent loss of some of the best habitat available to this population. Dolores County has over 26,000 acres enrolled in the CRP program, some of which is used seasonally by sage-grouse. San Juan County (Utah) has similar growth projections, although it has no platted subdivisions. San Juan County has 36,800 acres enrolled in CRP, of which about half is within occupied sage-grouse habitat.

Gunnison County had a relatively low population density (5 people/mi^2), with population and housing increases of 25 and 30%, respectively, projected to 2020. Presumably much of this increase will occur in the towns of Gunnison and Crested Butte, as opposed to sagebrush habitats used by sage-grouse.

Mesa County, which contains the Piñon Mesa population, had a relatively high population density of 55 people/mi^2, and high projected increases of 40% in population and 56% in housing units by 2020 (Table 21). Although these current and projected densities are heavily influenced by Grand Junction, the proximity of Piñon Mesa to Grand Junction suggests it could be influenced by this growth.

The Poncha Pass population resides in Saguache County, which had a low population density of 3 people/mi^2 and a projected increase of 34% by 2020, with a similar increase in housing units. This may underestimate risk to sage-grouse, because the scenic aspect of this portion of the San Luis Valley may concentrate development in and near occupied sage-grouse habitat. In addition, northern Saguache County is becoming somewhat of a bedroom community for people working in Salida, due to the high-cost housing in Chaffee County. This may accelerate development of housing on small tracts of land on the south side of Poncha Pass and increase traffic on Highway 285.

The San Miguel population occupies several areas in San Miguel County. Although San Miguel County had 9 people/mi^2 in 2000, most residents live in the town of Telluride or several smaller communities, including Norwood. This county has experienced a 28% increase in cropland between 1987 and 1997, which probably resulted in loss of sagebrush habitats, and which could continue to some degree. The population in San Miguel County is expected to increase markedly by 2020 (71% increase in people and 62% in housing units).

Conversion to Agriculture Uses

Assessing risk of permanent loss of sagebrush habitats from conversion to land uses other than urban or exurban development is difficult since models of this type of loss have not been developed. Fortunately, the available evidence suggests conversion to agricultural uses has largely ceased. Acreage in cropland actually declined in Delta (3% decline), Dolores (2%), Gunnison (22%), Mesa (6%), Montrose (17%), and Saguache (10%) Counties from 1992-1997 (Colorado Agricultural Statistics). Acreage in cropland increased by 28% during this period in San Miguel County, although it is not clear whether this increase was caused by conversion of sagebrush. Acreage in pastureland also decreased in Dolores, Gunnison, Mesa, San Miguel, and Montrose Counties, by 7, 20, 6, 13, and 21%, respectively, from 1992-1997. Pastureland increased in Delta County by 7% and Saguache County by 1% during this period.

Spatially Explicit Analysis of Impacts of Additional Housing Units

We used 2 methods to further assess the risk of additional housing development in GUSG habitat. The intent of this analysis is to identify areas where risk of housing development is important, to aid agencies and work groups in habitat protection efforts. Details of the analysis are provided in Appendix F.

Dr. David Theobald, Natural Resource Ecology Lab, Colorado State University, developed a Spatially Explicit Regional Growth Model (SERGoM v1), designed to depict the location and density of current and projected future private land housing units across the coterminous U.S. Although the current model has not yet been published (Theobald, in review), the general procedure and rationale for a previous version of the model are described in Theobald (2003). Future growth in housing units was based on Census Bureau county-level projections for population growth. The number of housing units this growth was apportioned to was determined using the county-level average of people/household, taken from 2000 census data. Growth in housing units was allocated spatially using a formula that considered recent (1990-2000) housing growth rates for a specific location and accessibility to the nearest urban core. Assumptions of this approach are that: (1) future growth patterns will be similar to those found in the past decade; (2) people/household in the future will match that in the 2000 census data; (3) future growth is likely to occur nearby current high growth areas or "hot spots"; (4) housing units cannot occur on public land, water areas, etc.; (5) growth will be concentrated in areas closer (in terms of travel time, not just distance) to urban core areas over major roads; and (6) housing density will not decline over time (housing growth projections are additive to current housing densities).

We applied Dr. Theobald's model and resultant predicted housing density dataset in a GIS analysis to evaluate the potential acreage impacted by development in 2020 for each population of GUSG (for detailed report, see Appendix F). We are not aware of any published work that indicates what level of housing development impacts or eliminates sage-grouse use of habitat. In this initial analysis we chose 320 acres/housing unit as the threshold below which we expect impacts, and above which we do not. This estimate was used with the following rationale: (1) over 38,500 acres within 1.86 miles of leks in the Gunnison Basin have more than 1 housing unit/320 acres now (2000), yet grouse use has continued; (2) only 4 of 41 active leks have no housing units within 1.86 miles; and (3) 35 of 41 active leks have at least some area with housing densities greater than 1 unit/320 acres. This threshold was chosen keeping in mind the large amount of public (and therefore protected) habitat in the Gunnsion Basin. We do not suggest that if the large block of public land were developed at this density (1 housing unit/320 acres) that grouse would not be impacted.

The modeled housing density in 2000 is shown in Fig. 26, while projected housing densities (without intervention) in 2020 are shown in Fig. 27 (note that white areas are the protected lands; i.e., public). Areas of growth in housing are identified in Fig. 28. Numerical estimates of acreage in each housing class modeled for 2000, projected to 2020, and increases from 2000 to 2020 by housing density class are shown for the smaller populations (Table 1, Appendix F) and for the Gunnison Basin (Table 2, Appendix F). The challenge in wisely allocating habitat protection dollars is to protect important areas where development will occur at a density that precludes use by sage-grouse, or will significantly impact grouse. At the same time there is little point in allocating resources to areas already impacted so as to preclude grouse use, or to areas where housing densities will be so low as to have negligible impact to grouse. Consequently, we identified areas and acreages projected to increase from housing densities of 1 unit per 320 acres or larger to 1 unit per 320 acres or less. Our results indicate, for the most part, that housing outside of urban areas progresses through housing density classes, therefore the key areas are those that move from 1 unit per 320 acres or more to 1 unit per 160-320 acres, although occasionally densities may jump to the 80-160 acre/housing unit class.

The model predicting development to unsuitable housing densities seemed to perform poorly (underestimated development) outside the Gunnison Basin, where second home development or proximity to population centers or high growth areas such as Grand Junction, Montrose, or Telluride may trump local demographic growth as causes of development. In some cases, the model suggested little or no future development in areas already platted with lots marketed for sale. Clearly, we have a long-term need to develop better predictive models which take these factors into account (see "Habitat Protection from Permanent Loss" rangewide strategy, pg. 223, Objective 1, Strategy 9). In the interim, we used another approach to identify habitats at greatest risk of development in the next 3-5 years. Typically, land is subdivided into smaller parcels prior to sale and development. It is these smaller (<80 acres) parcels that are probably most immediately susceptible to development to densities that would negatively impact grouse. Larger parcels may be subdivided, but this process will occur over a longer time horizon, allowing time to respond. We mapped private land parcels by parcel size categories for each population (excluding Gunnison Basin; see figures in Appendix F) as a tool to help agencies, work groups, and land trusts in assessing development risk and prioritizing habitat protection efforts for GUSG (see Appendix F). We present an analysis of future development by population using both methods of assessing risk.

Fig. 26 Modeled housing densities for unprotected lands, 2000.

Fig. 27 Projected housing densities for unprotected lands, 2020.

Fig. 28. Areas of growth from modeled year 2000 to 2020, for areas less than 320 acres per unit, on unprotected lands.

Prioritization of Habitat Protection Efforts

We incorporated the information from Table 21 and the analyses of risk of permanent habitat loss from housing development (see Appendix F) into an assessment of the relative importance of each population to the overall conservation of GUSG. We also examined the relative amount of acreage these threats applied to (private property not already protected by easement) to develop a priority ranking for habitat protection (Table 22). This priority ranking is not absolute; individual properties in populations with a medium priority may have greater importance than individual properties in higher ranking populations. Also, state boundaries, administrative boundaries, and other factors influence rankings at those levels. Rankings are relative to one another; a medium ranking is not meant to imply that habitat protection is not important in that population. Rather, habitat loss is likely to be less of an immediate threat in a population with a medium ranking than in a population with a high ranking. This table and the rankings within are intended as a guide to assist agencies in planning, and ultimately in maximizing the efficiency of habitat protection efforts.

Table 22. Relative conservation importance, threat of permanent habitat loss, area subject to threat, and assessment of priority ranking for protection of habitat among populations of GUSG.

Population	Conservation Importance	Threat of Habitat Loss from Housing Development	Private Land, Not Protected, acres (%)	Protection Priority
Cerro Summit - Cimarron – Sims Mesa	Uncertain	High	25,709 (69%)	Medium
Crawford	High	Medium	5,283 (15%)	Medium
Dove Creek	High	High	23,237 (82%)	Medium-high
Monticello, Utah	High (portions)	Low	53,178 (89%)	Medium-high
Gunnison Basin	Very High	Medium	156,055 (26%)	High
Piñon Mesa	High	High	20,052 (52%)	High
Poncha Pass	Low	Medium	4,900 (24%)	Low-medium
San Miguel Basin	High	High (portions)	52,522 (52%)	Very high

It is apparent from this analysis that the threats that could cause permanent or long-term habitat loss for GUSG are substantial, yet vary widely across populations. Substantial public ownership in Crawford, the Gunnison Basin, Poncha Pass and portions of the San Miguel Basin will help mitigate some of these threats, as will no-development easements held by CDOW, UDWR, NRCS, and non-governmental organizations (NGO's). Conversely, substantial portions of the Dove Creek - Monticello, Piñon Mesa, Cerro Summit – Cimarron – Sims Mesa, and some portions of the San Miguel Basin areas are privately owned and are located in areas where significant population growth is expected. Some increase in housing and other development can probably be accommodated in these areas without significantly impacting GUSG, but we hypothesize that densities much in excess of 1 housing unit/320 acres will cause GUSG populations to decline. Greatest impacts are likely when seasonal habitats most important to GUSG, such as areas used during moderate to severe winters, or lek/nesting/brood-rearing areas, are lost. This distinction is probably lost in small populations where, because of small size and existing or potential fragmentation, any loss of habitat may negatively impact grouse.

Any attempt at prioritizing the importance of populations for protection purposes is likely to be polarizing, yet it is necessary to ensure that scarce resources accomplish the greatest good towards the protection of the species. We attempted to incorporate the relative conservation importance of the population (based largely on population size and hence, viability), current population and housing density, projected increase in population and housing density to 2020, amount of land already protected by virtue of public ownership or easement, energy development potential, and known subdivisions to categorize priority for protection from permanent loss.

The San Miguel Basin rated the highest (very high) in terms of protection priority, by virtue of high conservation importance, high projected growth, high energy development potential, and large amounts of private land, not otherwise protected (52%; Table 22). This population appears to at least minimally serve to genetically connect several of the other populations.

The Gunnison Basin and Piñon Mesa rated high in terms of protection priority, for different reasons. The Gunnison Basin has a large amount of public land, but some very important habitat areas are on private land and subject to development. Current human densities are relatively low, and future population increases are low relative to other populations, but the extreme conservation importance of this population suggests any loss of important habitat must be prevented or mitigated. Over half the area of Piñon Mesa is privately owned, and current and future housing densities are projected to be much higher than Gunnison. These threats, and the size of this sage-grouse population warrant a high protection priority.

The Dove Creek - Monticello population ranked medium-high in priority. Its conservation importance is high because of past and potential future population size, but threats in general are not as immediate as in some other populations. If energy development increases in this area, priority for this population may increase. Relatively low current human densities, and low growth rates, predominately concentrated in urban areas, suggest housing development threats are lower than other areas. The major exception to this is the Secret Canyon subdivision on the Dove Creek side, a 2,700-acre area already platted into ranchette lots, with some already developed. Lack of water and power are presently restricting development, but this should be seen as an opportunity for acquiring or protecting key parcels and not as a long-term impediment to development.

Crawford has high conservation importance, but is largely (85%) publicly owned or protected, which makes it less of a priority (medium) for protection than other populations. This doesn't mean remaining key parcels should not be considered.

Cerro Summit – Cimarron – Sims Mesa, or at least some of this area, is facing significant growth and development potential. The uncertain status of this population in terms of size, viability, and connectivity to other populations makes it relatively lower in priority for protection (medium) than other populations. This ranking should be revisited if additional research indicates a significant change in status.

