January 17, 2023

Dixie National Forest

Powell Ranger District

Attn: Christopher Wehrli, District Ranger

PO Box 80

Panguitch, Utah 84759

RE: COMMENTS ON THE PROPOSED SHOWALTER PIPELINE PROJECT ENVIRONMENTAL ASSESSMENT

Hello,

Native Ecosystems Council, the Alliance for the Wild Rockies, Yellowstone to Uintas Connection, and Center for Biological Diversity would like to submit the following comments in regards to the Environmental Assessment (EA) for the proposed Showalter Pipeline Project on the Powell Ranger District of the Dixie National Forest. Please note we have attached 4 reports with these comments: Decline of the North American avifauna by Rosenberg and others (2019); Ecology and Management of Neotropical Migratory Birds: a synthesis and review of critical issues (1995) by T. Martin and D. Finch; relevant portions cited in these comments of the Vertebrate Information Compiled by the Utah Natural Heritage Program: a progress report (2003); and Proposal for a System of Federal Livestock Exclosures on Public Rangelands in the Western United States by Bock et al. 1993, Conservation Biology 7-731-733. Although adherence to Forest Plan direction for management of the Greater Sage-Grouse (hereafter "sage grouse") is the one of the most significant issues we have for this proposed project, management of all birds that occur in this project area is necessary, especially those identified as species of conservation concern, as is the sage grouse. The added references help identify what some of these other birds species are.

1. It is clear that the 2 grazing allotments that define the project area (Pines and East Pines Cattle and Horse Allotments) need to have revised Environmental Assessments (EAs) as an integral part of any proposed modifications.

Information provided in the EA clearly demonstrate that the 2 grazing allotments are not working as intended, due to insufficient water availability. This brings up the question, since the allotted AUMs were based on total acres of these allotments, and not water availability in general, what has been the impact of grazing as a result? With cattle significantly limited in the areas available for grazing due to unreliable or nonexistent water, what has been the grazing pressure on those areas cattle have been able to graze? It seems like the current allotment management plans have resulted in severe overgrazing pressure in many areas of these allotments. The existing condition and use of these 2 allotments needs to be defined, along with existing impacts. If there are severely overgrazed areas, how will these area be addressed in the future? If overgrazing has been a general impact of past management, why will this change with new management? Many questions about past management need to be addressed in new EAs for these allotments, including how Forest Plan direction for sage grouse has been met, as well as how grazing impacts in riparian areas has affected these areas. Also, if more water is made available, will there be more cows put on these 2 allotments? How many acres that were generally ungrazed will now have much more extensive grazing pressure? Overall, the addition of 6 new stock tanks to each allotment will create huge changes in the existing grazing program, changes that need to be evaluated. In particular, there will be many more acres of this landscape that will not be grazed much more heavily than occurred in the past. These changes in grazing pressure need to be evaluated as to how this will impact wildlife, including both game and nongame species, including the sage grouse and mule deer.

In summary, adding new water developments is new management for the 2 grazing allotments, with greatly expanded grazing acres. which requires a new EA and public involvement for each allotment.

2. Please include an economic analysis with the new EAs that are required to address new management of the affected allotments.

We would like to know what the costs of the proposed water improvement will be, and how these costs relate to the financial costs and benefits of the grazing programs on both allotments.

3. The impacts on water distribution/availability of this landscape as will be impacted by a new well needs to be fully evaluated and defined to the public; how will adverse impacts on ground water reductions (e.g., drying out of springs important to wildlife) be mitigated?

The EA wildlife analysis suggests that stock tanks created with this project will improve water availability for wildlife. It is not clear what the basis for this claimed improvement is. Most nongame wildlife will not use stock tanks for watering, including the sage grouse. The loss of spring water sources for nongame species, including small mammals, will not be replaced with stock tanks. As a result, there will be a reduction in water availability for nongame wildlife, an impact that cannot be mitigated. The total acreage of land where water availability will be reduced for wildlife needs to be included in this analysis, along with the expected reductions in these populations. In addition, it is very common that stock tanks are a high risk factor for nongame birds and other wildlife, since escape ramps in these tanks are frequently not implemented. What will be the expected average mortality level to birds per new stock tank, based on the average failure rate of putting in escape ramps for wildlife?

4. Although the impacts of both past and increased grazing levels on sage grouse were completely ignored in the project Biological Evaluation, this impact needs to be fully assessed in an EA.

As was noted in the attached information on vertebrate information provided in the Utah Natural Heritage Program, population data collected on sage grouse since the late 1960s indicate statewide population declines; population declines have been largely attributed to decreasing suitability of sagebrush steppe habitat, which has resulted in the loss and fragmentation of sage grouse habitat; impacts include increases in invasive non-native plants, particularly cheatgrass, which has resulted in dramatic changes to habitat structure and species composition in many areas; this grass is also involved in altered fire cycles and the associated conversion of large areas from shrub steppe habitat to nonnative grassland; changes to sagebrush steppe habitat are also a result of overgrazing by livestock.

There are many factors that will degrade sage grouse habitat on these 2 allotments with the addition of 12 new tanks. The Forest Plan direction for sage grouse requires that impacts of new water developments be evaluated. These impacts include the creation of 12 new severely degraded areas, roughly 125 acres each, around each new stock tank, totally 1500 acres of severely degraded sage grouse habitat. As already mentioned, water availability for sage grouse will likely significantly decrease with springs drying out. Also, with the potential addition of more cattle to these allotments, current impacts on riparian areas and wet areas from cattle will increase as well, to the detriment of sage grouse. The agency needs to address what the current trend for sage grouse is within these allotments, based on lek counts. We would like to know the lek counts for the last 20 years at a minimum. If counts are down, what are the suspected reasons? Would this indicate that grazing may be having an adverse impact, such as limiting good spring nesting cover?

Even though the project BE noted that ravens are a significant predator on sage grouse eggs and chicks, there was no acknowledgement that ravens benefit from stock tanks. The proposed 12 new stock tanks in this sage grouse breeding habitat will result in increased predation rates on sage grouse nests and chicks. This adverse impact on sage grouse in order to promote livestock grazing is a violation of the Forest Plan direction for sage grouse.

While there was no information provided in the project BE regarding the loss of nesting cover for sage grouse from cows, this is an important factor for sage grouse nesting success, which is why the Forest Plan direction includes a recommendation of at least 7 inches of residual grass cover in spring sage grouse nesting habitat. The agency needs to map all known sage grouse nesting areas in the project area, and define if these residual grass levels are being met. Given that there are a number of sage grouse leks within or next to the project area, nesting habitat clearly exists on these allotments. IN the past, has the agency met the required cover levels as per the Forest Plan in these areas? If sage grouse management is not going to be implemented as is required by the Forest Plan, the agency needs to amend the sage grouse management direction for the Dixie Forest Plan to remove management requirements, including adequate nesting cover.

Overall, the planned increase in grazing impacts in this sensitive species habitat for sage grouse will clearly be highly detrimental. Currently, the lack of water availability for cows, as was noted in the EA, has greatly restricted the distribution and thus level of grazing across this landscape. Although there has apparently been no monitoring of sage grouse nesting habitat use on these 2 allotments, one can assume that these areas of limited grazing have provided high quality sage grouse nesting habitat. With their removal, sage grouse suitable nesting habitat will also be reduced.

We would also like to know what the current levels and distribution of cheatgrass are on these 2 allotments. If livestock grazing is increased over much of these allotments, what is the expected increase in cheatgrass as well? How would an increase in cheatgrass affect sage grouse habitat quality?

It appears that the agency has been implementing various programs to improve sage grouse habitat, such as removing juniper trees. So why would the agency then implement projects (increased livestock grazing) that will be counter-productive to improving sage grouse habitat?

5. Increasing grazing with livestock on these 2 allotments will be detrimental for many landbirds, both migrants and permanent residents; the rationale for reducing habitat for these landbirds needs to be fully disclosed to the public, including how these habitat reductions adhere to the Migratory Bird Treaty Act (MBTA).

Landbirds in North America have experienced a loss of roughly 3 billion birds since the mid—1970s (Rosenberg et al. 2019); the authors warn of a faunal collapse unless these declines are addressed. For the land area affected by the 2 grazing allotments for the Showalter project, there are a potential 62 bird species characteristic of aridlands, 56.5% that are in decline. Id. The habitat in the Showalter project area is largely shrub steppe and grassland. There are a number of sensitive bird species, in addition to the sage grouse, that may occur in this project area, species that would be harmed by increases in livestock grazing. These include Birds of Conservation Concern for Region 16 (Southern Rocki89es and Colorado Plateau, or also the adjacent Great Basin region, # 9). Some species of conservation concern also include the Utah Partners in Flight Priority Species, and sensitive specie identified by the Utah Natural Heritage Program. Examples of these species and their sensitivity to grazing are provided below.

The Northern Harrier is noted to be highly sensitive to grazing (Martin and Finch 1995). This species is a BCC for the Great Basin Region.

The long-billed curlew is noted to be highly sensitive to grazing (Martin and Finch 1995).This species is a sensitive species for the state of Utah, and a Priority species for the Utah Partners in Flight.

The short-eared owl is noted to be sensitive to gazing (Martin and Finch 1995). This species is a BDD, and a sensitive species for the state of Utah.