Finally, Poncha Pass was given a low-medium ranking. Its small area and small population size preclude it from having a high conservation importance, and although there is some threat of development, it is not as high as in other areas. Opportunities to protect the remaining 4,900 acres not already publicly owned should be pursued opportunistically, particularly when parcels in higher ranking populations are not available.

C. Habitat Linkages Among GUSG Populations

Theory and Background

Using corridors to link isolated populations is often proposed as a conservation strategy for species in fragmented landscapes (Mann and Plummer 1995, Meffe and Carroll 1997, Rosenberg et al. 1997). It is assumed the linkage will increase movement between populations and will decrease the probability of extinction of the species by stabilizing population dynamics (i.e., reducing the threat of demographic stochasticity) and reducing the threat of inbreeding depression. However, studies have been unable to demonstrate that individuals actually use corridors, much less whether corridors influence the demographic parameters that increase the probability of survival of the species (Simberloff and Cox 1987, Hobbs 1992, Beier and Noss 1998).

Habitat linkages do not necessarily mean corridors. Corridors are defined as narrow, linear strips of habitat typically used by a species that connect larger blocks of habitat and are surrounded by unsuitable (unused) habitat (Turner et al. 2001). We have defined linkages as a heterogeneous landscape, within the historical range of GUSG, composed of isolated patches of landcover types frequently used by sage-grouse (for a list of landcover types see Tables 23 [pg. 165] and 24 [pg. 166]). Habitat within linkages is composed of a mosaic of contrasting land forms, landcover types, and land uses.

The effectiveness of a potential linkage will depend on the ability of GUSG to move among the isolated patches in a landscape; i.e., the relative "connectivity" of patches in a landscape (Taylor et al. 1993). The ability of sage-grouse to disperse may be influenced by the landscape composition (how much of the suitable landcover types are present in the landscape), configuration (the size and shape) of the patches, distance between patches in the landscape (Dunning et al. 1992), as well as the physical nature (land forms) of the landscape that can either facilitate or impede dispersal (Henein and Merriam 1990). These factors are not completely independent. Increased habitat composition is typically correlated with increased patch size and decreased distance between patches. The effectiveness of a potential linkage will also depend on the quality of habitat in the isolated patches and the relative ability of sage-grouse to use (or move through) the surrounding unsuitable habitat. The effectiveness of linkages may also depend on predator behavior. The linear nature of corridors or the fragmented patches of habitat in a linkage may lead to greater predator foraging efficiency (Phillips et al. 2003).

Methods are available for quantifying landscape composition and configuration (Turner 1989, Turner et al. 1991, McGarigal and Marks 1995) and connectivity (Fahrig and Paloheimo 1988a, 1988b, Heinen and Merriam 1990). There are very few empirical data on the connectivity of landscapes for a given species; however, the idea has led to the development of increasingly complex percolation (or diffusion) models (Czaran 1998). These models involve generating 2-dimensional grids ("landscapes"). Each cell of the grid is assigned a particular landcover type (most models use only 2 landcover types: “used” and “not used”). The arrangement of the cells within the grid is manipulated to represent varying degrees of patch size, shape and distribution. By varying movement capabilities (dispersal distance), the models can be used to analyze the ability of a hypothetical animal to move ("percolate") across the grid. These models have shown that changes in landscape composition, patch size, distance between patches, corridor length and width can affect species dispersal, abundance and probability of extinction (Fahrig 1997, 2001, 2002, Haddad 1999, With 2002). These models have also illustrated thresholds in habitat fragmentation that affect a species' ability to move through landscapes (With and Crist 1995, With 2002) and the species' probability of extinction (Fahrig 2001, 2002). In these models, increasing fragmentation has little effect on movement and species persistence until a critical threshold of fragmentation impedes the ability of individuals to disperse and survive (i.e., the distances between patches become too large and the amount of habitat in the landscape becomes too small).

While percolation models are instructive, the question remains whether our proposed linkages contain the appropriate habitat to be effective avenues for movement between populations by sage-grouse. Seasonal movement and dispersal patterns of GUSG are not known well enough to be able to predict whether the birds will use linkages, or if they do, what composition and configuration of landcover types within the linkage will best facilitate movement and keep confounding factors (such as predation) to a minimum. Our GIS analysis has identified extensive potential areas for linkages between current populations (see "Mapping Potential GUSG Habitat Linkages" below), but the quality of the landcover types, relative to movement requirements, remains unknown. It is also not certain that sage-grouse will restrict dispersal movements to landcover types frequently used during seasonal movements, or if they will use atypical sage-grouse habitats (e.g., agricultural lands and right-of-ways). Furthermore, it is not clear what the effect of current population distributions will have on the probability of individuals using linkages. Individuals from small populations may be less likely to disperse across linkages (i.e., behave more like a non-migratory population) than individuals from larger populations that may already exhibit migratory behaviors. Understanding the effect of landscape structure on dispersal patterns of GUSG is a critical step toward evaluating the effectiveness of the proposed population linkages.

Mapping Potential GUSG Habitat Linkages

We used GIS data to describe potential habitat linkages among populations in Colorado and Utah. Data used for Colorado were recently available through the CVCP (Colorado Division of Wildlife 2004b). In this data set, vegetation layers were derived from 30-m Landsat TM satellite imagery. For Utah we used the vegetation layer from the Utah Gap Analysis (also from 30-m Landsat TM satellite imagery; Edwards et al. 1995). Soils data layers would have been beneficial in the delineation, but these data are not available in digital format in all areas.

We selected vegetation classes that contain current sagebrush communities, as well as those classes that may have contained sagebrush communities historically (e.g., piñon-juniper - sagebrush mix). Linkages are comprised of a non-contiguous and patchy mix of the classes (Tables 23 and 24).

Table 23. Vegetation classes from the Colorado Vegetation Classification Project used to identify GUSG habitat linkages in Colorado (Colorado Division of Wildlife 2004b).

Class Name	Class Description
Rangeland	Consists of grass/forb range, shrub/brush range, or mixed range.
Shrub/Brush Rangeland	Consists primarily of sagebrush, saltbrush, greasewood, and snakeweed.
Bitterbrush Community	Shrubland principally dominated by bitterbrush. Often associated with rabbitbrush, sagebrush, greasewood, various grasses, and mixed cacti.
Salt Desert Shrub Community	Low-elevation shrublands found on alluvial salt fans or flats. Component species may include: saltbushes, greasewood, sagebrushes, horsebrushes, and spiny hopsage.
Sagebrush/Grass Mix	Codominant sagebrush shrubland and perennial grassland.
Sagebrush Community	Sagebrush with rabbitbrush, bitterbrush.
Sagebrush/Gambel Oak Mix	Shrubland codominated by big sagebrush and Gambel oak.
Mesic Mountain Shrub Mix	Oak dominant with sagebrush, snowberry, grass.
Snowberry/Shrub Mix	Mountain deciduous shrubland dominated by mountain snowberry. Often associated with Saskatoon serviceberry, sagebrush, squawbush, rabbitbrush and Gambel oak.
Sagebrush/Greasewood	Shrubland co-dominated by sagebrush and greasewood. Secondary species may include rabbitbrush.
Shrub/Grass Forb Mix	Mixed grass/forb and shrub/grass rangeland.
Sagebrush/Mesic Mountain Shrub	Co-dominant sagebrush mesic mountain shrubland consisting of mountain big sagebrush and any combination of mountain snowberry, service berry, squaw apple or bitterbrush often with a grass/forb understory. Understory species may include, among others, elk sedge, bluegrass, needlegrass, arrowleaf balsamroot, lupines, penstemons, Indian paintbrush, and mariposa lily. Often found at the higher elevations of the sagebrush zone, on north facing slopes, in basins, or on other mesic sites.
Sagebrush/Rabbitbrush Mix	Co-dominant sagebrush and rabbitbrush shrubland. Principal shrub species include basin big sagebrush, Wyoming big sagebrush, rubber rabbitbrush, sticky rabbitbrush, or small rabbitbrush.
Xeric Mountain Shrub Mix	Deciduous woodland (or tall shrubland) dominated by mountain mahogany or curlleaf mountain mahogany. Associated species may include sagebrush, rabbitbrush, Mormon tea, or scattered piñon pine or Utah juniper.
Serviceberry/Shrub Mix	Deciduous woodland (or tall shrubland) dominated by Utah and Saskatoon serviceberry. Primary associated shrub species include big sagebrush, mountain snowberry, and Gambel oak.
Piñon-Juniper-Sagebrush Mix	Co-dominant piñon-juniper and sagebrush.
Piñon-Juniper--Mountain Shrub Mix	Co-dominant piñon -juniper and oak, mountain mahogany or other deciduous shrubs.
Juniper/Mountain Shrub Mix	Co-dominant juniper species and oak, mountain mahogany, or other deciduous shrubs.
Juniper/Sagebrush Mix	Co-dominant woodland and shrubland. Woodland consists of Utah juniper at densities around 25%. Big sagebrush grows in the interspaces between the trees and may comprise 25% cover or more.

Table 24. Vegetation classes from the Utah Gap Vegetation Layer used to identify GUSG habitat linkages in Utah (Edwards et al. 1995).

Class Name	**Class Description**
Sagebrush/Perennial Grass	Co-dominant sagebrush shrubland and perennial grassland. Principle shrub species include sagebrush. Principle grass species include bluebunch wheatgrass, sandburg bluegrass, crested wheatgrass, needlegrass, sand dropseed, blue grama, Thurber's needlegrass, western wheatgrass, Indian ricegrass, and galleta. Associated principal shrub species include rabbitbrush, bitterbrush and oak. Associated principal grass species include cheatgrass.
Sagebrush	Shrubland principally dominated by big sagebrush, black sagebrush, low sagebrush or silver sagebrush. Primary associated tree species include juniper, piñon, mountain mahogany and ponderosa pine. Primary associated shrub species include rabbitbrush, snakeweed, winterfat, shadscale, and bitterbrush.
Grassland	Perennial and annual grasslands. Principle perennial grass species include bluebunch wheatgrass, sandburg bluegrass, crested wheatgrass, basin wildrye, galleta, needlegrass, sand dropseed, blue grama, Thurber's needlegrass, western wheatgrass, squirreltail, and Indian ricegrass. Principle annual grass species include cheatgrass. Primary associated shrub species include sagebrush, shadscale, greasewood, and creosote. Primary associated tree species include juniper.
Agriculture	Row crops, irrigated pasture and hay fields, dry farm crops.*

* The vegetation classification in Utah does not distinguish between agricultural and CRP lands. CRP lands are used by sage-grouse as brood areas, hence the agriculture class is included in Utah for this analysis.

Potential linkages were added to existing mapped areas that include occupied, potential, and vacant/unknown habitats (Fig. 29). Hence, a habitat identified as a linkage may not in and of itself link existing occupied habitat polygons, but the combination of linkage, vacant/unknown, and potential habitats will link occupied habitat polygons. These linkages should be considered only as potential areas.

Fig. 29. Potential linkages in GUSG habitat.

D. Population Viability Analysis

Concepts and Principles

Population viability analysis (PVA) is a risk analysis tool that has been used for about 20 years by conservationists and biologists to predict the relative probability of extinction for a wildlife population under various management scenarios, in order to aid in decision-making for population management (Shaffer 1991, Boyce 1993, McCarthy et al. 2001, Reed et al. 2002). In most cases, PVA uses available population information to develop a model (a simplified representation of a real system) that simulates how the population functions (Shaffer 1991, Boyce 1993). The model can then be used to project various future scenarios and predict resulting outcomes for the population. The model may incorporate many factors that affect the status of a population, such as environmental stochasticity (e.g., normal variation in weather and available food supply), demographic stochasticity (e.g., breeding success, survival), catastrophes (e.g., drought, disease), genetic stochasticity (e.g., inbreeding, genetic drift), and interaction among these factors (Gilpin and Soulé 1986, Shaffer 1991). These factors enter the life of an individual as events that occur with particular probabilities, rather than with absolute certainty, at any given time (see Appendix G).

An individual with extensive knowledge of a population may have a “mental model” of how the population behaves, but this information is difficult to share with others and cannot be assessed objectively or quantitatively. Computer simulations are regularly used in PVA to allow for complex models that are explicitly stated and can be tested (Shaffer 1991, Appendix G).

PVA is particularly effective in making “relative” predictions, such as how a population or species may be affected by various alternative management strategies, or the relative risk to different populations, allowing managers to prioritize conservation efforts among the populations (Beissinger and Westphal 1998, Boyce 2001, Ellner et al. 2002, McCarthy et al. 2003). Another strength of PVA is the complexity that it can accommodate; multiple factors and their interactions can be integrated into the process of evaluating a population’s relative extinction risk (Shaffer 1991, McCarthy et al. 2003). In addition, sensitivity analysis can identify the parameters in the model (e.g., adult survival rate) that have the largest impacts on the modeled population (Reed et al. 2002). PVA results can be used to identify future research needs by exposing the parameters for which data are weakest or lacking (Reed et al. 2002), which is particularly important if sensitivity analysis shows those parameters are key to the population’s persistence.

One of the criticisms of PVA is that the increasing availability of user-friendly PVA software allows some users to generate population persistence predictions without a full understanding of assumptions and limitations in the model, and while ignoring weaknesses in data supporting the model (Beissinger and Westphal 1998, Boyce 2001, Reed et al. 2002). “Absolute” predictions, such as a precise probability of population extinction, are not realistic, but relative predictions are more reliable (Beissinger and Westphal 1998, Ellner et al. 2002, McCarthy et al. 2003). Because a PVA uses a model, it will not present a complete picture of the system of interest, but an approximation of it, and results must be used with this in mind (Reed et al. 2002, McCarthy et al. 2003). PVA will likely be based in part on inadequate data (Beissinger and Westphal 1998, Boyce 2001), especially because data for populations at risk may be limited (Shaffer 1991, Boyce 1993) and the populations may be difficult to study. However, if the limitations are recognized, a PVA can offer an opportunity to direct future research towards obtaining more reliable data, more precise estimates of population parameters, to modify the model to improve its performance, and to frame testable hypotheses about how the population/system functions (Boyce 1993, Beissinger and Westphal 1998, Reed et al. 1998, McCarthy et al. 2003). McCarthy et al. (2003:987) concluded that, "The process of parameter estimation, model construction, prediction, and assessment should be viewed as a cycle rather than a one-way street."

Current Model

Thus, as with many analytical tools, PVA can be very useful in the decision-making process for managing species at risk, but only if used properly (Boyce 1993, Beissinger and Westphal 1998, Ellner et al. 2002, McCarthy et al. 2003). We contracted with the Conservation Breeding Specialist Group (CBSG) to develop a PVA for GUSG (see full report in Appendix G). Dr. Philip Miller of CBSG used a simulation software program called *VORTEX* (Miller and Lacy 2003b) to estimate relative extinction probabilities and loss of genetic diversity over time for various population sizes, and to determine the sensitivity of GUSG population growth rates to various demographic parameters.

VORTEX is a Monte Carlo model that simulates the effects of deterministic forces as well as demographic, environmental, and genetic stochastic events on wild populations. It is an individual – based model that follows the fate of each animal in a theoretical population as the individual encounters various life and environmental events during a given year. These events occur with a user-specified probability, and the model will run for a user-specified number of consecutive years. By following the entire population, it is possible to estimate relative population extinction risk and loss of genetic diversity in a specified time period.

Baseline Parameters and Simulations

Demographic parameters used in the GUSG PVA included type of breeding system, age at first reproduction, several measures of reproductive success, sex ratio, mortality rates, and environmental carrying capacity. We also incorporated a severe drought frequency of a single event (which persists for 3 years) each 100 years, and estimated an increase in chick mortality over the 3-year drought period. For each parameter, we used available data, prioritized as follows: (1) GUSG data; if not available, then (2) GRSG data from Colorado and Utah; if not available, then (3) GRSG data from other locations (see Appendix G for specific data sources used). These are 2 closely related grouse species and we do not expect demographic parameters to vary greatly between these species. Some recruitment data collected by CDOW in 2002 are specific to GUSG. These data were collected during a period of severe prolonged drought and resultant population decline, so any results obtained from these data must be interpreted accordingly. We chose a time interval of 50 years for population projections because we felt uncertainty at 100 years was too great to allow reasonable predictions.

Parameters that we did not incorporate in the PVA included density-dependent reproduction, effects of disease, inbreeding depression, and habitat loss or fragmentation. We have no data to determine which or how demographic rates will be affected by these factors. West Nile virus is a potential threat to GUSG (see "Disease and Parasites", pg. 103). However, our lack of knowledge about the disease precludes us from being able to make reasonable predictions at this time. West Nile virus should be included in future analyses as we learn more about the epidemiology of the virus. Inbreeding depression can potentially influence population parameters in small populations (see "Genetics", pg. 109); however, we currently have no data to evaluate whether inbreeding is a significant factor or whether there is a population size threshold at which inbreeding becomes significant (i.e., which GUSG populations might be at risk because of inbreeding). We have no data to determine which or how demographic rates will be affected by habitat loss or fragmentation. Without this information, the only alternative in the VORTEX software is to truncate the carrying capacity used in the simulations (see Appendix G, pg. G-26). We have no information that allows us to conclude GUSG (or GRSG) demography is density dependent, or to even estimate what the effect might be on GUSG population dynamics. We used the GUSG PVA to evaluate the relative risk of extinction for each population under the current conditions (i.e., the risk of extinction if nothing changes). It is the aim of this plan to minimize additional habitat loss. Therefore, we concluded that a valid GUSG PVA should not include these potential factors until we have some reliable data that can be used to estimate how specific demographic parameters are influenced by the various factors.

Baseline simulations using recent demographic data showed long-term (50 years) growth rates ranging from –5% to 15%, depending on the data sets used to estimate different parameters. This baseline model analysis is instructive in that it provides plausible upper and lower bounds on population growth that are reasonable in the shorter-term (i.e., on the order of 5-10 years).

Relative Extinction Risk

This stage of analysis investigated the relationship between the sizes of GUSG populations and their relative probability of extinction, based on a range of potential intrinsic rates of population growth. Because of the inherent uncertainty in understanding current trends in GUSG population size, we elected to develop the risk analysis under multiple scenarios that differed in their underlying growth rates. This should provide insight into the future potential dynamics of dispersed populations that may be assumed to be growing or declining at rates within the scope of this analysis. We are thereby developing a sort of "template" upon which the future of a given population may be evaluated under presumed conditions of growth and size.

There were a total of 99 separate models (11 long-term growth rates ranging from -4% to 15%, and 11 initial population sizes ranging from 20 to 3,000; Fig. 30; see Table 6 in Appendix G, pg. G-20). Results suggest that very small GUSG populations (< 25) are at a high risk of extinction during a 50-year period, even when the population is expected to increase in size over the long-term. In contrast, individual GUSG populations can be considered "secure" (< 5% extinction probability) if they contain 500 birds or more and have a stable population size (Fig. 30).

Based on this analysis, an attempt was made to fit an equation to the relative extinction risk data at a long-term average growth rate of zero so that an estimate of extinction risk could be obtained for any desired population size. A slightly modified dose-response curve, used primarily in the biomedical community, was used as it seemed an appropriate descriptor of the relationship between population size and relative extinction risk (see Appendix G). The fit of this equation for our data was excellent.

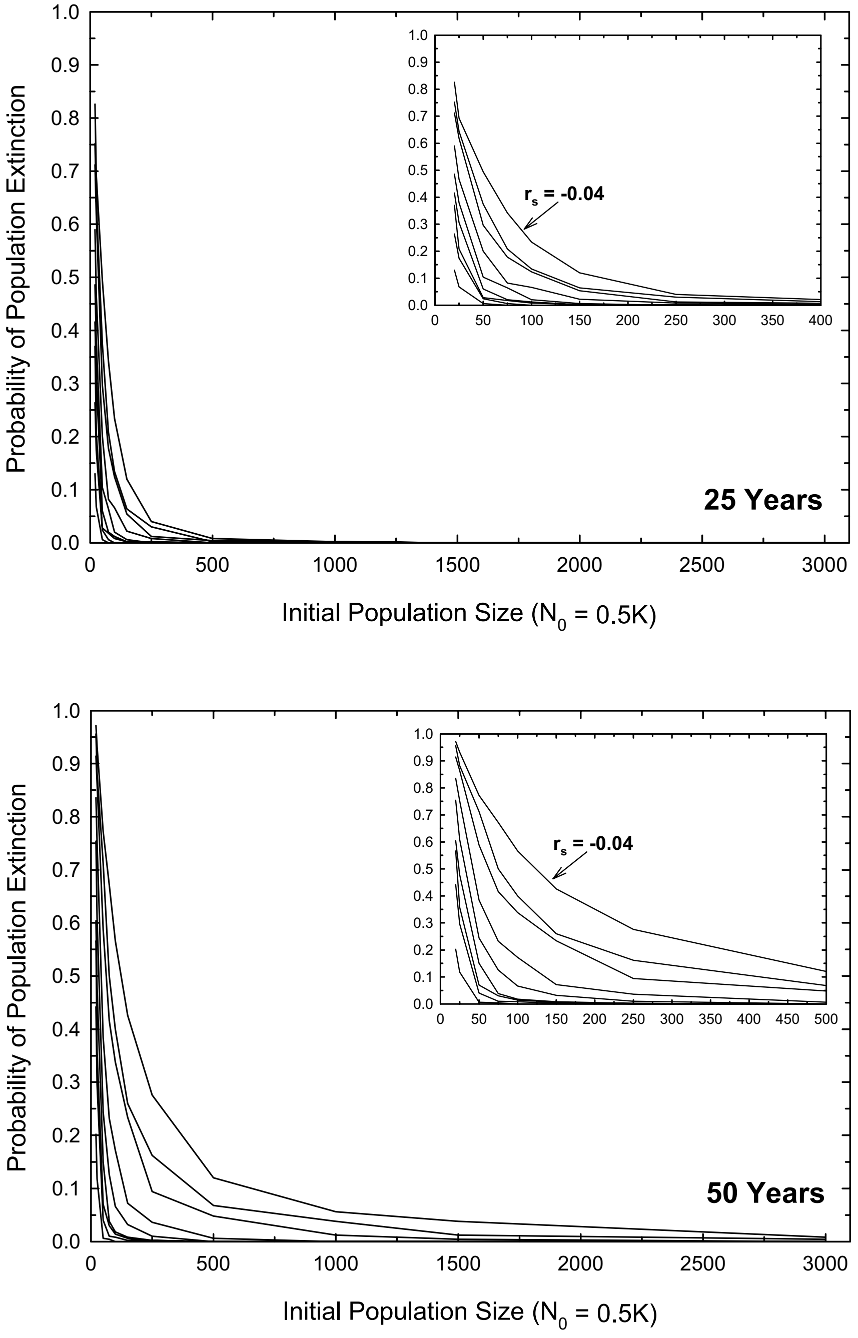

Fig. 30. GUSG population risk analysis. Plots show relative risk of extinction after 25 years (top panel) and 50 years (bottom panel) for simulated populations with specific long-term expected annual stochastic growth rates ranging from -0.04 (towards the top-right portion of each primary panel) to 0.15 (towards the bottom-left portion of each primary panel). For a given initial population size, higher growth rates lead to lower risks of extinction. Smaller inset panels magnify the results for smaller initial population sizes.

Sensitivity Analysis

One of the advantages of a detailed, individually-based population model is that demographic parameters can be varied one at a time across the normal range of variation while holding all others constant to see which have the greatest impact on population growth. An analysis of the sensitivity of population growth rates to variation in various demographic rates (e.g., nest success, adult survival, chick survival) can be an invaluable aid in identifying priorities for detailed research and/or management projects targeting specific elements of the species' population biology and ecology.

To conduct this demographic sensitivity analysis, we identified biologically plausible minimum and maximum values for each of a subset of demographic parameters (Fig. 31). For each of these parameters, we constructed 2 simulations, with a given parameter set at its prescribed minimum or maximum value, and all other parameters remaining at their baseline value. The performance of these alternative models was then compared to that of our starting baseline model (in this case the model that relied most heavily on GUSG data). This analysis suggested that GUSG population dynamics are most sensitive to variation in adult female reproductive success and chick mortality.

The next step was to develop a set of models with the goal of identifying minimum levels of survival necessary to prevent GUSG population decline. A total of 60 individual models were constructed that provided all possible combinations of 2 levels of reproductive success, 5 levels of chick mortality, and 6 levels of adult mortality. This approach also more effectively addressed the relationship between reproductive success and age-specific mortality required for population growth (Fig. 32). Several conclusions can be drawn from the results of these models: (1) greater adult mortality results in less flexibility in allowable levels of chick mortality; (2) higher levels of reproductive success allow for higher levels of acceptable mortality; and (3) the models are more sensitive to chick than adult mortality. Note that this does not necessarily mean these parameters can be improved with management or that these are the parameters that put GUSG populations most at risk of extinction.