Although grazing impacts are not clear, livestock grazing could result in trampling of burrows used by burrowing owls. This species is a sensitive species for the state of Utah.

The Showalter Project BE identifies the Brewer's sparrow as a species of concern that is present in the project area. Martin and Finch (1995) state that this species is adversely impacted by grazing.

While not a species of conservation concern, the brown-headed cowbird is strongly positively affected by grazing (Martin and Finch 1995); this species is responsible for reduction of nesting success for a significant number of western bird avifauna. Id.

Although grazing impacts are not clear, livestock grazing could result in trampling of burrows used by pygmy rabbits. This species is a sensitive species for the state of Utah, and a sensitive species for the Intermountain Region of the Forest Service. Livestock overuse and weed invasions are identified as important factors contributing to degradation of sagebrush habitat (Vertebrate Information Compiled by the Utah Natural Heritage Program).

The Utah prairie dog is a threatened species classified by the USFWS. Impacts of grazing can be adverse based on livestock removal and/or competition for crucial nutritious, succulent plants that provide moist vegetation throughout the summer; colonies without such vegetation can be decimated by drought; higher moisture content in the vegetation allow greater population density (Vertebrate Information Compiled by the Utah Natural Heritage Program).

Martin and Finch (1995) provide recommendations for livestock grazing in shrub-steppe habitats. These include first to significantly reduce or exclude livestock grazing from shrubsteppe habitat, one benefit being increased vegetation cover for protection of nest sites; these areas could be protected areas for avifauna. Second, restore perennial bunch grasses, as many species depend upon these seed resource. Third, avoid fragmentation and water developments in important habitats for species of conservation concern. Fourth, avoid conservation of shrub-steppe habitats to non-native grasses, and restore areas where this has done back to shrubsteppe habitats. Fifth, determine methods for recovering soil cryptograms to increase soil moisture and seedling germination, reduce soil erosion, and enhance productivity. And sixth, initiate long-term research to help understand the direct and indirect effects of grazing on shrubsteppe avifauna, including how livestock affects the distribution of the brown-headed cowbird.

The recommendations by Martin and Finch (1995) to remove livestock grazing from many areas of shrubsteppe habitat is consistent with the recommendations of Bock et al. (1993) to established a system of ungrazed reserves for wildlife, to benefit those wildlife species that have a low tolerance for grazing. The size of these reserves should be at least 2500 acres. Id. These areas would be roughly 20% of the that landscape that has been leased for grazing, and would be permanently set aside from grazing use by livestock.

6. The Dixie National Forest needs to include action alternatives for the 2 grazing allotments where the Showalter water project is planned, alternatives that would promote multiple use and wildlife species of conservation concern by promoting wildlife habitat within large portions (at

least acres each) the allotments, instead of managing this landscape only for private livestock.

The 2 allotments in the Showalter Project Area are clearly a perfect area to begin a progressive new management approach, whereby wildlife has at least equal value as private livestock. We believe that at least 20% or more of these 2 allotments could be removed from livestock grazing, since water availability is already limiting grazing. These removed areas would be permanently set aside for wildlife reserves. Various alternatives could include a different number and location of these reserves. The cost of these alternatives needs to be compared to the cost of increasing livestock management. These alternatives are needed in order for the agency to comply with the National Environmental Policy Act (NEPA) to develop action alternatives that address public issues.

Regards

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Mike Garrity, Director, Alliance for the Wild Rockies

Jason Christensen, Director, Yellowstone to Uintas Connection

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Diversity

Proposal for a System of Federal Livestock Exclosures on Public Rangelands in the Western United States

Of all the issues facing students and stewards of grasslands in the American West, none has proven more contentious than livestock grazing on public lands. As a partial solution to this problem, we propose establishment of a system of Federal Livestock Exclosures, whereby 20% of each parcel of land presently leased to a livestock grower would be set aside as a permanently ungrazed reserve.

North American grasslands have proven difficult to conserve, to restore, and even to understand ecologically. While most grasslands probably are as durable as any natural system, they can persist only in a rather narrow environmental window, and so they are especially vulnerable to human perturbation. Layered over the natural dynamism of grassland ecosystems are two factors that have made grassland conservation vexing. First, the major environmental forces controlling grasslands are moisture regime (drought), fire, and grazing by large mammals. These operate (or used to operate) on long temporal and large geographic scales. Consequently, it would be difficult if not impossible to create grassland preserves large enough to function absolutely as did their prehistoric counterparts. A second problem is that most North American grasslands were occupied and modified by Europeans so quickly and so long ago that they were changed before we had a chance to study them, let alone conserve them.

While a part of the debate about livestock grazing springs from the clearly different agendas of different factions, another comes from an ambiguity about the actual ecological consequences of grazing by large mammals, native or domestic. On the one hand, many grasses and grasslands evolved with large grazers (bison), and while this does not necessarily translate to any dependence of grasslands on grazing, it does speak to their potential tolerance of it. On the other hand, not all grasslands have equal evolutionary associations with large grazing mammals, and in any event the activities of fenced, predator-proofed, domestic grazers are not likely to be equivalent to those of their free-ranging endemic predecessors.

Domestic livestock today are temporally and spatially ubiquitous in many parts of the American West. This creates two problems. First, if components of the native flora and fauna are intolerant of the activities of grazing mammals (and the evidence suggests some are), these species have comparatively few places left to live. Second, the lack of large representative tracts of ungrazed grassland in many areas makes it nearly impossible to determine the actual consequences of livestock grazing. It has been an experiment largely without a control.

A Federal Livestock Exclosure (FLEX) system would serve two purposes. First, each exclosure would function as an ecological benchmark, against which the consequences of livestock grazing on that particular grazing lease or allotment could be measured objectively and irrefutably. Second, the exclosure system as a whole would provide millions of hectares of previously unavailable habitat for those plants and animals that are intolerant of the activities of large, hooved, grazing mammals.

The conceptual value of livestock exclosures has long been recognized, and many already exist. A very few are large. Most are little more than tiny ungrazed islands in a sea of cattle and sheep. Collectively they comprise only a trivial percentage of western public rangelands, and individually most are far too small to support viable populations of anything like the full complement of native grassland plants and, especially, animals.

There are at least 86 million ha (212 million acres) of federal land being grazed by domestic livestock in 17 western states. Most are managed by the U. S. Bureau of Land Management or the U. S. Forest Service. The exclosure program could be restricted to these two agencies and still create up to 15 million ha of ungrazed habitat. Spreading the exclosures across all of the thousands of existing grazing leases would result in a highly desirable mosaic of grazed and ungrazed landscape units. The minimum size for an effective exclosure probably should be about 1000 ha. Much larger units could and should be created on large allotments, while smaller allotments with common boundaries might be pooled

2500 ac

for the purpose of establishing other exclosures of minimally acceptable size. Each exclosure should be ecologically representative of its region, and each should continue to experience events such as fire and recreational use at levels typical of the allotment as a whole.

We anticipate that resistance to this proposal will take five specific forms. These five contentions, and our responses to them, are as follows:

(1) *Previous exclosure studies show that livestock do not affect or are beneficial to rangelands and their wildlife.* Most livestock exclosure studies suggest that, in fact, livestock operate as keystone species in rangeland ecosystems. They do not necessarily preclude vegetation and wildlife, but they frequently determine which species of plants and animals will thrive and which will diminish. Some exclosures have not changed much following livestock removal because (a) certain grasslands have a very tight evolutionary association with native grazers (such as arid parts of the Great Plains), (b) they have been so altered historically by domestic grazers that they cannot return to their original condition (such as desert grasslands of the Southwest), (c) insufficient time has elapsed for post-grazing changes to manifest themselves, or (d) the exclosures are too small to function as intact grassland ecosystems. This last possibility rarely has been considered.

We think that livestock are highly destructive of many components of most grassland ecosystems. Nevertheless a federal livestock exclosure system, expanded to the level we suggest, should be equally valuable for advocates as well as opponents of livestock grazing, as long as both are equally interested in the truth.

(2) *Livestock grazing is important economically.* By most accounts the percentage of American red meat produced on western public rangelands is very small (about 5–15%). A 20% reduction in that amount (1–3%) will not cause any American to be deprived of red meat, nor will it require us to import the product from foreign markets. Nevertheless, it is true that some livestock growers and their local economies would be negatively affected by a 20% loss in income. Therefore, we suggest that the following steps be included as a part of implementing the exclosure system. (a) In some cases it should be possible to increase stocking levels on the remaining 80% of the allotment without causing much ecological change. This is true because the consequences of livestock grazing appear to be all-or-nothing phenomena, unless grazing is so light as to be economically impractical, or so heavy that it completely destroys the grassland. Within the broad range of "moderate" grazing, sensitive species always are reduced, while tolerant species predominate, even if pastures are "rested" on a rotational basis. (b) Clearly there are many circumstances where an increase in stocking levels on the remaining portion of the allotment would be unacceptable. Grazing-sensitive habitats such as wetlands and riparian communities are examples. Some arid allotments may be hanging so close to the ecological brink that increased grazing would permanently change the grasslands into something else. In such cases, we would suggest substantial reductions in grazing fees, in lieu of increased stocking levels, as a means of making an exclosure more economically palatable. This may become a particularly attractive approach if, as seems likely, federal grazing fees are increased in the near future.