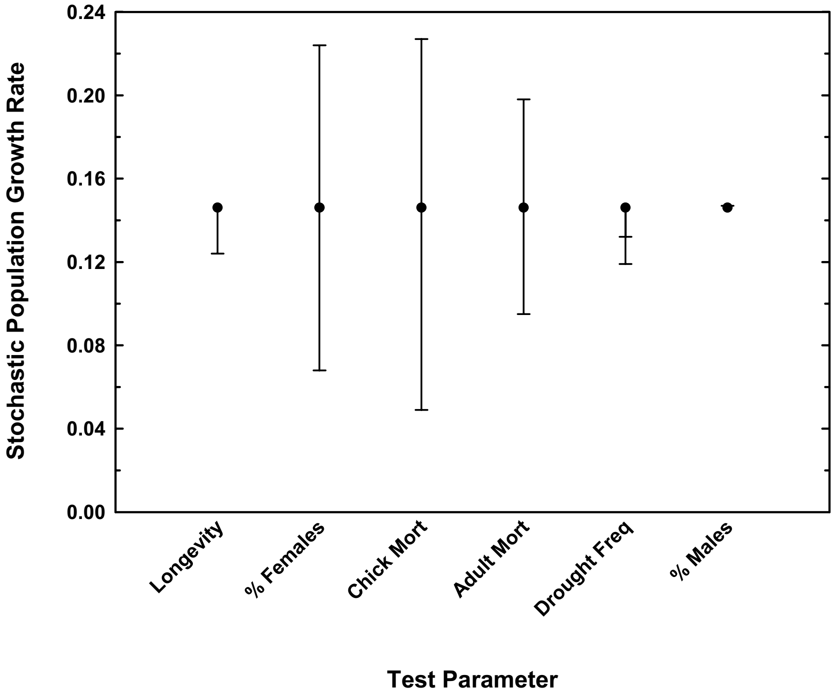

Fig. 31. Demographic sensitivity analysis of a simulated GUSG population. Stochastic population growth rate for a set of models in which the specific parameter is varied across a range of biologically plausible values. The baseline model growth rate of 0.146 is given by the central data point for each parameter. The general model of sage-grouse population dynamics is most sensitive to uncertainty in those parameters giving the widest range in simulated population growth rates.

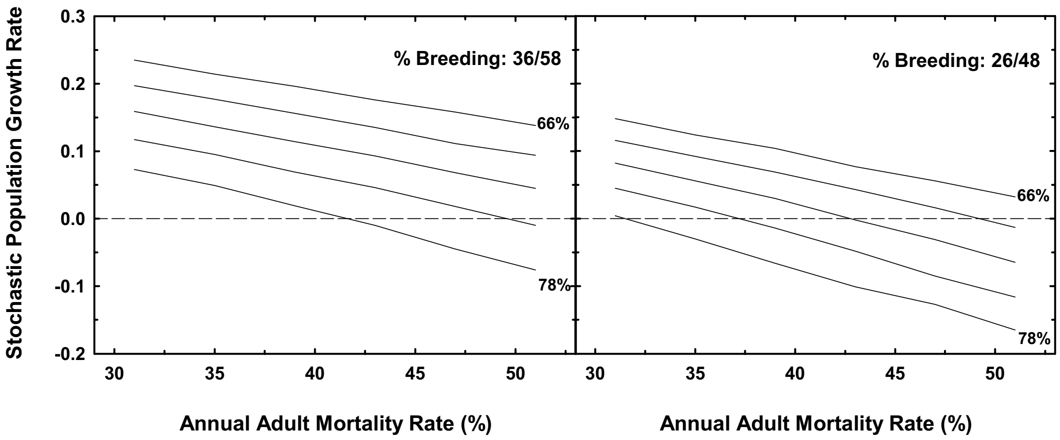

Fig. 32. Gunnison sage-grouse population mortality analysis. Plots give average population growth rate (*r*) as a function of annual mortality rate of adults with individual lines corresponding to different levels of chick mortality. Two panels correspond to variable levels of adult female reproductive success.

Maintenance of Genetic Diversity

The question of genetic diversity maintenance in GUSG populations of various sizes was also addressed. We ran a series of models with initial population sizes of 2,000, 2,500, and 3,000, with carrying capacity set at twice the initial size. The percent of adult males participating in breeding was set at 3 different levels: 10%, 20%, and 33%. Results show that from 90-94% of the genetic diversity can be maintained over 50 years using these parameters (see Table 8 in Appendix G, pg. G-24). Increasing population size from 2,000 to 3,000 birds does not markedly improve the maintenance of genetic diversity at any given proportion of males participating in breeding.

Population Augmentation and Relative Extinction Risk

Several GUSG populations are small enough that sequential drought years, disease, or extreme weather events could push numbers or genetic diversity below the level from which they can recover without population augmentation. In our PVA exercise we simulated the effect on relative extinction risk of adding additional birds to populations (ranging from 100 – 300 birds) when the populations declined, because of stochastic variation, by 50% of the initial population size.

We modeled infusions of 10–40 birds, and had the model make as many transplants as necessary (in a 50-year timeframe) to bring the population back above the 50% reduction trigger. Augmentations effectively reduced the populations' extinction probabilities to zero (Table 25). Genetic diversity retained after 50 years was increased by 21-64%, depending on initial population size, resulting in an overall genetic diversity retained through transplants of 66 – 82% (Table 25).

Under the assumptions used in this model, augmentation with releases of 10 birds were as successful at reducing extinction probability as augmentation with 40 birds, and nearly as successful in retaining genetic diversity (Table 25). Although the model augmented more often when fewer birds were transplanted (because the population stayed below the 50% trigger longer), the total number of birds released over the 50-year period was lowest when 10-bird releases were simulated. For example, for populations of 100 birds, using 10-bird releases resulted in an average of 6.7 releases and a total of 67 birds released over 50 years. For the same population size, 40-bird releases occurred 2.7 times, but the resulting total number of birds released was 108 (Table 25). Looking at any single release size, it took more releases (and hence more total birds) to keep larger populations above the 50% trigger than for smaller populations. This is simply because it takes more birds to increase a larger population by a certain proportion than to increase a smaller population by the same proportion.

We conducted a second augmentation scenario identical to the first except that the trigger for augmentation was a 75% decline from the initial population size. For any combination of release size and original population size, it took fewer releases to raise populations above the 75% (Table 26) than the 50% trigger (Table 25). However, the genetic diversity retained by a population that only had to stay above a 75% decline was lower than one that had to remain above the 50% trigger (Tables 25 and 26).

Table 25. Results of simulations augmenting populations of 100, 200 and 300 birds with 10-40 sage-grouse when populations declined by 50%[a]. Simulations resulted in an extinction probability of approximately 0.

Initial Population Size: 100 (transplant trigger = 50)				
# Birds Transplanted Per Augmentation	Mean # Times Population Received Transplants	Mean # Birds Moved in 50 Years	Genetic Diversity	Mean Population Size (SD)
10	6.7	67	0.657	108 (54)
20	4.4	88	0.666	116 (52)
30	3.4	102	0.690	119 (50)
40	2.7	108	0.675	124 (49)
If no augmentation occurs:				
Probability of extinction within 50 years is 0.334			0.421	93 (63)

Initial Population Size: 200 (transplant trigger = 100)				
# Birds Transplanted Per Augmentation	Mean # Times Population Received Transplants	Mean # Birds Moved in 50 Years	Genetic Diversity	Mean Population Size (SD)
10	8.1	81	0.735	213 (110)
20	5.2	104	0.749	225 (106)
30	3.8	114	0.762	245 (105)
40	3.1	124	0.768	241 (99)
If no augmentation occurs:				
Probability of extinction within 50 years is 0.132			0.563	183 (127)

Initial Population Size: 300 (transplant trigger = 150)				
# Birds Transplanted Per Augmentation	Mean # Times Population Received Transplants	Mean # Birds Moved in 50 Years	Genetic Diversity	Mean Population Size (SD)
10	8.6	86	0.794	313 (169)
20	5.6	112	0.809	341 (160)
30	4.9	147	0.819	364 (159)
40	4.0	160	0.820	359 (154)
If no augmentation occurs:				
Probability of extinction within 50 years is 0.060			0.656	268 (191)

[a] Simulations were run for a 50-year time period. Growth rate was set at 0. Initial genetic diversity was assumed to be 1.0.

Table 26. Results of simulations augmenting populations of 100, 200 and 300 birds with 10-40 sage-grouse when populations declined by 75%. [a] Simulations resulted an extinction probability of approximately 0.

Initial Population Size: 100 (transplant trigger = 25)				
# Birds Transplanted Per Augmentation	Mean # Times Population Received Transplants	Mean # Birds Moved in 50 Years	Genetic Diversity	Mean Population Size (SD)
10	3.0	30	0.591	89 (57)
20	1.8	36	0.601	96 (57)
30	1.5	45	0.600	100 (57)
40	1.2	48	0.594	99 (55)
If no augmentation occurs:				
Probability of extinction within 50 years is 0.334			0.421	93 (63)

Initial Population Size: 200 (transplant trigger = 50)				
# Birds Transplanted Per Augmentation	Mean # Times Population Received Transplants	Mean # Birds Moved in 50 Years	Genetic Diversity	Mean Population Size (SD)
10	2.9	29	0.685	184 (116)
20	2.2	44	0.688	187 (117)
30	1.6	48	0.681	197 (117)
40	1.1	44	0.693	206 (109)
If no augmentation occurs:				
Probability of extinction within 50 years is 0.132			0.563	183 (127)

Initial Population Size: 300 (transplant trigger = 75)				
# Birds Transplanted Per Augmentation	Mean # Times Population Received Transplants	Mean # Birds Moved in 50 Years	Genetic Diversity	Mean Population Size (SD)
10	3.7	37	0.740	279 (184)
20	2.3	46	0.753	288 (173)
30	1.6	48	0.749	300 (176)
40	1.6	64	0.766	294 (171)
If no augmentation occurs:				
Probability of extinction within 50 years is 0.060			0.656	268 (191)

[a] Simulations were run for a 50-year time period. Growth rate was set at 0. Initial genetic diversity was assumed to be 1.0.

Under the simplistic modeling approach used by *VORTEX*, which does not factor in variability in survival rates of released birds, more frequent but smaller releases seem more effective. The potential for stochastic events to completely remove small numbers of released birds suggests releases of at least 20 birds may be warranted if triggers are hit, and perhaps larger for larger initial population sizes. Under the admittedly simplistic model *VORTEX* uses for augmentations, a trigger for beginning transplant efforts at a 75% reduction from the initial population size (or target) was as effective at reducing relative extinction risk and almost as effective at maintaining genetic diversity, yet required less than half as many transplant efforts and birds.

Conclusions

As mentioned earlier, PVA is a useful tool for evaluating relative extinction risks of various size populations, but not for predicting a precise risk of extinction for each GUSG population. We have explored the relative extinction risk and loss of genetic diversity in different population sizes, as well as the relative effects of different population augmentation scenarios. The PVA illustrates that (1) there is a low risk of extinction for the species range-wide (i.e., the risk of extinction for a stable population of 500 is less than 5% over a 50-year period); (2) there is a great deal of uncertainty in the risk of extinction in the small populations due to our lack of information on the demography of GUSG and the effects of habitat loss, fragmentation and disease on GUSG behavior and population trends; (3) 90-94% of the genetic diversity can be maintained over 50 years; and (4) augmentation of approximately 10 birds every 5 years into the smaller populations can significantly reduce their risk of extinction.

Even though the smaller GUSG populations have a relatively high probability of extinction, they are vital to the long-term success of GUSG. Multiple populations across a broad geographic area provide insurance against a catastrophic event threatening the entire species. In addition, the aggregate number of individuals across all populations increases the probability of demographic persistence and preservation of overall genetic diversity by providing an important genetic reservoir.

We used the GUSG PVA to assist us in evaluating the relative conservation value of each GUSG population (Table 41, pg. 303) and to identify critical gaps in our knowledge of GUSG (see “Research” rangewisde strategy, pg. 247, Objective 1, Strategies 4 and 6; Objective 2, Strategy 1; Objective 5, Strategies 1, 2, 3 and 4; and Objective 6). The predictive nature of the model can be improved with more reliable data of GUSG behavior and population trends.