(3) *Exclosures would be costly to build and to maintain.* Since many allotments already are fenced into large units for implementation of livestock rotation programs, it should be possible to designate one or several of these as a livestock exclosure without cost. However, new fence construction and maintenance would be required in many instances. In no case should initial costs of exclosure construction be born by the livestock grower. However, maintenance of fences could be required of the lessee, at least where responsibilities for other similar improvements traditionally are part of the lease agreement. One strategy for initial construction would be to have materials supplied out of operating budgets of land management agencies, as these become available, with labor provided by volunteers. Local chapters of conservation organizations, hunting and fishing clubs, and outdoor youth groups would be good labor sources. We are naive neither to the skills required to build good fences, nor to the social dynamic as it presently exists among conservationists, livestock growers, and land management agencies. In our experience, however, it would be a mistake to underestimate the numbers, ability, and energy of individuals with an interest in the environment. Furthermore, this sort of cooperative activity would be a desirable alternative to the sometimes acrimonious, often unproductive, and personally remote relationship that exists today between ranchers and the general public.

(4) *Other range restoration methods work better than livestock exclusion.* This may well be true in some cases. However, most range improvement efforts, like grazing itself, have been experiments without permanently ungrazed controls. Fertilizing, bulldozing, root plowing, chaining, mowing, shredding, prescribed burning, contour furrowing, waterspreading, herbicides and pesticides, reseeding with native or exotic grasses, short-duration rotational grazing, and goat browsing, may well cause grasslands to change. These activities should never take place on the exclosures, one of whose precise purposes would be to monitor grassland condition in the absence of such intrusive manipulations.

(5) *Livestock should be removed from all public rangelands.* This position is defensible on purely ecological grounds, and it is one that plays loudly to our environmental instincts and experience. Nevertheless, it

is impractical, insensitive, and probably unnecessary to so completely disregard the human element.

Conservationists recognize that there are worse things to do to a natural ecosystem than to graze it. An arid grassland with some cattle is better than a strip mine or a suburb or a shopping mall, and these are very real threats being imposed on western landscapes and water tables by people besides ranchers, except when ranchers and developers are the same people. We have found that many ranchers have an informed appreciation for grassland plants and animals that transcends things purely economic. Such individuals make valuable allies.

At the same time, we call upon livestock growers, and particularly their lobbying groups, to be honest with themselves and with the public. It is time to stop predator control programs that most Americans find obscene, and that cannot be justified economically in any event. It is time to abandon the generalized claim that plant and wildlife species benefit from livestock grazing. Some may, but others do not. Predators *are* wildlife, and so are the birds, rodents, lizards, and insects that thrive in an ungrazed environment. Most important, it is time to acknowledge that ranching is grass farming for profit and to stop justifying it as some sort of higher environmental obligation.

A program of large federal livestock exclosures, permanently protecting 20% of public rangelands from grazing, could re-impose on the western American landscape something like the environmental mosaic formerly maintained by natural ecological forces. It would provide refuge for indigenous flora and fauna now threatened by the ubiquity of domestic grazers. It would establish an invaluable system of reference points from which to quantify the ecological consequences of grazing. Finally, and ideally, implementation of the program could begin to make allies out of groups presently contending for control of American rangelands.

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Ecology *and* Management *of* Neotropical Migratory Birds

A Synthesis and Review of Critical Issues

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ECOLOGY AND MANAGEMENT OF NEOTROPICAL MIGRATORY BIRDS

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LIVESTOCK GRAZING EFFECTS IN WESTERN NORTH AMERICA

1995

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INTRODUCTION

Livestock grazing is the most widespread economic use of public lands in western North America (Platts 1991). Approximately 86 million hectares of US Federal land in 17 western states are used for livestock production (Sabadell 1982). In the American West, grazing by domestic ungulates began in the 1840s, increased rapidly in the 1870s, and peaked around 1890 (Young and Sparks 1985). By 1900 much rangeland vegetation had been altered by the combination of extreme drought and high intensity grazing (Yensen 1981, Young and Sparks 1985). Range-management practices, including grazing systems (Appendix) and fenced pastures, were initiated in the early 1900s to help restore damaged rangelands (Behnke and Raleigh 1979). By the mid-1960s, management by allotment (designated areas for a prescribed number of livestock under one plan of management) had become an accepted practice on public lands, and is still in use today (Platts 1991).

Grazing by domestic livestock is probably the most controversial issue facing managers of public lands in the American West. This controversy is due in part to the competing economic, social, and conservation interests involved. A unique factor to grazing, as opposed to other land uses, is the fact that herbivory by native hoofed mammals has been a natural, ecological, and evolutionary force in certain nonforested ecosystems, including many in central and western North America (Stebbins 1981, McNaughton 1986). Domestic livestock has greatly intensified the influence of grazing in most of these ecosystems historically, and this influence has been especially damaging to those ecosystems where native grazing ungulates were scarce or absent (e.g., Mack and Thompson 1982, Milchunas et al. 1988, Schlesinger et al. 1990). Nonetheless, for certain habitats it is arguable that livestock grazing simulates a natural ecological event that some native flora and fauna tolerate, or perhaps require. Therefore, assertions about the effects of grazing on Neotropical migratory birds and other organisms must be habitat and species specific, and based on field data.

Birds generally do not respond to the presence of grazing livestock but to the impacts on vegetation as a result of grazing (Bock and Webb 1984). Cattle compact soil by hoof action, remove plant materials, and indirectly reduce water infiltration, all of which can result in decreased vegetation density (Holechek et al. 1989). In turn, these alterations of the structure and floristics in plant communities are known to affect some breeding bird species negatively, while other species respond positively.

Increased numbers of Brown-headed Cowbirds, created by the presence of cattle, is another indirect impact (i.e., nest parasitism) on many breeding Neotropical migratory landbirds (Robinson et al., Chapter 15, this volume). In presettlement times, cowbirds inhabited the Great Plains of central North America and were associated with giant bison herds of that region. Their range, now encompassing most of North America (American Ornithologists' Union 1983), expanded when Europeans arrived with their livestock and cleared forests (Mayfield 1965).

Cowbirds are now associated with domestic livestock, and are sufficiently numerous to pose major threats to the continued survival of several species that are regularly parasitized (Rothstein et al. 1980, 1984).

Livestock grazing is a primary land-use in four habitats important to Neotropical migrants: (1) grasslands of the Great Plains and Southwest; (2) shrubsteppe communities in the Intermountain region; (3) riparian plant communities of the arid West; and (4) montane coniferous forests. The objectives of this chapter are to evaluate the consequences of grazing by domestic ungulates on migratory landbirds using western habitats, and to provide perspectives on management as it relates to conservation of western Neotropical migrants.

METHODS

We reviewed a variety of federal publications, scientific journals, and unpublished reports for studies regarding effects of livestock grazing on landbird communities in western North America. The synthesized information is presented for Neotropical migrants in the following habitat sections: (1) grasslands, (2) shrubsteppe, (3) riparian, and (4) coniferous forests. We evaluated neotropical migrants as listed by Gauthreaux (1992), which excludes waterbirds and most shorebirds. This list includes landbirds that breed in North America and whose winter ranges predominantly extend into the Neotropics (also known as migratory landbirds).

The results of bird survey data are presented in a tabular format to facilitate comparisons between species and vegetative types. A number of important limitations exist in the information presented in the tables. Sizes and numbers of study sites, and season and intensity of grazing varied among studies. In all of the studies listed, data were obtained on the relative abundances of birds in variously grazed habitats, compared either to ungrazed or to lightly grazed sites. We listed a response as positive or negative only in those cases where the differences between treatments were >20%.

For every study, we recorded each bird species as one that increased (+), decreased (−), or was unaffected in abundance as a result of cattle grazing. In each habitat section, we provided a qualitative assessment on patterns in bird responses to grazing. In some cases where data were available, we evaluated differences in bird responses according to grazing intensity and vegetative type (grasslands), and seasonality of grazing (riparian).

When abundance data for species were recorded by two or more studies in shrubsteppe and riparian vegetation, bird responses were analyzed statistically. Abundance data were standardized to evaluate species and guilds most vulnerable to grazing disturbances. We tested the hypothesis that grazing did not affect abundances of species and ecological guilds. Standardized means were tested for differences using a paired *t*-test and derived in the following manner: $Sg = 2Ng/(Nu + Ng)$ and $Su = 2Nu/(Nu + Ng)$; where Sg = standardized mean number of individuals or pairs in a grazed treatment, Ng = number of individuals or pairs in a grazed treatment, Nu = number of individuals or pairs in an ungrazed treatment, and Su = standardized mean number of individuals or pairs in an ungrazed treatment. These proportional data were transformed with an arcsine to obtain a normal distribution. This statistical approach was not applied to grasslands because of the graded response shown by many bird species depending on grassland type.