E. Population Augmentation

Translocation

Translocation of GUSG has been proposed as a means to augment small populations. A donor population would provide birds to augment either the population size or genetic diversity of a smaller recipient population, or to establish a new population. Current techniques for transplanting prairie grouse are labor intensive, expensive, and only moderately successful (Toepfer et al. 1990). The typical approach for transplanting sage-grouse has been to obtain birds during the spring. The grouse are captured at night on or near leks, using spotlights and long-handled nets (Giesen et al. 1982, Wakkinen et al. 1992). Birds are transported to the release area and released at daybreak the following morning, using a "soft-release" technique (Musil et al. 1993). This involves placing the birds in a release box on a lek and remotely opening the door when display activity begins at dawn. Ideally, birds walk out of the box and associate the release area with breeding activity.

CDOW has had some success with this technique (see Poncha Pass "Population Information", pg. 93), as have others (Musil et al. 1993), but capturing sufficient numbers of individuals can be difficult. In addition, adult males captured in the spring have already established a territorial affiliation with leks. Some transplanted males have been depredated when they move long distances in an apparent attempt to return to these leks. Juvenile males move much less and appear more willing to accept the release lek and area, presumably because they have not yet established a behavioral affiliation with a lek. Transplanting only juveniles makes obtaining sufficient numbers of birds even more problematic because there are relatively few of them, and they tend not to roost on and near leks where they can be more easily captured.

To date, female sage-grouse translocated in the spring have not attempted to nest during the year of capture, whether caught early or late in the breeding season (CDOW, unpublished data). Thus, translocated hens must survive for a year from release to contribute to population growth. With an average adult female survival of about 65% and nest success of 50% or less, many hens must be moved for a transplant to result in females successfully breeding and further augmenting the recipient population.

The Gunnison Basin GUSG population is the obvious source population for transplant purposes, both because it is so much larger than all others (and thus more able to absorb the loss), and because it has the most genetic diversity. Concern has been expressed that loss of birds trapped during the breeding season may impact the population in the Basin. Although the number of birds trapped and transported out of the Gunnison Basin has been small (51 over 3 years; 2000-2002), it is apparent that removing females during spring will reduce recruitment of young. Because mortality is already high in early life stages for sage-grouse (eggs, chicks, and juveniles), removing individuals in any of these stages for transplantation will likely not add to the mortality of that stage. Thus, moving eggs, chicks, or young of the year, instead of yearling and adult birds during the breeding season, would be far less likely to negatively impact the Gunnison Basin population.

Captive Breeding

Captive breeding could also be used to provide birds for transplant or augmentation purposes. Although the original breeding stock would likely come from the Gunnison Basin, releases beyond that point would come from captive-born progeny, which would eliminate any further impact to the Gunnison Basin population. Extensive experience by Colorado and many other states has illustrated that although raising some gallinaceous birds in captivity is relatively easy, establishing wild populations from these captive-reared birds is very difficult, expensive, and only rarely successful. Failures are usually due to extremely poor survival and reproduction of captive-reared birds (Trautman 1982, Krauss et al. 1987, Leif 1994).

Excessive mortality is usually blamed on behavioral differences between captive-reared and wild-reared birds. Leif (1994) showed that even when captive-reared female pheasants were held over winter and released into high quality habitats just prior to nesting, high mortality and nest abandonment meant they produced only 9% as many young as wild hens in the same habitat. Liukkonen-Anttila (2001) studied differences in morphology and physiology of captive- and wild-reared birds in an attempt to explain the high mortality of released birds. He found significant differences in morphology and physiology caused by captive conditions and diets that may increase mortality of released birds. His findings suggest that some increases in survival might be possible if birds are exposed to more natural diets and allowed adequate space to develop flight and cardiac muscles prior to release.

Sage-grouse

Captive rearing and release programs for grouse are relatively uncommon compared to efforts with turkeys or exotic game birds like pheasants. Bump et al. (1947) raised about 2,000 ruffed grouse in captivity. Even after 12 years of refinement of techniques the authors still noted a propensity for captive-reared chicks to die in large numbers in the first month of life, a trait common to all captive efforts studied, and to the wild. Efforts to raise sage-grouse in captivity date to 1958, when a Texas game bird breeder obtained 30 eggs of GRSG from Wyoming (Pyrah 1960). Twenty-four of the 30 eggs collected hatched (80%), and 17 chicks reached approximately 4 weeks of age. Losses were attributed to accidents, stomach worms, coccidiosis, and inversion of the proventriculus. Only 2 grouse survived to 8-months.

Idaho began a sage-grouse captive breeding program in 1960. Efforts included having captive hens produce young, rearing chicks from eggs collected in the wild, and testing various nutrition plans on sage-grouse (Pyrah 1963, 1964). Success in egg incubation was variable (Pyrah 1963, 1964), and many first-year birds succumbed to disease (salmonellosis, *Pseudomonas aeuginosa*, and aspergillosis; Pyrah 1963). Attempts at captive mating were largely unsuccessful (Pyrah 1963). Survival of the few chicks produced by captive hens was poor and was attributed to poor maternal nutrition during laying (Pyrah 1963). Hatching of eggs collected in the wild was better (87%), and 61% of the chicks hatched survived through the summer (Pyrah 1963). Chick mortality resulted from accidents, disease, and vitamin E deficiency. Wild-caught chicks were more difficult to handle than captive-reared chicks, and “ate sparingly of prepared feed and gained little weight because of it” (Pyrah 1963:8. A diet of pelleted ration with 20% protein, supplemented with “greens and mealworms” was most successful.

Batterson (1997) described successfully propagating sharp-tailed grouse and sage-grouse in captivity in Oregon, without providing details. Batterson and Morse (1948) described an artificial propagation experiment, where 9 eggs were obtained from an abandoned sage-grouse nest and placed under a bantam hen on April 20, 1942. Seven chicks hatched, of which 1 was stepped on and killed by the hen the first day. The 6 survivors were successfully reared to 6-weeks of age when they were released. No information was obtained on subsequent survival.

Wiseman and Bird (1969) conducted a study to develop a ration that would maintain sage-grouse in captivity. They collected 9 eggs from a wild nest in Sweetwater County, Wyoming, and successfully hatched 9 chicks. One chick had its leg severed by the incubator and another had extremely short legs and was destroyed.

Huwer (2004) used sage-grouse chicks hatched and imprinted in captivity to evaluate the extent to which forb abundance affects chick growth rates. She collected 44 eggs from wild sage-grouse nests in spring of 2002 in Middle Park, and successfully hatched 36 (82%) in an incubator. These chicks were imprinted to humans, and subsequently exposed, beginning at 3-days of age for a total of 29 days, to sites with high, medium, or low forb abundance. Mortality during the first week was high; survival to 30-days was 25%. In 2003, 46 of 68 eggs hatched (68%), and survival of chicks through the entire 54-day study period was 68 %.

Other Prairie Grouse

There have been numerous published reports on attempts to propagate other prairie (lekking) grouse in captivity, including lesser prairie chickens (Coats 1955), greater prairie chickens (Trautman et al. 1933, Handley 1935, Ramey 1935, Etter 1963, Shoemaker 1964, McEwen et al. 1969, Kruse 1984), and sharp-tailed grouse (McEwen et al. 1969). Some of these efforts to breed adults and rear young in captivity were successful, although fertility and hatchability rates were often below those seen in the wild; but survival after release was not reported.

Recently, extensive research has been conducted on the endangered Attwater's prairie chicken, in an attempt to develop methods for reintroduction in Texas. In 1990, research began into captive breeding of greater prairie chickens as surrogates for lesser prairie chickens (Jurries et al. 1998). Researchers encountered photoperiod and temperature problems, but ultimately had 3 of 4 hens successfully breed. Eggs collected from wild Attwater's prairie chicken nests were also successfully hatched. However, problems arose with the deaths of 2 wild males brought into captivity, (who died from impaction of the gastrointestinal tract resulting from dietary supplements). Another grouse died of avian pox. The facility also suffered an outbreak of the viral disease, avian reticuloendotheliosis, and was quarantined. Data from this facility and other captive-breeding facilities in Texas indicate the source of the disease was from the outside, likely from migratory birds.

Captive breeding of Attwater's prairie chickens also occurred at the Fossil Rim Wildlife Center and Houston Zoological Gardens. In 1992, eggs collected from wild nests hatched, but most chicks were lost to toe and leg deformities or to an outbreak of infectious enteritis (Smith 1993). Only 5 of the 42 chicks produced survived to breeding age. During 1995-96, 14 hens laid 126 eggs, egg viability was 48%, hatching success was 80% (49 chicks) and 21 chicks were raised to at least 8 weeks of age. Three birds were lost to great-horned owl depredation in the pens and 9 birds were released on the Attwater's Prairie Chicken National Wildlife Refuge.

At the Houston Zoo, 8 females produced 165 eggs, of which 154 were viable; 108 chicks hatched, and 78 chicks survived to 8 weeks. Sixty Attwater's from the Houston Zoo were ultimately released into the wild. A pilot release of 13 males occurred in August of 1995, of which 2 survived to March of 1996. "Refined techniques" resulted in the survival of 31 of 69 Attwater's released in 1996 to the 1997 breeding season. Fifty chicks were released in 1997, supplementing a wild population of 58 birds. There are now captive breeding facilities in Abilene, College Station, Houston, San Antonio, and Tyler, Texas. Ultimately, over 500 eggs were produced.

Recently, several adult pairs were released into individual protected enclosures. This approach has not been successful, suffering nest abandonment, depredation of eggs and young by snakes and fire ants and loss of young to unknown causes. Survival of captive-reared Attwater's prairie chickens released in August to the following spring has been as low as 15% and averaged only 36% despite refinement of release techniques (Preisser and Yelin 1999).

Summary

The literature survey on this topic suggests it is likely, given a substantial commitment of funds and staffing, that GUSG could be successfully bred and raised in captivity. Production capability would not be large because sage-grouse don't breed well in captivity (and as a result they tend to lay infertile eggs) and they are determinate layers who won't continue to lay as eggs are removed (A. D. Apa, CDOW, personal communication). Research into methodologies to collect sperm and artificially inseminate captive hens, or pen construction that would facilitate captive breeding would be beneficial to increase the proportion of eggs that are fertile. There is very limited information on sage-grouse to indicate how likely captive-produced young would be to survive in the wild. However, there is a great deal of relevant information from research on other gallinaceous birds to suggest it will be very low, unless innovative strategies are developed and tested.

Potential Approaches for GUSG

There may be other manipulative strategies to enhance genetic diversity or increase populations of grouse that fall short of captive breeding and release, but that have a higher likelihood of success and would contribute to conservation of these species. We briefly evaluate 5 of these ideas, roughly in order of decreasing potential for success and increasing risk to existing populations.

(1) *Transplant Eggs to Populations in Need*

Oyler-McCance (1999) recommended trapping and releasing 6 GUSG hens into the Dove Creek subpopulation to enhance apparently low genetic diversity. This strategy is further elaborated in the Conservation Strategy (see "Genetics" rangewide strategy, pg. 208). Transplants to augment a failing population at Poncha Pass have been successful to date, but thus far females released have not nested in the year they are trapped and transplanted. One alternative could be to use radio-transmitters to locate nests during laying, and transfer eggs from the source (Gunnison Basin) population or from captive production to nests in populations that need demographic rescue or augmentation to enhance genetic diversity. Clutch size in birds with precocial young that do not require parental feeding may be regulated by nutrition of the hen at the time of laying. Sage-grouse clutch sizes typically range from 7-9, but it is possible that hens could brood and raise substantially larger clutches. This would require further investigation. The technique would require radio-marked females so their nests could be located. Artificial eggs could be placed in the nest bowls so that some eggs remain and prevent abandonment. Other eggs lost to predators could then be replaced with eggs produced in captivity. This would be a means of "ensuring" successful nesting. Given the substantial investment in this approach, it may be worthwhile to evaluate techniques to protect nests from predators (see "Predation" rangewide strategy, pg. 243).

(2) *Incubate Eggs in Captivity to Reduce Depredation Losses*

Nest success in grouse seldom exceeds 50%, and can be substantially lower. Another possible method to increase nest success could be to remove eggs from grouse nests and incubate them in captivity, then replace either eggs or chicks in the nest. This strategy was used very successfully with peregrine falcons where egg-shell thinning was the main problem. Hard plastic eggs were substituted when the real eggs are removed so the female continued to incubate. Huwer (2004) found that GRSG hens in Colorado readily accepted chicken eggs (which are larger and a different color than sage-grouse eggs) when their eggs were removed, continuing to lay and ultimately incubating the clutch. Four of 4 GUSG, and 3 of 3 GRSG hens accepted hard plastic eggs the same size, shape, and color as wild eggs (A. D. Apa, CDOW, personal communication). Using this approach, eggs could be replaced 2-3 days prior to hatch so that normal imprinting occurs, or experiments could be conducted to see if hens accept newly hatched chicks and vice-versa. Pilot studies with GRSG suggest chicks less than 5-days old readily accept, and are adopted by, wild hens (A. D. Apa, CDOW, personal communication).

(3) *Supplement Wild-reared Broods with Captive-produced Young*

For this strategy to be successful, a key assumption is that hens must be willing to adopt captive-reared chicks. There is substantial evidence, only recently collected, to suggest that this technique is possible. The CDOW released 3, 14-day old GRSG chicks to another brood hen last spring when a radio-marked brood hen died. Those chicks were successfully adopted. In a pilot study conducted in the spring of 2004, 17, 1-7 day-old captive reared GRSG chicks were released with wild females with chicks of similar age (A.D. Apa, CDOW, unpublished data). The survival rate at 50 days was 0.42, similar to the survival rate of wild chicks at 50 days (0.38; A.D. Apa, CDOW, unpublished data). CDOW researchers have also observed brood mixing where radio-marked chicks joined broods of different hens. This has also been observed with radio-marked chicks and hens in Oregon (M. Gregg, personal communication) and Idaho (N. Burkpile, personal communication). CDOW researchers have also observed an instance where radio-marked chicks from a depredated hen were adopted by a non radio-collared hen. Apa (CDOW, personal communication) described a hen, known to be unsuccessful in her nesting attempt, who adopted and successfully raised a chick from another brood. Research with radio-marked GRSG chicks in Idaho indicates there is substantial brood mixing among sage-grouse hens (N. Burkpile, personal communication). This suggests that captive-produced chicks can be released into existing broods. The big advantage to this approach is that only broods, not nests, need to be located or disturbed. It is not known whether chicks produced in captivity will accept brood hens, to what extent this might be dependent on chick age at time of release, or whether survival would be similar to wild chick survival. As mentioned above, preliminary information suggests that chicks less than 5-days old readily accept, and are adopted by, wild hens (A. D. Apa, CDOW, personal communication). This will be further evaluated through research.

(4) *Raise Grouse in Captivity and Release to Populations in Need*

This option would be an operational captive breeding and release program. It would require extensive research to evaluate the best methods for raising grouse, including pen construction, diets, artificial insemination, and disease prevention, as well as the best way to reintroduce grouse to the wild. It is the highest risk technique, in that probability of success is low, and there is potential for either introducing disease into existing populations or shifting genetic frequencies over time. The rapid expansion of both chronic wasting disease and whirling disease show how easily release or escape of captive-reared wildlife can create serious disease problems in the wild. If this option is explored it must be under extremely tight disease prevention protocols. Rearing facilities should be placed within the area where release will occur, and the source of birds must be local as well to minimize risk of spreading disease.

(5) *Maintain a Captive Flock as a Genetic Diversity Bank*

The Gunnison Basin is probably the only GUSG population large enough to maintain the genetic diversity needed to offset genetic drift and to ensure the species can adapt to future challenges. At least conceptually, we envision the Gunnison Basin as a source of genetic diversity and individuals that can be used to augment low diversity or population size in case of catastrophic events in other populations. This assumption would be challenged if a catastrophic event like a disease outbreak or prolonged drought occurs in the Gunnison Basin. It may be prudent to maintain a captive flock or flocks (zoos serve this purpose for other species) with diverse genetic makeup to allow us to introduce these genotypes or bring populations back in case of crisis.

F. Analysis: GUSG Population Size in Relation to the Amount of Available Habitat

Model Development

One of the key questions in the conservation and management of GUSG is how much habitat is needed to sustain a given population size over time. We examined this relationship using the mean high male counts at leks and the amount of available habitat within each sage-grouse population. Note that high male counts were used instead of population estimates that are derived from adjusted lek counts. Adjusted lek counts make assumptions that may introduce additional error that cannot be accounted for in model estimates. We used long-term lek counts only if there was a specific protocol and consistent effort for counting leks. We used 10 years of lek count data, 1995-2004, for GUSG populations (except for Poncha Pass, which included only 3 years, 2002-2004; Table 27).

Because there is such a large gap in available habitat between the Gunnison Basin population and the other GUSG populations, we conducted a separate analysis that combined lek counts from both GRSG (Fig. 33) and GUSG populations. We used 6 years of data, 1999-2004, for GRSG populations (except for Middle Park, which included 7 years of data, 1998-2004, and North Park, which included 32 years, 1973-2004; Table 27). Mean high male counts were weighted for each population by the number of years of counts included in the mean. The Sims Mesa and Cerro Summit - Cimarron populations were not included in the analysis. Assessments of the number of males in this population are unreliable because access to these areas has been difficult in spring. We did not include lek counts from the Piceance Basin or Moffat County - Zone 2 (Fig. 33) due to inconsistent lek counts or because the habitat is not comparable to that in other sage-grouse populations.

Table 27. Summary statistics for number of males counted on leks used in the regression for GUSG populations (*n* = number of years of lek counts and includes the 2004 lek counts, * = GRSG population).

Population	***n***	**Area (acres)**	**Mean # of Males**	**Standard Deviation**	**Minimum**	**Maximum**
Poncha Pass	3	14,781	8.0	1.00	7	9
Piñon Mesa	10	24,185	26.3	4.79	16	33
Crawford	10	34,908	39.9	11.58	24	55
San Miguel Basin	10	85,999	62.5	16.46	42	91
Dove Creek – Monticello	10	86,483	66.5	24.16	33	104
Moffat Co. – Zone 1*	6	110,068	168.9	38.93	133	241
Moffat Co. – Zone 4*	6	125,842	159.9	102.45	32	329
Moffat Co. – Zone 6*	6	167,453	353.4	89.67	261	479
Moffat Co. – Zone 3B*	6	207,487	546.9	196.03	236	742
Moffat Co. – Zone 3A*	6	227,087	501.4	153.76	356	709
Middle Park*	7	234,620	264.0	46.61	182	313
Moffat Co. – Zone 3C*	6	280,661	114.5	34.10	56	142
Moffat Co. – Zone 5*	6	300,643	255.3	28.41	202	282
North Park*	32	405,041	877.2	319.21	446	1,521
Gunnison Basin	10	530,464	605.4	95.30	449	723

Fig. 33. Locations of GRSG populations used in analysis of sage-grouse population size and amount of available habitat.

A GIS analysis was used to estimate the amount of available habitat within each population. For this analysis, available habitat is a subset of the vegetation cover types within Occupied Habitat (for definition, see pg. 54): we selected only the cover types within Occupied Habitat that would most likely be used by sage-grouse (e.g., sagebrush or sagebrush-grass communities). Note that this results in different total acreages for each population area than those given in the Conservation Assessment ("Status and Distribution of Individual Populations", pg. 56).

We used several mathematical models to examine the relationship between the mean high count of males on leks and the amount of available habitat (Figs. 34 and 35). We included a linear model that assumed a density independent relationship between the number of individuals in a population and the amount of available habitat. A linear model assumes a constant relationship between population density and the amount of available habitat. The relationship should be linear as long as there is no change in the behavior (e.g., movement patterns) or spatial correlation of sage-grouse as the amount of habitat changes. However, since the plot of the data seems to indicate a nonlinear relationship between the 2 variables, we examined 2 nonlinear models: a quadratic and an exponential model (Figs. 34 and 35). Due to the small number of populations (n = 6 GUSG and n = 9 GRSG populations), we restricted the number of models to a quadratic (i.e., a second degree polynomial) to avoid over-fitting the data. Nonlinear models assume population densities may not be independent of the amount of habitat available and that the behavior and spatial correlation of individuals changes as the amount of available habitat changes. For instance, habitat in smaller populations may be of poorer quality and therefore, may have a lower than predicted population density. In contrast, populations with large amounts of available habitat may have a lower than predicted population density if individuals do not use all available habitat or space use by individuals increases with increasing available habitat.

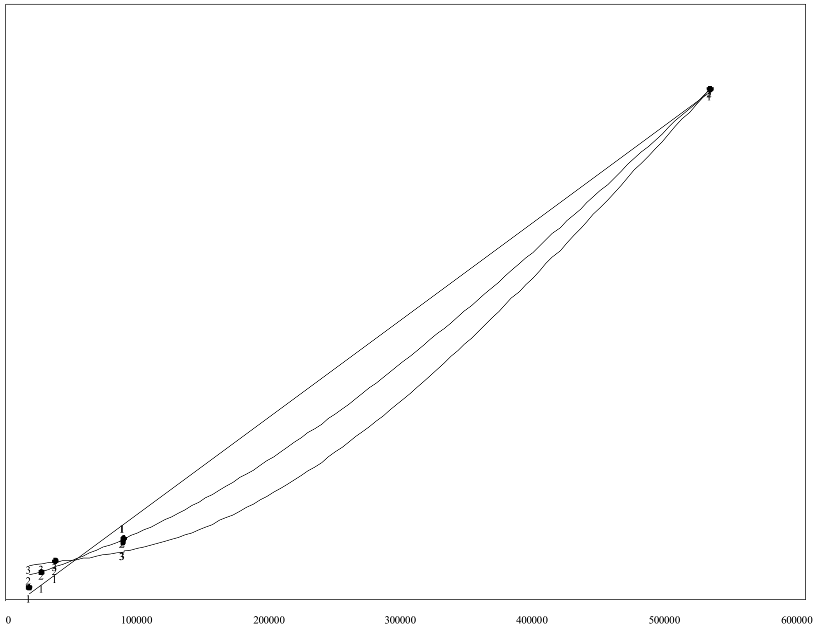

Fig. 34. All models relating the number of males (mean high count at leks) within each GUSG population (● = mean high count of males at leks for each population, n=6; area is in acres). 1 = linear model, 2 = quadratic model, 3 = exponential model.

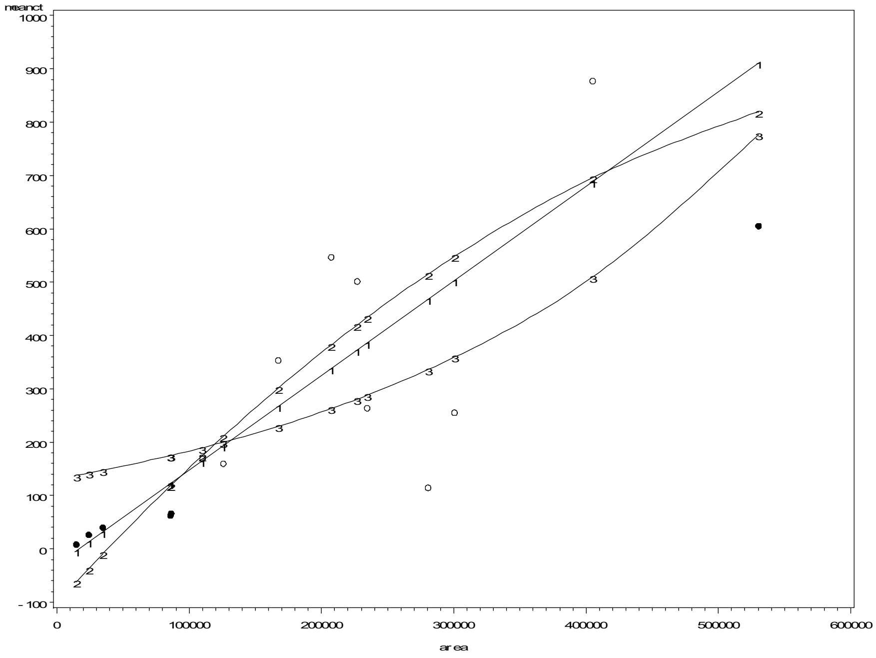

Fig. 35. All models relating the number of males (mean high count at leks) within each GUSG and GRSG population (● = mean high male counts for GUSG populations, n = 6; ○ = mean high male counts for GRSG populations, n = 9; area is in acres). 1 = linear model, 2 = quadratic model, 3 = exponential model.