For the guild analyses, species were categorized into groups associated with nest type (open nesting or cavity nesting; appropriate only for riparian habitats), nest location (ground, shrub, or sub canopy/canopy), and foraging behavior (insectivore, carnivore, nectarivore, or omnivore), based on characteristics described by Harrison (1979), Ehrlich et al. (1988), and Martin (1993). Finally, species were evaluated by their migratory status (Tables 12-1, 12-2, 12-4; Gauthreaux 1992). In coniferous forest vegetation, we lacked information on any bird responses to livestock grazing. Therefore, we based our conclusions on knowledge about effects of livestock on vegetation, and the known habitat requirements of the birds.

response. Vesper and Savannah Sparrows both breed widely across North America, and winter across the southern United States from California to Florida; the former appears generally unresponsive to grazing, while the latter usually has been negatively affected (Tables 12-1 to 12-4).

Breeding Bird Survey data suggest that grassland birds as a group are showing greater population declines than any other avian assemblage in North America (Robbins et al. 1993, Knopf 1994a). This probably is attributable to habitat modifications including livestock grazing, in addition to fire suppression, prairie dog control, cultivations, and planting exotic grasses.

SHRUBSTEPPE OF THE INTERMOUNTAIN REGION

Characteristics of Shrubsteppe Habitats

Shrubsteppe habitats in western North America are characterized by woody, mid-height shrubs and perennial bunchgrasses (Fautin 1946, Daubenmire 1978, Dealy et al. 1981, Tisdale and Hironaka 1981, Short 1986). Shrubsteppe typically is arid with annual precipitation over much of the region averaging less than 36 cm (Daubenmire 1956, Richard and Vaughan 1988, Rogers and Rickard 1988). Periodic drought, extreme temperatures, wind, poor soil stability and only fair soil quality (Fautin 1946, Wiens and Dyer 1975, Short 1986) manifest a stressful environment for biotic communities.

The shrubsteppe has been delineated in various ways (Kuchler 1964, Wiens and Dyer 1975, Risser et al. 1981). Major differences depend on the inclusion of salt desert shrublands of the Great Basin, shrubsteppe of the southwestern United States, or pinon-juniper types (Short 1986).

Historical Perspective and Dynamics of Shrubsteppe Habitats

Major changes in native shrubsteppe vegetation, particularly the rapid loss of forbs and grasses, took as little as 10–15 years under severe overgrazing that accompanied early settlement of the West (Kennedy and Doten 1901, Cottam and Stewart 1940, Brougham and Harris 1967, McNaughton 1979, West 1979). Some plant species may have been extirpated from the region or driven to extinction, but we assume that most of the species present today were also important historically (Dealy et al. 1981, Tisdale and Hironaka 1981).

Little doubt exists that sagebrush (*Artemisia* spp.) has always been an important component of the Intermountain landscape (Vale 1975, Braun et al. 1976), with a variety of sagebrush vegetative types dominating large areas (McArthur and Welch 1986). Other important shrubs include saltbush (*Atriplex* spp.), rabbitbrush (*Chrysothamnus* spp.), and bitterbrush (*Purshia tridentata*) (West 1979, Tisdale and Hironaka 1981, Yensen 1981, McArthur and Welch 1986). The region is characterized by perennial bunchgrasses (also known as caespitose grasses) including the genera *Agropyron*, *Poa*, *Stipa*, *Elymus*, and *Festuca* (Fautin 1946, West 1979, Yensen 1981). Few rhizomatous or sod-forming grasses occur and they play only a minor role in the ecosystem, in contrast to the prairies farther east (Mack and Thompson 1982).

Domestic livestock grazing has caused major changes in plant species composition of shrubsteppe habitats including loss of the cryptogam layer from trampling, loss of native seral grasses, reduced perennial grass cover, reduced forb cover, increased shrub cover, and invasion by exotic species, particularly cheatgrass (*Bromus tectorum*) (Yensen 1981).

Prior to European settlement, cryptogams such as the lichen Parmelia chlorochroa (MacCracken et al. 1983), covered all undisturbed soil surfaces not populated by vascular plants. Because of the permanent loss of this layer through trampling by domestic livestock (Poulton 1955, Daubenmire 1970, Mack and Thompson 1982), we do not know what role this stratum played in the original ecosystem. Increased soil temperatures, increased erosion, lower soil moisture, lower productivity and a lower rate of seedling establishment are likely negative consequences of the loss of cryptogams (MacCracken et al. 1983).

Because herbaceous species are more palatable than shrubs during the growing season, grazing tends to increase shrub cover, and decrease palatable forbs and grasses (Pickford 1932, Cottam and Stewart 1940, Smith 1967, Tisdale et al. 1969, Smith and Schmutz 1975, Page et al. 1978, Ryder 1980, Blaisdell et al. 1982). More intense grazing will eliminate even less palatable species and lead to domination by woody, unpalatable and spiny species (Ellison 1960).

Generally, cattle grazing favors shrubs and forbs over grasses while sheep grazing shifts the balance towards grass (Allred 1941, Costello and Turner 1941, Tisdale 1947, Cooper 1953, Robertson 1971, Urness 1979). Season of use is also an important influence in shrubsteppe. For example, heavy spring sheep grazing reduces grasses and increases sagebrush, whereas heavy fall sheep grazing has the opposite effect (Craddock and Forsling 1938, Mueggler 1950, Ellison 1960, Laycock 1967). Livestock grazing can also increase the density of junipers (Cottam and Stewart 1940, Woodbury 1947, Springfield 1976, Little 1977) and reduce vegetation diversity (Wiens and Dyer 1975, Reynolds and Rich 1978).

While exotic annuals are found essentially everywhere in the shrubsteppe, it is clear that their dominance increases with disturbance such as livestock grazing. Piemeisel (1938) and Young et al. (1979) submit that cheatgrass, at least, cannot significantly invade healthy shrubsteppe habitats.

Evaluation of Grazing in Shrubsteppe Habitats

Shrubsteppe habitats did not coevolve with large herds of grazing animals, and plant species are not adapted to withstand severe or continuous grazing (Mack and Thompson 1982). Post-Pleistocene native ungulates in shrubsteppe only included American bison and pronghorn. Bison numbers were estimated at 40 million when Europeans arrived (England and DeVos 1969) but it is unlikely that large herds occurred west of the Rockies (Gustafson 1972, Grayson 1977). Few prehistoric bison records exist from the Columbia Plateau (Schroedl 1973) and records are rare elsewhere in the region (Mack and Thompson 1982).

Caespitose grasses depend on seed production rather than rhizomes or stolons to maintain their populations. The effects of grazing, both removal of vegetation and mechanical damage from trampling, are more serious for caespitose species (Mack and Thompson 1982). Consequently, sagebrush–perennial bunchgrass communities are adapted to small, dispersed groups typified by pronghorn, mule deer and elk. Although these species form groups on winter ranges, they largely rely on woody vegetation at that time of year. This lack of adaptation to concentrations of large herbivores has led to "striking susceptibility" of shrubsteppe vegetation to the impact of domestic ungulates (Larson 1940, Tisdale 1961, Dyer 1979, Mack and Thompson 1982).

Classic approaches to grazing management in shrubsteppe habitats are discussed by Stoddart et al. (1975) and Laycock (1983), with novel strategies infrequent, controversial and slow to be substantiated (Savory and Parsons 1980, Savory 1988). The most noteworthy long-term trend on public land in shrubsteppe has been the reduction of destructive season-long cattle grazing where animals are released in early spring and removed in late fall (Appendix).

Multipasture rest–rotation systems (Appendix, Stoddart et al. 1975) have become popular for cattle and are a significant improvement in cattle management for shrubsteppe habitats. The rest–rotation method typically produces more uniform grazing across the landscape rather than areas of high use and areas of little or no use. The system also requires more fencing, water developments, prescribed burns, seedings or other manipulations. Ultimately, more cattle may be allowed in an allotment.

Shrubsteppe Avifauna

While more than 50 species of neotropical migrants breed in this region, the shrubsteppe bird community typically has 2–7 regular breeding species. Densities vary between 100 and 600 individuals/km^2 with over half the individuals at a site belonging to the most common species. Irregular precipitation patterns in shrubsteppe habitats have resulted in annual redistributions of individual birds locally and regionally (Wiens and Rotenberry 1981a, Wiens 1985).

Certain associations exist between bird species and particular plant species, perhaps in response to arthropod abundance or availability (Wiens and Rotenberry 1981b). Some shrubsteppe birds show a high degree of selectivity for grass seeds of certain species (Goebel and Berry 1976). Thus, selective removal of particular plant species by livestock could have direct effects on individual bird species.

Avian predators occupying shrubsteppe habitats are influenced by their small-mammal prey. Small-mammal community composition, densities and distribution vary with vegetation structure (Feldhamer 1979, Rogers and Hedlund 1980, Gano and Rickard 1982, McGee 1982) and species diversity declines as grazing intensifies (see Kochert 1989). However, the specific ecological relationships between small mammals and shrubsteppe raptors are essentially unknown.

Avian Responses to Livestock Grazing in Shrubsteppe Habitats

We found information from 15 studies that evaluated grazing effects on 34 Neotropical migrants that breed in shrubsteppe vegetation (Table 12-2). Birds considered in this evaluation range from sagebrush obligates to much more widespread species that are only peripherally associated with shrubsteppe (see references in Table 12-2). In a qualitative assessment of grazing effects on shrubsteppe birds, 12 species responded positively, 12 negatively, and 10 species showed no clear response.