Model Selection

We used an information-theoretic approach to evaluate the model that best describes the relationship between the 2 variables (Akaike 1973, Burnham and Anderson 1998). Akaike Information Criterion (AIC) is a refinement of maximum likelihood techniques for parameter estimation and is derived from the Kullback-Leibler distance used in information theory (Kullback and Leibler 1951). The Kullback-Leibler distance is a measure of the difference between the data ("reality") and the model used to estimate reality. More specifically, AIC is the maximum log-likelihood for a model with a set of parameters (θ) for a given set of data (y) (AIC = $-2\ln[L(\theta|y) + 2\text{ K}]$, where K is the number of parameters in the model). As the number of parameters in the model increases, the precision of the model increases and the difference between the model and reality typically decreases (i.e., $-2\ln[L(\theta|y)$ gets smaller). However, additional parameters do not always contribute significant information to a model. AIC penalizes a model by the number of parameters used to fit the data (i.e., 2K gets bigger while $-2\ln[L(\theta|y)$ gets smaller). The objective is to select a model that does not over-fit (large number of parameters and highly precise) or under-fit (a simple model with few parameters but not very precise) a given set of data. The model with the smallest AIC value is considered the most parsimonious (i.e., the best balance between simplicity and precision) and therefore, the most reasonable model for a given set of data.

Due to the small number of Gunnison and GRSG populations, we used the corrected AIC (AICc, Hurvich and Tsai 1989) to rank the models. Since AIC (and AICc) is a relative ranking technique, we computed the Akaike weight (w_i) to illustrate the relative likelihood of each model (Akaike 1978). Note that the Akaike weights sum to 1.0. The larger the weight the more plausible the model for making inferences based on the data. All models were log-transformed in order to better meet the assumption of homogeneity of variances across dependent variables (area) and to make the residuals of the linear and nonlinear models comparable for model selection. The original (real scale) data were used to compute the parameter estimates for each model.

Based on the AICc and the Akaike weights (w_i), the linear model is the best model for relating the mean high male lek counts to the amount of available habitat for both GUSG data (Table 28A) and for GRSG and GUSG data combined (Table 29A). The quadratic model ($w_i = 0.25$) could be considered along with the linear model ($w_i = 0.73$) for the GUSG data given that the Akaike weight of the linear model is only approximately 3 times greater than the quadratic model. A general rule of thumb is that a superior model should have approximately 8 times the Akaike weight over competing models (Burnham and Anderson 1998). However, the linear model is more intuitive given the negative y-intercept, which implies a minimum area of available habitat is necessary to support a sage-grouse population (Fig. 35). Using parameters from the discrete linear model (Table 28B) for estimating the number of males on leks using GUSG data is,

$$\hat{y} = -17.05 + 0.0012(\text{area}) + \varepsilon$$

Both the AICc and the Akaike weights (w_i) indicate the linear model ($w_i = 0.82$) is the best model for relating the mean high male lek counts to the amount of available habitat for the combined GRSG and GUSG data (Table 29A). The Akaike weight for the linear model is more than 4 times greater than the quadratic ($w_i = 0.18$) and exponential models ($w_i = <0.01$). Note that the coefficients for the additional parameter in the quadratic models (β_2) for both sets of data do not add much information to either model (Tables 28B and 29B). Using parameters from Table 29B, the discrete linear model for estimating the number of males on leks using the combined GRSG and GUSG data is,

$$\hat{y} = -30.0 + 0.002(\text{area}) + \varepsilon$$

Table 28: A) Log-transformed data for model selection, B) parameter estimates using real scale data for analysis of the number of males (mean high count at leks) for a given amount of habitat within each sage-grouse population (GUSG only).

A. Log-transformed Data: Model	**d.f.**	**MSE**	**F**	**P >F**	**R^2**	**AICc**	**w_i**
1. Linear $\ln(\hat{y})=\ln(\beta_o + \beta_1 x) + \varepsilon$	4	0.71	99.4	<0.001	0.961	- 1.38	0.73
2. Quadratic $\ln(\hat{y})=\ln(\beta_o + \beta_1 x + \beta_2 x^2)+ \varepsilon$	3	0.83	43.0	0.006	0.966	0.73	0.25
3. Exponential $\ln(\hat{y})=(\ln\beta_o + \gamma * area) + \varepsilon$	4	2.03	32.2	0.005	0.889	5.05	0.02

B. Real Scale Data: Model	**β_o**	**β_1**	**β_2**	**γ**
1. Linear $\hat{y} = \beta_o + \beta_1 x + \varepsilon$	-17.05	0.0012	-	-
2. Quadratic $\hat{y} = \beta_o + \beta_1 x + \beta_2 x^2 + \varepsilon$	14.48	0.0005	<0.0001	-
3. Exponetial $\hat{y} = \beta_o e^{\gamma * area} + \varepsilon$	31.48	-	-	0.0056

Table 29. A) Log-transformed data for model selection, B) parameter estimates using real scale data for analysis of the number of males (mean high count at leks) for a given amount of habitat within each sage-grouse population (GUSG and GRSG data combined).

A. Log-transformed Data: Model	**d.f.**	**MSE**	**F**	**P >F**	**R^2**	**AICc**	**w_i**
1. Linear $\ln(\hat{y})=\ln(\beta_o + \beta_1 x) + \varepsilon$	13	2.08	93.9	<0.0001	0.878	13.3	0.82
2. Quadratic $\ln(\hat{y})=\ln(\beta_o + \beta_1 x + \beta_2 x^2)+ \varepsilon$	12	2.25	43.5	<0.0001	0.879	16.3	0.18
3. Exponential $\ln(\hat{y})=(\ln\beta_o + \gamma * area) + \varepsilon$	13	4.20	39.9	<0.0001	0.754	24.6	<0.01

B. Real Scale Data: Model	**β_o**	**β_1**	**β_2**	**γ**
1. Linear $\hat{y} = \beta_o + \beta_1 x + \varepsilon$	-30.0	0.002	-	-
2. Quadratic $\hat{y} = \beta_o + \beta_1 x + \beta_2 x^2 + \varepsilon$	-101.7	0.003	<0.0001	-
3. Exponential $\hat{y} = \beta_o e^{\gamma * area} + \varepsilon$	131.1	-	-	.00334

Using the mean high male lek count loses information about year to year variation. Therefore, we repeated the above analysis using lek counts from each year instead of the mean. Using the individual lek counts results in the same mean and predicted values from the regression, but the variances are larger and potentially more realistic (Fig. 37). A summary of statistics and parameter estimates for the real scale data for GUSG and for the combined GRSG and GUSG data are provided (Table 30).

The linear model is the most parsimonious and reasonable for both sets of data. However, we recommend the linear model using only GUSG data (Figs. 36 and 37). The combined data (Fig. 38) provided additional information that illustrates the efficacy of the linear model and the potential variability in the number of males for a given amount of habitat. The steeper slope for the combined data implies a more rapid growth in the number of males on leks in response to adding habitat to the population. However, it is more reasonable to use the GUSG data for making inferences about GUSG populations.

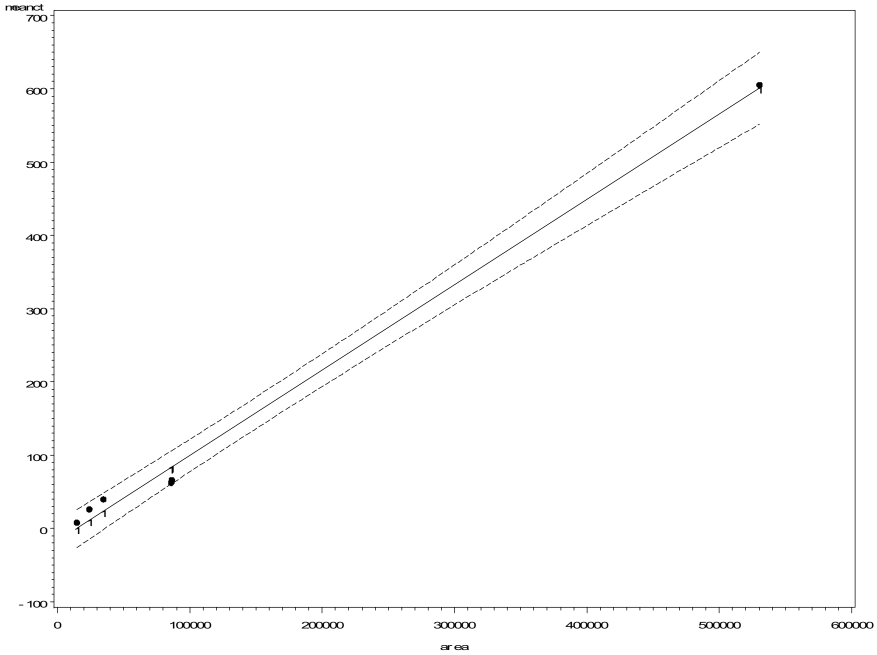

Fig. 36. Linear model (with 95% confidence interval [C.I.] for mean values) relating the number of males (mean high count at leks) within each GUSG population (● = mean high count of males at leks for each population, n = 6; area is in acres).

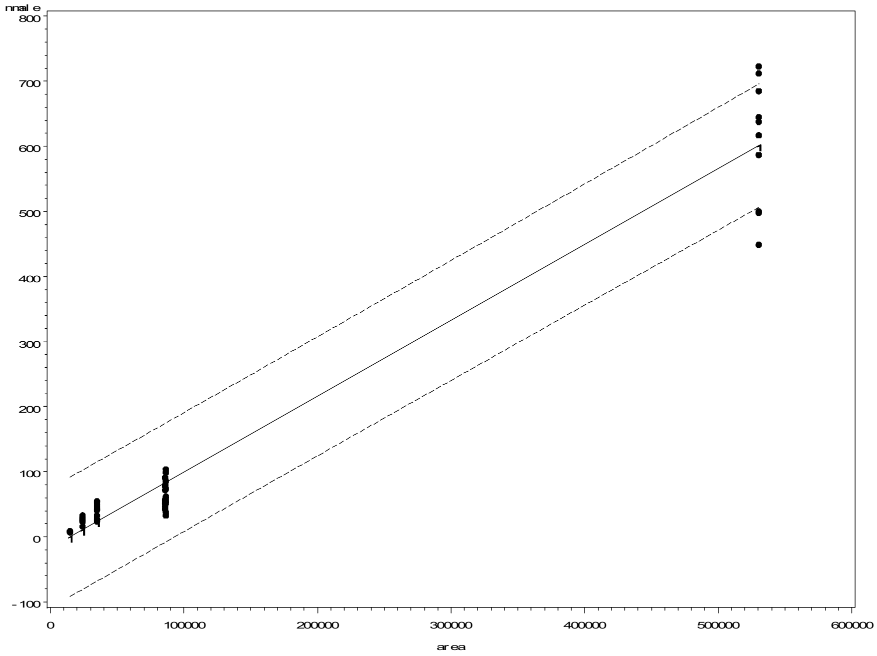

Fig. 37. Linear model (with 95% C.I. for individual predicted values) relating the number of males (mean high count at leks) within each GUSG population (● = mean high count of males at leks for each population, n = 6; area is in acres).

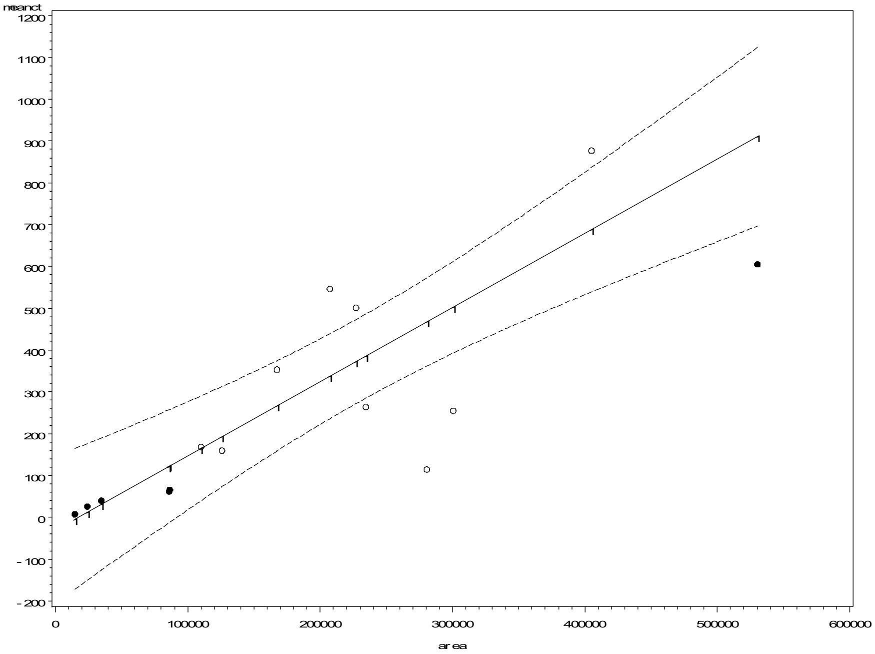

Fig. 38. Linear model (with 95% C.I. for mean values) relating the number of males (mean high count at leks) within each GUSG and GRSG population (● = mean high male counts for GUSG populations, n = 6; ○ = mean high male counts for GRSG populations, n = 9; area is in acres).

Computing 95% C.I. for Predicted Values of the Number of Males Counted on Leks

We recommend using lek counts from individual years (instead of the mean lek counts) to compute a 95% C.I. for predicted number of males (Fig. 37). As mentioned earlier, this model includes the year to year variation in lek counts and is therefore potentially more realistic (i.e., the variance, estimated by the *MSE*, is larger). The 95% C.I. for the number of males (y_i) for a given amount of habitat (x_i) is computed as,

$$\text{C.I.} \;=\; \hat{y} \;\pm\; \mathrm{t}_{0.5,\text{d.f.}} \sqrt{MSE\left[1+\frac{1}{n}+\frac{(xi-\bar{x})^2}{SS(X)}\right]}$$

and the 95% C.I. for the expected mean number of males ($\bar{y}$) for a given amount of habitat is computed as,

$$\text{C.I.} \;=\; \hat{y} \;\pm\; \mathrm{t}_{0.5,\text{d.f.}} \sqrt{MSE\left[\frac{1}{n}+\frac{(xi-\bar{x})^2}{SS(X)}\right]}$$

where $\hat{y}$ is the predicted number of males on leks for a given amount of available habitat (i.e., $\hat{y} = \beta_o + \beta_1(x_i)$), $t_{0.5,d.f.}$ is the critical value for the *t*-distribution for a given number of degrees of freedom ($t_{0.5,51} = 2.009$ for GUSG populations and $t_{0.5,132} = 1.978$ for combined GRSG and GUSG populations), the parameters β_o and β_1 are from Table 30, *MSE* is an estimate of variance from Table 30, *n* is the number of lek counts($n = 53$ for GUSG counts and $n = 134$ for combined GRSG and GUSG counts), x_i is the amount of habitat for a given population, $\bar{x}$ is the mean available habitat computed from values in Table 27 ($\bar{x} = 129{,}470$ acres for the combined GUSG populations, $\bar{x} = 189{,}048$ acres for GRSG and GUSG populations), and *SS(X)* is the sum of squares for available habitat across all populations. *SS(X)* = 197,714,267,836 for GUSG data. *SS(X)* = 301,073,810,470 for GRSG and GUSG data combined.

For example, if there are 50,000 acres of habitat, the predicted number of males on leks ($\hat{y}$) is estimated as,

$$\hat{y} = \beta_o + \beta_1(x_i)$$
$$\hat{y} = -17.05 + 0.0012(50{,}000)$$
$$\hat{y} = 43$$

The parameters β_o and β_1 are for GUSG data (see Table 30). Using the values given above and in Table 30 for GUSG data, the 95% C.I. range for expected number of males is computed as,

$$\begin{aligned} \text{C.I.} &= 43 \pm 2.009\sqrt{2{,}029.5\left[1 + \frac{1}{53} + \frac{(50{,}000 - 129{,}470)^2}{197{,}714{,}267{,}836}\right]} \\ &= 43 \pm 92.8 \end{aligned}$$

The 95% C.I. range for the mean number of males is computed as,

$$\begin{aligned} \text{C.I.} &= 43 \pm 2.009\sqrt{2{,}029.5\left[\frac{1}{53} + \frac{(50{,}000 - 129{,}470)^2}{197{,}714{,}267{,}836}\right]} \\ &= 43 \pm 20.4 \end{aligned}$$

Therefore, the expected number of males could potentially range from 0 to 136 males in any given year and the mean number of males over time should be between 23 and 63 males. See Table 31 for 95% C.I. for other possible amounts of available habitat.

Table 30. Summary statistics and parameter estimates of a linear model using lek counts from each year (real scale data) to analyze the number of males for a given amount of habitat for A) GUSG only and B) GRSG and GUSG combined.

A. Gunnison sage-grouse only							
Model	**d.f.**	**MSE**	**F**	**P > F**	**R^2**	**β_o**	**β_1**
$\hat{y} = \beta_o + \beta_1 x + \varepsilon$	51	2,029.5	1,252.9	<0.0001	0.961	-17.05	0.0012
B. Greater and Gunnison sage-grouse							
$\hat{y} = \beta_o + \beta_1 x + \varepsilon$	132	56,539.2	192.5	<0.0001	0.593	-30.0	0.002

Table 31. 95% C.I. for expected number of males (95% C.I. : y_i) and expected mean number of males (95% C.I. : $\bar{y}$) for a select range of habitat acreages.

Habitat (acres)	$\hat{y}$	**95% C.I.: y_i**		**95% C.I.: $\bar{y}$**	
		Minimum	**Maximum**	**Minimum**	**Maximum**
20,000	7.0	0	101	0	32
25,000	13.0	0	107	0	38
50,000	43.0	0	136	23	63
85,000	85.0	0	177	70	100
150,000	163.0	72	254	150	176
300,000	343.0	245	441	306	380
500,000	583.0	464	701	507	659

G. GUSG Population Targets Development

Population targets must take into account that even healthy sage-grouse populations fluctuate tremendously over time and should be used as relative indicators of a given population's status and trend. A good example is in North Park, Colorado, where lek counts of GRSG have been monitored with similar intensity of effort for over 30 years. The average number of males counted on leks was 862, but that average was punctuated by counts as low as 497 (1986) and counts as high as 1,521 (1979) (Fig. 39). Thus, even in an area of relatively stable habitat, 2-3 times more males were counted in high years than in low years. Total males counted in low and high years were 60 and 176% of the long-term average, respectively. Given this variation, lek counts in most years will be substantially above or below the long-term average (Fig. 39).

We chose population targets for GUSG that are based on a long-term population average and modified by the potential for GUSG to expand into vacant or potentially suitable habitat (see Table 32, pg. 256). The long-term population averages for GUSG are based on 10 years of lek counts from 1995-2004 (except for Poncha Pass which included only 3 years of lek counts, 2002-2004; see Table 27, pg. 187 for mean and standard deviation of number of males counted on leks for each GUSG population). Vacant and potentially suitable habitat is based on a GIS analysis and modified by expert opinion of CDOW and UDWR biologists. These modifications are unique for each GUSG population (see discussion for each population in the "Local Conservation Targets and Strategies", beginning pg. 255). We also present a population range that illustrates the normal expected population fluctuation.

Since population targets are based on current population estimates and potential habitat conditions, they should be modified as habitat conditions and availability change. We do not know, and can not predict, the effect of changes in landscape features (e.g., habitat composition, patch configuration, and land use patterns) on GUSG behavior and population dynamics; therefore, population targets should be modified as we improve our knowledge of landscape features and how they are used by GUSG (see "Adaptive Management Process" pg. 302). We anticipate increases in population levels and targets upon implementation of the habitat management strategies described within the RCP that will minimize the likelihood of extinction or endangerment of GUSG.

We also define a minimum population threshold below which certain conservation actions should be initiated and expedited. If a series of population estimates for a given population continually declines toward a threshold, managers should increase efforts to evaluate the decline and potential conservation actions before the population passes the threshold. We set a reasonably conservative threshold at 30% below the current population targets. In the North Park data set (Fig. 39), the number of GRSG males counted on leks declined below a 30% threshold (approximately 600 males) in 6 of 31 years. Since North Park is a relatively stable population, the 30% threshold creates an error rate (false-positives) of 19% (6/31). If the first 3 years of the data set are excluded (many lek locations were still being discovered), then male counts fell below the threshold in only 3 of 28 years, for a false positive rate of about 11%. This seems to give a reasonable probability of detecting real long-term declines while protecting against panic when population declines are within normal ranges of variation. While the population targets should be modified as conditions change, the threshold is based on current conditions and will not change (i.e., we consider current population and habitat conditions to be the baseline for evaluating future GUSG trends as well as the basis for determining whether to expedite conservation activities even as population levels increase).

The PVA developed for GUSG (see "Population Viability Analysis", pg. 168) describes the relative risk of extinction across a range of population sizes. The PVA illustrates, 1) that there is a low risk of extinction for the species rangewide (i.e., the risk of extinction for a stable population of 500 is less than 5% over a 50-year period) and 2) there is a great deal of uncertainty in the risk of extinction in the small populations due to our lack of information on the demography of GUSG. The uncertainty is increased by our lack of knowledge about how habitat loss, habitat fragmentation, disease, and landscape changes may affect the viability of GUSG populations. Therefore, we did not use the PVA to set specific population targets.

Despite these uncertainties, the risk of extinction is relatively higher in the smaller populations than in the Gunnison Basin. The smaller populations are important to the long-term viability of GUSG because they: 1) increase species abundance rangewide, 2) minimize the threat of catastrophic events to the species since the populations are widely distributed across the landscape, and 3) provide additional genetic diversity not found in the Gunnison Basin.

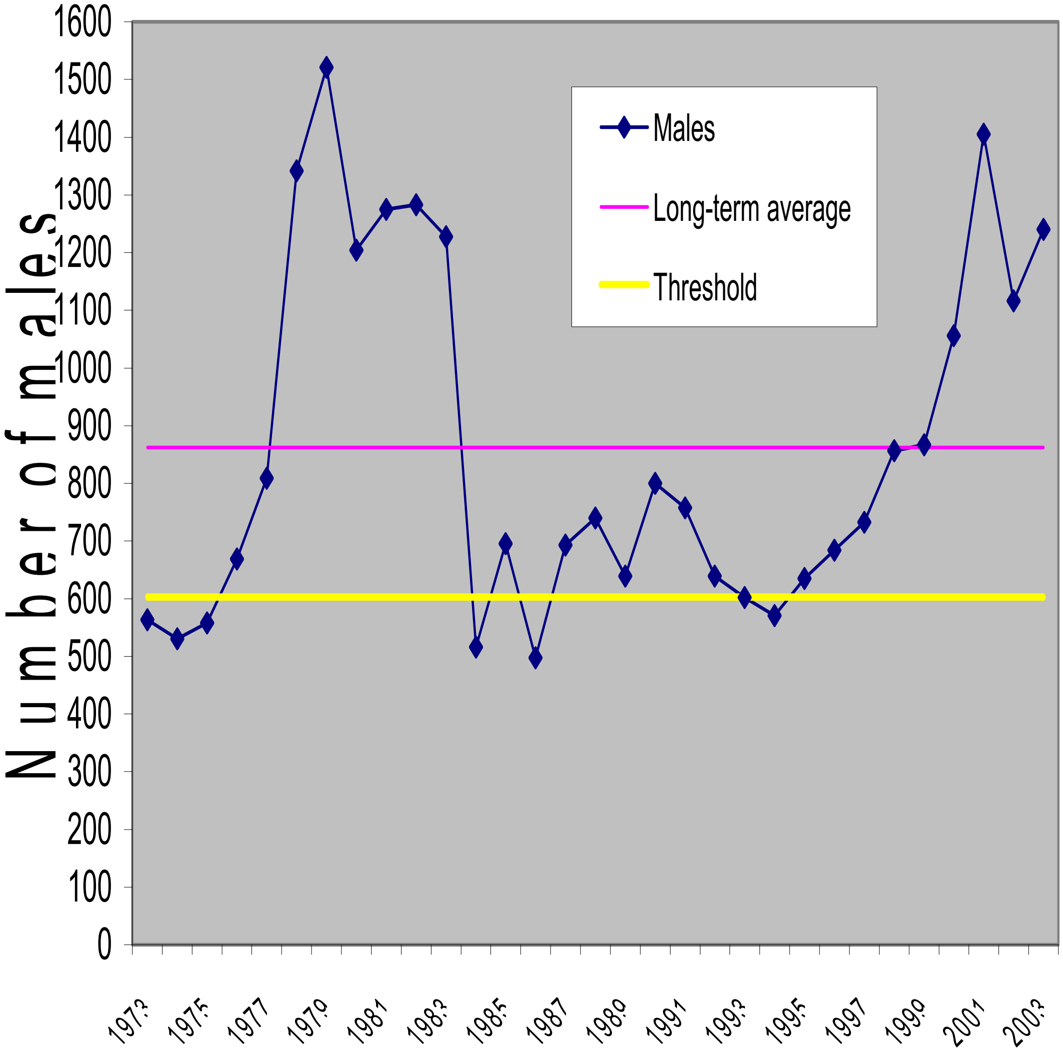

Fig. 39. Example of lek count target-setting in a long-term stable GRSG population in North Park, Colorado, from 1973-2004.