The referenced studies reported abundance information on 31 migrant species and only 14 of those were evaluated by two or more studies (Table 12-3), thus limiting conclusions based on quantitative data. No species or ecological guild showed significant differences ($P < 0.05$) in abundances between grazed and ungrazed treatments. Of the 34 species evaluated, only six are considered long-distance migrants and 28 short-distance migrants (Gauthreaux 1992). Combined in groups, neither long-distance migrants [standardized means 0.81 vs 1.20 (grazed vs ungrazed), $T = 1.01$, $P = 0.33$] nor short-distance migrants [standardized means 1.00 vs 0.99 (grazed vs ungrazed), $T = -0.09$, $P = 0.92$) appeared particularly vulnerable to livestock grazing. These results should be viewed with caution considering the little quantitative data available and a lack of information about pristine shrubsteppe habitats (i.e., no controls from which to judge grazing effects).

Most studies were conducted with cattle on a short-term basis during the growing season (references in Table 12-2). Effects of other kinds of livestock (McKnight 1958, Hanley and Brady 1977) during other times of the year might differ substantially. Nevertheless, we make some tentative conclusions based on the limited published information, and knowledge about the effects of grazing on vegetation and the known habitat requirements of the birds.

Wiens and Dyer (1975) suggested that the ecological plasticity of many shrubsteppe birds would make them unresponsive to moderate levels of livestock grazing. Major avifaunal shifts may occur only after some threshold of habitat change has passed. Such thresholds may have passed historically, when livestock were first introduced into the region. However, virtually no pristine ecosystems exist where this hypothesis might be tested. As a result, our conclusions about the effects of grazing on Neotropical migrants must be largely speculative.

Distinguishing between historical and current livestock impacts is important when categorizing bird responses to grazing. For example, species requiring shrubs as nest sites may have benefitted from early, grazing-related increases in shrubs across the West. They may now be harmed by heavy grazing that removes herbaceous cover. Brewer's Sparrows may be an example, and we consider this species to be negatively affected by grazing (Tables 12-2 and 12-3).

Brewer's Sparrow populations have declined significantly both in the western United States and over their entire range during the last 25 years (Robbins et al. 1993, Peterjohn et al., chapter 1, this volume). As this species is the most typical, widespread

Table 12-2. Responses to cattle grazing by Neotropical migrant landbirds breeding in shrubsteppe habitats of western North America.

Species	Migrant Status[a]	Region	Shrubland Type	Grazing Intensity[b]	Response to Grazing	Reference
Northern Harrier	B	Nevada	Greasewood/Great Basin wild rye	Heavy	–	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds and Trost (1981)
		Oregon	Various	Variable	–	Kochert (1989)
		Idaho	Various	Variable	+	Martin (1987)
Swainson's Hawk[d]	A	Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds and Trost (1981)
		Oregon	Various	Heavy	–	Littlefield et al. (1984)
		West	Various	Variable	–	Kochert (1989)
Red-tailed Hawk[d]	B	Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds and Trost (1981)
		West	Various	Variable	Mixed	Kochert (1989)
Ferruginous Hawk	B	Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds and Trost (1981)
		West	Various	Variable	Mixed	Kochert (1989)
Golden Eagle	B	Idaho	Various	Heavy	–	Kochert (1989)
		Idaho	Big sage	Variable	+	Nydegger and Smith (1986)
American Kestrel[d]	B	Nevada	Low sage/Idaho fescue	Heavy	–	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds and Trost (1981)
		West	Various	Variable	–	Kochert (1989)
Prairie Falcon	B	Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds and Trost (1981)
Long-billed Curlew	A	Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds and Trost (1981)
Mourning Dove	B	Nevada	Greasewood/Great Basin wild rye	Heavy	+	Page et al. (1978)
		Nevada	Shadscale/Indian ricegrass	Heavy	–	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds (1980)
		West	Various	Variable	+	Kochert (1989)
Burrowing Owl	A	West	Various	Variable	+	Snyder and Snyder (1975)
		Idaho	Big sage/bluebunch wheatgrass	Variable	Mixed	Rich (1986)
		Idaho	Big sage	Variable	–	Gleason (1978)
Long-eared Owl	B	West	Various	Variable	Mixed	Kochert (1989)
Short-eared Owl	B	Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds (1980)
Common Nighthawk	A	Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds and Trost (1981)
Common Poorwill	B	Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds and Trost (1981)
Northern Flicker[e]	B	Nevada	Low sage/Idaho fescue	Heavy	0	Page et al. (1978)
		Nevada	Big sage/bluebunch wheatgrass	Heavy	+	Page et al. (1978)
Gray Flycatcher	A	Nevada	Shadscale/Indian ricegrass	Heavy	+	Page et al. (1978)
		Nevada	Nevada bluegrass/sedge	Heavy	+	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds and Trost (1981)
Say's Phoebe	B	Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds and Trost (1981)
Horned Lark	B	Nevada	Greasewood/Great Basin wild rye	Heavy	–	Page et al. (1978)

(continued)

Table 12-2 (*cont.*)

Species	Migrant Status[a]	Region	Shrubland Type	Grazing Intensity[b]	Response to Grazing[c]	Reference
Horned Lark (*cont.*)		Nevada	Shadscale/Indian ricegrass	Heavy	–	Page et al. (1978)
		Nevada	Low sage/Idaho fescue	Heavy	–	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds (1980)
		Utah	Shadscale/sand dropseed	Heavy	Mixed	Medin (1986)
		Idaho	Big sage	Not reported	+	Olson (1974)
		Idaho	Various	Variable	+	Rotenberry and Knick (1992)
Tree Swallow[d]	B	Nevada	Nevada bluegrass/sedge	Heavy	–	Page et al. (1978)
Rock Wren	B	Nevada	Shadscale/Indian Ricegrass	Heavy	+	Page et al. (1978)
		Nevada	Big sage/bluebunch wheatgrass	Heavy	+	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds and Trost (1981)
Mountain Bluebird	B	Nevada	Low sage/Idaho fescue	Heavy	+	Page et al. (1978)
Sage Thras	B	Nevada	Greasewood/Great Basin wild rye	Heavy	+	Page et al. (1978)
		Nevada	Nevada bluegrass/sedge	Heavy	+	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds (1980)
		Idaho	Big sage	Not reported	+	Olson (1974)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds and Rich (1978)
Loggerhead Shrike	B	Nevada	Shadscale/Indian ricegrass	Heavy	+	Page et al. (1978)
		Nevada	Low sage/Idaho fescue	Heavy	+	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	0	Reynolds (1980)
		Utah	Shadscale/sand dropseed	Heavy	0	Medin (1986)
Green-tailed Towhee	A	Nevada	Big sage/bluebunch wheatgrass	Heavy	–	Page et al. (1978)
Vesper Sparrow	B	Nevada	Greasewood/Great Basin wild rye	Heavy	+	Page et al. (1978)
		Nevada	Shadscale/Indian ricegrass	Heavy	+	Page et al. (1978)
		Nevada	Low sage/Idaho fescue	Heavy	–	Page et al. (1978)
		Nevada	Big sage/bluebunch wheatgrass	Heavy	–	Page et al. (1978)
		Nevada	Nevada bluegrass/sedge	Heavy	–	Page et al. (1978)
		Idaho	Big sage	Not reported	–	Olson (1974)
Black-throated Sparrow	B	Utah	Shadscale/sand dropseed	Heavy	Mixed	Medin (1986)
Sage Sparrow	B	Nevada	Greasewood/Great Basin wild rye	Heavy	–	Page et al. (1978)
		Nevada	Shadscale/Indian ricegrass	Heavy	–	Page et al. (1978)
		Nevada	Nevada bluegrass/sedge	Heavy	+	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	–	Reynolds (1980)

Table 12-2 (*cont.*)

Species	Migrant Status[a]	Region	Shrubland Type	Grazing Intensity[b]	Response to Grazing[c]	Reference
Sage Sparrow (*cont.*)		Idaho	Big sage	Not reported	+	Olson (1974)
		Great Basin	Big Sage	Variable	+	Wiens and Rotenberry (1981b), Wiens (1985)
Brewer's Sparrow	B	Nevada	Greasewood/Great Basin wild rye	Heavy	−	Page et al. (1978)
		Nevada	Shadscale/Indian ricegrass	Heavy	+	Page et al. (1978)
		Nevada	Low sage/Idaho fescue	Heavy	−	Page et al. (1978)
		Nevada	Big sage/bluebunch wheatgrass	Heavy	+	Page et al. (1978)
		Nevada	Nevada bluegrass/sedge	Heavy	+	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	−	Reynolds (1980)
		Idaho	Big sage	Not reported		Olson (1974)
Savannah Sparrow	B	Nevada	Nevada bluegrass/sedge	Heavy		Page et al. (1978)
White-crowned Sparrow	B	Nevada	Greasewood/Great Basin wild rye	Heavy		Page et al. (1978)
		Nevada	Big sage/bluebunch wheatgrass	Heavy	+	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds and Trost (1981)
Red-winged Blackbird	B	Nevada	Nevada bluegrass/sedge	Heavy	−	Page et al. (1978)
Western Meadowlark	B	Nevada	Greasewood/Great Basin wild rye	Heavy	+	Page et al. (1978)
		Nevada	Shadscale/Indian ricegrass	Heavy	+	Page et al. (1978)
		Nevada	Low sage/Idaho fescue	Heavy	−	Page et al. (1978)
		Nevada	Big sage/bluebunch wheatgrass	Heavy	+	Page et al. (1978)
		Nevada	Nevada bluegrass/sedge	Heavy	−	Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	−	Reynolds (1980)
		Idaho	Big sage	Not reported	−	Olson (1974)
		Idaho	Various	Variable	0	Rotenberry and Knick (1992)
		Great Basin	Big sage	Variable	−	Wiens and Rotenberry (1981b)
Brewer's Blackbird	B	Nevada	Shadscale/Indian ricegrass	Heavy	+	Page et al. (1978)
		Nevada	Nevada bluegrass/sedge	Heavy		Page et al. (1978)
		Idaho	Big sage/bluebunch wheatgrass	Moderate	0	Reynolds and Trost (1981)
Brown-headed Cowbird	B	Idaho	Big sage/bluebunch wheatgrass	Moderate	+	Reynolds and Trost (1981)
		Great Basin	Various	Variable	+	Rich and Rothstein (1985)

[a] Status "A" contains long-distance migrants, those species that breed in North America and spend their nonbreeding period primarily south of the United States. Status "B" contains short-distance migrants, those species that breed and winter extensively in North America, although some populations winter south of the United States.
[b] Grazing intensity as reported by original authors in the references listed.
[c] Grazing effects on abundance: + = increase; − = decrease; 0 = no effect, as reported by original authors.
[d] Species forages in shrubsteppe vegetation but nests in other adjacent habitat.

Table 12-3. Standardized relative abundance for 31 species of migratory landbirds. Original data were taken from 15 studies conducted in shrubsteppe habitats. Sample size is the number of studies from which the data were derived. Standard errors (SE) and P values were calculated by a paired t-test. No species' abundances differed significantly ($P < 0.05$) between treatments.

Species	Forage Guild[a]	Nest Layer[b]	Nest Type[c]	Sample Size	Standardized Mean: Grazed	Standardized Mean: Ungrazed[d]	SE	P Value
Northern Harrier	CA	GR	O	2	0.60	1.40	1.77	0.58
Swainson's Hawk	CA	CA	O	1	1.67	0.33	—	—
Red-tailed Hawk	CA	CA	O	1	0.67	1.33	—	—
Ferruginous Hawk	CA	GR	O	1	0.67	1.33	—	—
American Kestrel	CA	CA	C	2	0.80	1.20	2.21	0.75
Prairie Falcon	CA	CA	C	1	0.00	2.00	—	—
Long-billed Curlew	GI	GR	O	1	0.00	2.00	—	—
Mourning Dove	GI	SH	O	3	0.78	1.22	1.88	0.82
Short-eared Owl	CA	GR	C	1	0.00	2.00	—	—
Common Nighthawk	AI	GR	O	1	0.67	1.33	—	—
Common Poorwill	AI	GR	O	1	2.00	0.00	—	—
Northern Flicker	BI	CA	C	2	1.50	0.50	1.57	0.50
Gray Flycatcher	AI	SH	O	3	1.33	0.67	2.09	0.67
Say's Phoebe	AI	CA	O	1	2.00	0.00	—	—
Horned Lark	GI	GR	O	5	0.42	1.58	0.71	0.07
Tree Swallow	AI	CA	C	1	0.00	2.00	—	—
Rock Wren	GI	GR	C	3	2.00	0.00	0	0
Mountain Bluebird	AI	CA	C	4	2.00	0.00	—	—
Sage Thrasher	FI	SH	O	3	1.76	0.24	0.86	0.11
Loggerhead Shrike	SA	SH	O	4	1.42	0.58	1.01	0.26
Green-tailed Towhee	GI	GR	O	1	0.00	2.00	—	—
Vesper Sparrow	GI	GR	O	5	0.80	1.20	1.54	0.70
Black-throated Sparrow	FI	SH	O	1	1.23	0.77	—	—
Sage Sparrow	GI	SH	O	4	0.93	1.07	0.37	0.69
Brewer's Sparrow	FI	SH	O	6	0.83	1.17	0.69	0.45
Savannah Sparrow	GI	GR	O	1	0.00	2.00	—	—
White-crowned Sparrow	OM	GR	O	3	1.12	0.88	1.08	0.87
Red-winged Blackbird	OM	SH	O	1	0.00	2.00	—	—
Western Meadowlark	GI	GR	O	6	1.00	1.00	1.22	1.00
Brewer's Blackbird	OM	SH	O	3	0.54	1.46	0.63	0.24
Brown-headed Cowbird	OM	—	P	1	1.78	0.22	—	—

[a] Foraging-guild abbreviations: AI = aerial insectivore; BI = bark insectivore; FI = foliage insectivore; GI = ground insectivore; CA = carnivore; NE = nectarivore; OM = omnivore.
[b] Nest-layer abbreviations: SH = shrub-nesting species; GR = ground-nesting species; CA = subcanopy/canopy-nesting species.
[c] Nest-type abbreviations: O = open; C = cavity; P = parasite.
[d] Includes lightly grazed and fall-grazed treatments.

and common shrubsteppe bird in many locations, their decline is a major cause for concern in sagebrush ecosystems.

Other shrubsteppe species such as Gray Flycatcher, Rock Wren, Green-tailed Towhee, Sage Thrasher, and Lark and Sage Sparrows have shown no significant population trends over the western United States (Peterjohn et al., Chapter 1, this volume). However, Lark Sparrows and Rock Wrens show significant range-wide declines. Black-throated Sparrows, which inhabit more xeric shrub communities, have also shown significant population declines.

Data for two other species also suggest that shrubsteppe bird communities are changing, whether or not livestock grazing is implicated as a major cause. Long-billed Curlews and Burrowing Owls both breed in habitats characterized by a lack of shrubs and large areas of relatively low vegetation. Both species showed significant population increases between 1966 and 1991 in the western United States (Peterjohn et al., Chapter 1, this volume).

Little information exists on responses to grazing by migratory raptors in shrubsteppe (Kochert 1989). Our designations of raptor species increasing or decreasing in abundance were based on grazing-induced habitat alterations, which affect small mammal populations, nest cover and substrates. Intensive grazing and fire suppression favors encroachment by shrubs and trees (especially *Juniperus* spp.). Ground-nesting raptors (e.g., Northern Harrier, Short-eared Owl, and Ferruginous Hawk) are often negatively affected by grazing practices that reduce nest cover (Duebbert and Lokemoen 1977). Raptors and their rodent prey often decrease under conditions with reduced amounts of herbaceous cover and increased shrub densities (see Kochert 1989). Other prey species (e.g., jackrabbits) respond positively to dense shrub conditions (Nydegger and Smith 1986), potentially benefitting their primary predator, the Golden Eagle. Increases in juniper trees could increase availability of nest sites (e.g., Long-eared Owls and Red-tailed Hawks) (Kochert 1989) and perch sites of some raptor species. Another potentially significant indirect effect of grazing on migrants in shrubsteppe is nest parasitism by Brown-headed Cowbirds. However, almost no data are available for shrubsteppe (Rich 1978, Rich and Rothstein 1985) and the degree of impact caused by cowbirds is unknown.

Shrubsteppe birds generally respond negatively after deliberate conversions of native shrub habitats to exotic vegetation for the foraging benefit of livestock (Best 1972, Schroeder and Sturges 1975, Reynolds and Trost 1980, 1981, Castrale 1982). However, some responses may not be detected due to lack of clear population responses due to time lags, site tenacity by individuals and scale of treatment (Wiens and Rotenberry 1985). Thus, short-term before and after surveys in this avian community may be "dangerously misleading" (Wiens et al. 1986).

WESTERN RIPARIAN HABITATS

Characteristics of Riparian Habitats

Riparian zones include assemblages of plant and animal communities occurring at the interfaces between terrestrial and aquatic ecosystems. In arid portions of western North America, riparian areas create well-defined, narrow zones of vegetation along ephemeral, intermittent, and perennial streams and rivers, and are most conspicuous in steppe, shrubsteppe, and desert regions. The diversity and productivity of these systems compared to surrounding uplands are largely attributable to biotic and nutrient exchanges between aquatic and adjacent upland areas (Gregory et al. 1991).

Western riparian woodlands vary from extensive floodplain forests dominated by cottonwoods (*Populus* spp.) along large rivers to narrow bands of aspen (*Populus tremuloides*) woodlands and willow (*Salix* spp.) thickets along small mountain streams. Plant composition varies geographically and climatically, with higher elevation areas often composed of alder (*Alnus* spp.), birch (*Betula* spp.), and dogwood (*Cornus* spp.). Sycamore (*Platanus* spp.), cherry (*Prunus* spp.), hawthorn (*Crataegus* spp.), and hackberry (*Celtis* spp.) are typically found at lower elevations and in drier climates.

Historical Perspective of Riparian Habitats

The critical and disproportionate value of riparian habitat to wildlife has been recognized only within the last two decades (Johnson et al. 1977, Knopf et al. 1988a). Riparian vegetation is used by wildlife more than any other vegetation type (Thomas et al. 1979). Yet, riparian areas are among the most threatened habitats on the continent because they are favored for many land uses including livestock grazing, agriculture, water management, timber harvest, recreation, and urbanization (e.g., Thomas et al. 1979, Knopf et al. 1988a).

Livestock grazing has caused geographically extensive impacts on western riparian zones (Carothers 1977, Crumpacker 1984, Chaney et al. 1990), and these areas are considered the most modified land type in the West (Chaney et al. 1990). Grazing on riparian bottomlands tends to be more damaging than on uplands (Platts and Nelson 1985), especially in arid regions where water, shade, succulent vegetation, and flatter terrain occur near streams (Behnke 1979, Chaney et al. 1990, Platts 1991). Livestock grazing affects riparian habitats by altering, reducing, or removing vegetation, and by actually eliminating riparian areas through channel widening, channel aggrading, or lowering the water table (see Platts 1991).

Evaluation of Grazing Systems in Riparian Habitats

Rangeland grazing practices have been reviewed and evaluated for riparian ecosystems (Platts 1981, Knopf and Cannon 1982, Kauffman et al. 1983, Kauffman and Krueger 1984, Skovlin 1984, Clary and Webster 1989, Platts 1991, Sedgwick and Knopf 1991, Kovalchik and Elmore 1992). Riparian habitats are known to be detrimentally affected by most grazing practices tested to date. This is not surprising because traditional grazing systems were developed for upland grasses, not for riparian plant species (see Platts 1991 for review).

Grazing systems are evaluated by the intensity and seasonality of use by livestock. Riparian areas generally are grazed most in summer and least in winter (Knopf et al. 1988b, Goodman et al. 1989). The resulting summer concentration of use in riparian zones is particularly damaging due to severe trampling and mechanical damage, soil compaction, and plant consumption by livestock. Thus, year-long and growing-season (spring–summer) grazing are particularly damaging to riparian vegetation (Kauffman and Krueger 1984, Platts 1991), and the associated bird communities (Crouch 1982).

Short-term, early spring grazing may be preferable to summer grazing (Clary and Webster 1989). Early season grazing can result in better distribution of livestock because upland vegetation is succulent at this time and because livestock may avoid the wetter riparian soils (Clary and Webster 1989, Platts 1991). However, impacts of soil compaction may be most severe at this time. Early season grazing, followed by complete removal of livestock, allows regrowth of riparian vegetation before the dormant period in autumn.

As herbaceous cover is depleted or as palatability of alternate forage decreases, livestock will shift to browsing riparian shrubs before leaf drop (Kovalchik and Elmore 1992). Therefore, most browsing damage to willows (*Salix* spp.) occurs in late summer and fall (Kauffman et al. 1983, Clary and Webster 1989, Sedgwick and Knopf 1991, Kovalchik and Elmore 1992). Alternatively, light-to-moderate autumn grazing appears to have the least impact on numbers of migratory birds during the breeding season (Kauffman et al. 1982, 1983, Sedgwick and Knopf 1987, Knopf et al. 1988b, Medin and Clary 1991).

In late fall and winter, water levels typically are low, streambanks are dry, and vegetation is dormant, thus minimizing the effects of trampling, soil compaction, erosion, and browsing (Rauzi and Hanson 1966, Knopf and Cannon 1982, Kauffman and Krueger 1984). However, fall–winter grazing should be carefully controlled to leave residual plant cover needed for streambank maintenance during subsequent high spring flows (Clary and Webster 1989).

Kauffman et al. (1982, 1983) evaluated the effects of late season grazing on ten common riparian communities in eastern Oregon by comparing plant and animal communities in enclosed and grazed areas (late August–mid-September, at 1.3–1.7 ha/animal unit month). Avian populations in all plant communities appeared to have little differential response to grazing treatments with respect to species richness, density, or diversity. Meadows and Douglas hawthorn (*Crataegus douglasii*) communities were more heavily used by cattle than other riparian communities, shrub use was light except on willow-dominated gravel bars, and use of plant communities with dense canopy cover (black cottonwood [*Populus trichocarpa*], Ponderosa pine [*Pinus ponderosa*], and thin-leafed alder [*Alnus incana*]) was light.

Sedgwick and Knopf (1987) evaluated the impact of fall (October–November) grazing on breeding densities of six Neotropical migrants (House Wren, Brown Thrasher, American Robin, Common Yellowthroat, Yellow-breasted Chat, and Rufous-sided Towhee) associated with the lower vegetative layer of a cottonwood (*P. sargentii*) riparian forest. Moderate, late-fall grazing had no apparent impact on densities of any of the species, implying that proper seasonal grazing of cottonwood bottomlands is compatible with migratory bird use of a site during the breeding season. Common Yellowthroats and Yellow-breasted Chats were the most individual in their habitat associations and most likely to respond negatively to higher levels of grazing.

Knopf et al. (1988b) compared plant and bird communities between healthy (historically winter grazed) and decadent (historically summer grazed) willow communities within a year. Population densities of habitat generalists (Yellow Warblers, Savannah Sparrows, and Song Sparrows) differed little between winter-grazed and summer-grazed willow communities. Densities of the species intermediate in habitat specialization (American Robins, Red-winged Blackbirds, and Brown-headed Cowbirds) differed more dramatically, while habitat specialists (Willow Flycatchers, Lincoln's Sparrows, and White-crowned Sparrows) were absent or accidental in decadent willows. Brown-headed Cowbirds showed the greatest tendency to increase in numbers in disturbed, summer-grazed riparian areas. Conversely, high local densities of habitat specialists (and possibly Red-winged Blackbirds) occurred in winter-grazed willow communities.

With prescribed, late-season grazing in a cottonwood floodplain in Colorado, herbaceous and shrub vegetation (excluding willows) appeared to be resilient to cattle grazing, at least during the initial 3 years after grazing began, following 31 years of nonuse (Sedgwick and Knopf 1991). The grazing program for this study was strictly controlled by season and intensity of use within the riparian zone. This is unlike most grazing programs, wherein the riparian zone is included as part of a larger allotment and the use of riparian vegetation often exceeds forage use on the uplands (Platts 1991).

Sedgwick and Knopf (1991) cautioned that even a 4 year study is a relatively brief time to study grazing impacts. Longer-term grazing effects may alter composition, structural diversity, and community succession patterns in riparian systems. For example, they were unable to assess grazing impacts on cottonwood seedling survival because seedlings were so few on their study area. Glinski (1977), however, found that cattle grazing reduced cottonwood-seedling establishment along an Arizona stream, and predicted that the future width of the riparian zone would be significantly reduced. Longer term (more than 3 years) studies of dormant-season grazing may well document alterations of herbaceous communities (Sedgwick and Knopf 1991).

Riparian Avifauna

Western riparian areas are key components of migratory bird habitats during all seasons of the year (Stevens et al. 1977, Henke and Stone 1979, Szaro 1980, Terborgh 1989). Riparian vegetation covers less than 1% of the landscape in the arid West, yet more species of breeding birds are found in this limited habitat than in the more extensive surrounding uplands (Knopf et al. 1988a). Migratory landbirds inhabiting western North America are thought to be particularly vulnerable to disturbance because their riparian habitats are fragmented and limited in distribution, thus probably restricting their total populations below those of their eastern counterparts (Terborgh 1989). Because the contribution of these productive areas to avian diversity is disproportionate to other western habitats, riparian woodlands are critical to overall conservation of the continental avifauna.

The highest densities of breeding birds for North America have been reported from southwestern riparian habitats (Carothers and Johnson 1975, Ohmart and Anderson 1986, Rice et al. 1983). More than two-thirds (127 of 166) of southwestern bird species nest in riparian woodlands, and Neotropical migrants comprise 60% of the 98 landbirds (Johnson et al. 1977). In arid portions of the West, several studies documented that most bird species nest in riparian habitats where Neotropical migrants comprise between 60% and 85% of the landbirds (Knopf 1985, Dobkin and Wilcox 1986, Saab and Groves 1992). Probably most migrant landbirds in the western United States are associated with riparian habitats during the breeding season (cf. Mosconi and Hutto 1982, Ohmart and Anderson 1986).

Avian Responses to Livestock Grazing in Riparian Habitats

We know of nine studies that provide some quantitative comparisons of species abundances in systems that were variously

Table 12-4 (*cont.*)

Species	Migrant Status[a]	Region	Riparian Type	Grazing Intensity[b]	Response to Grazing[c]	Reference
Brewer's Blackbird (*cont.*)		Colorado	Cottonwood/willow	Variable	–	Crouch (1982)
		Oregon	Willow	Variable	0	Taylor (1986)
		Idaho	Herbaceous	Heavy	+	Medin and Clary (1990)
Brown-headed Cowbird	B	California/Nevada	Aspen	Not reported	+	Page et al. (1978)
		Montana	Cottonwood/pine	Heavy vs light	+	Mosconi and Hutto (1982)
		Oregon	Willow	Variable	–	Taylor (1986)
		Colorado	Willow	Variable	+	Knopf et al. (1988b)
		Colorado	Willow	Heavy	+	Schulz and Leininger (1991)
Northern Oriole	A	California/Nevada	Aspen	Not reported	+	Page et al. (1978)
		Colorado	Cottonwood/willow	Variable	–	Crouch (1982)
		Montana	Cottonwood/pine	Heavy vs light	–*	Mosconi and Hutto (1982)
		Oregon	Willow	Variable	–	Taylor (1986)
		Nevada	Aspen/willow	Moderate	0	Medin and Clary (1991)
Hooded Oriole	A	California/Nevada	Aspen	Not reported	+	Page et al. (1978)
Western Tanager	A	Montana	Cottonwood/pine	Heavy vs light	+*	Mosconi and Hutto (1982)
		Colorado	Willow	Heavy	–	Schulz and Leininger (1991
Pine Siskin	B	Montana	Cottonwood/pine	Heavy vs light	+	Mosconi and Hutto (1982)
		Colorado	Willow	Heavy	+	Schulz and Leininger (1991
American Goldfinch	B	Colorado	Cottonwood/willow	Variable	–	Crouch (1982)
		Montana	Cottonwood/pine	Heavy vs light	–	Mosconi and Hutto (1982)
		Oregon	Willow	Variable	–	Taylor (1986)
Cassin's Finch	B	California/Nevada	Aspen	Not reported	–	Page et al. (1978)
		Montana	Cottonwood/pine	Heavy vs light	+	Mosconi and Hutto (1982
		Oregon	Willow	Variable	–	Taylor (1986)
		Nevada	Aspen/willow	Moderate	0	Medin and Clary (1991)
		Colorado	Willow	Heavy	–	Schulz and Leininger (19

[a] Status "A" contains long-distance migrants, those species that breed in North America and spend their nonbreeding period prim south of the United States. Status "B" contains short-distance migrants, those species that breed and winter extensively in North Ame although some populations winter south of the United States.
[b] Grazing intensity as reported by original authors in the references listed.
[c] Grazing effects on abundance: + = increase; – = decrease; 0 = no effect, as reported by original authors. Species whose abund differed statistically between treatments, as reported by original authors are indicated by an asterisk.

grazed by cattle (Table 12-4). These studies were conducted in six states and most were in cottonwood and willow riparian communities. The studies described the impacts of grazing by comparing avian populations on adjacent grazed and ungrazed sites (Page et al. 1978; Crouch 1982, Sedgwick and Knopf 1987, Medin and Clary 1990, 1991, Schulz and Leininger 1991), on adjacent sites that were subject to different levels of grazing (Mosconi and Hutto 1982, Taylor 1986), and on adja sites historically grazed during diffe seasons of the year (Knopf et al. 198 Season and intensity of grazing were always well defined, and the results four of the studies (i.e., Page et al. 1 Mosconi and Hutto 1982; Medin and C 1990, 1991) are compromised by the c plete absence of treatment replication evaluate the effects of grazing. De shortcomings, we generally found consi patterns and biologically interpretable responses by many members of the riparian avifauna.

These studies reported abundance data on 68 species of Neotropical migrant landbirds (Table 12-4). In a qualitative assessment of all studies combined, nearly half (46%) of these species decreased in abundance with cattle grazing, 29% increased with grazing, and 25% showed no clear response.

Forty-three of the 68 species were evaluated by two or more studies, and used in statistical analyses (see Methods section). Among these, few species showed significant differences ($P < 0.05$) in abundance between grazed and ungrazed treatments (Table 12-5). Species with significant or near-significant reductions in grazed treatments included Red-winged Blackbirds, Common Yellowthroats, and Willow Flycatchers. These species were about 1.5 times more abundant in ungrazed treatments (Table 12-5), indicating that these species are sensitive to changes resulting from livestock grazing. They also experience high rates of nest parasitism (Hofslund 1957, Sedgwick and Knopf 1988, Weatherhead 1989). All three species nest within the shrub layer and, in forested habitats, songbirds that nest in shrubs generally experience the highest rates of nest predation (Martin 1993). Cattle may further increase nest losses by exposing concealed nests to predators by reducing foliage densities or opening dense patches of vegetation to allow predator access (Knopf 1995).

At least seven more species were probably also harmed by grazing in riparian ecosystems. Three of these were evaluated in only one of the nine studies, but showed strongly negative responses: Veery, Nashville Warbler, and Fox Sparrow; all are ground or near-ground nesters. Other species showed uncertain or inconsistent responses to grazing, but likely would be negatively affected by grazing, based on knowledge of their habitat requirements. Conspicuous among these are the Yellow Warbler (see Taylor and Littlefield 1986), American Redstart, Gray Catbird, and Yellow-breasted Chat. Of these species with limited data but expected to decrease with grazing, only the Veery is experiencing population declines in the West and significant continental declines (Robbins et al. 1993, Peterjohn et al., chapter 1, this volume). Cattle grazing could be one factor contributing to their population declines.

American Robins, Killdeer, and Pine Siskins, all habitat generalists, showed the strongest trend of increasing in abundance with grazing. American Robins and Killdeer appear well adapted to human-modified landscapes in the West, e.g., both commonly nest in residential areas and prefer relatively open habitats (DeGraff et al. 1991, Dobkin 1993). Although Pine Siskins generally prefer coniferous habitats, some are found nesting in western riparian woodlands adjacent to pine forests (Mosconi and Hutto 1982, Schulz and Leininger 1991). This species nests in tree canopies, generally unaffected by livestock grazing in the short term.

Grouping species by nest type, we found that cavity-nesting species appeared least affected by cattle grazing [standardized mean 1.02 vs 0.97 ± 0.35 (grazed vs ungrazed), $T = 0.021$, $P = 0.98$]. Although not significant, abundance of open-nesting birds was more reduced by grazing practices [standardized mean 0.89 vs 1.11 ± 0.17 (grazed vs ungrazed), $T = -1.55$, $P = 0.12$]. These results support those of individual studies evaluating short-term grazing effects, in concluding that woodpeckers and other cavity-nesting species are relatively unaffected (Good and Dambach 1943, Mosconi and Hutto 1982) and sometimes increase in grazed pastures (Butler 1979, Medin and Clary 1991). Cavity-nesting birds place their nests in snags and dead limbs, and frequently forage in tree locations (bark) that are generally not used by cattle. Open-nesting species generally experience lower rates of nest success than cavity-nesting species (Martin and Li 1992), and cattle could further increase nest losses through physical damage to the herbaceous and shrub layers where open-nesting species often nest and forage.

Evaluating species by nest location, we found that ground-nesting species were most susceptible to disturbances created by livestock grazing (Table 12-6). Dark-eyed Juncos, and White-crowned, Savannah,

Table 12-5 (*cont.*)

Species	Forage Guild[a]	Nest Layer[b]	Nest Type[c]	Sample Size	Standardized Mean		SE	P Value
					Grazed	Ungrazed[d]		
Western Meadowlark	GI	GR	O	2	1.01	0.99	0.01	0.20
Yellow-headed Blackbird	OM	SH	O	1	1.18	0.82		
Red-winged Blackbird	OM	SH	O	4	0.46	1.54	0.35	0.4*
Brewer's Blackbird	OM	SH	O	4	1.31	0.70	1.06	0.43
Brown-headed Cowbird	GI		P	5	1.39	0.61	0.83	0.20
Northern Oriole	OM	CA	O	5	0.83	1.17	1.01	0.74
Hooded Oriole	OM	CA	O	1	2.00	0.00		
Western Tanager	FI	CA	O	2	1.34	0.66	1.91	0.63
Pine Siskin	OM	CA	O	2	1.96	0.04	0.40	0.09
American Goldfinch	OM	SH	O	3	0.61	1.39	0.30	0.11
Cassin's Finch	OM	CA	O	5	0.75	1.25	1.17	0.57

[a] Foraging-guild abbreviations: AI = aerial insectivore; BI = bark insectivore; FI = foliage insectivore; GI = ground insectivore; CA = carnivore; NE = nectivore; OM = omnivore.
[b] Nest-layer abbreviations: SH = shrub-nesting species; GR = ground-nesting species; CA = subcanopy canopy-nesting species.
[c] Nest-type abbreviations: O = open; C = cavity; P = parasite.
[d] Includes lightly grazed and fall-grazed treatments.

and Lincoln's Sparrows were ground-nesting species that experienced the greatest reductions in grazed areas. These ground nesters are also dependent on the grass–forb–shrub layer for foraging, making them particularly vulnerable to grazing disturbances (Sedgwick and Knopf 1987).

Canopy-nesting birds were least affected in the short term (Table 12-6). These data support other studies indicating that birds are impacted most by habitat perturbations in the vegetative zone in which they occur (Short and Burnham 1982; Verner 1984). Knopf (1995) noted that in the vertical plain, livestock grazing has little direct impact on birds nesting and foraging in forest canopies. However, cattle trampling and browsing of young trees can limit the number of trees that reach maturity, thus reducing future canopy layers.

Grazing had a differential effect on avian foraging guilds (Table 12-7). Aerial and bark insectivores were probably not greatly affected. Aerial insectivores do not rely upon vegetation for feeding substrates and bark insectivores exploit a substrate generally not used by cattle. In contrast, species dependent upon food resources produced directly (nectarivores) or indirectly (omnivores) by understory plants were less represented in grazed compared to ungrazed treatments. Similar responses to grazing by these guilds were observed on tropical wintering grounds of migratory landbirds (Saab and Petit 1992). Local reductions in nectarivores and omnivores could have widespread ramifications, because these species are important pollinators and seed dispersers, respectively (Feinsinger 1983; Herrera 1984).

Table 12-6. Standardized relative abundances for ground-, shrub-, and canopy-nesting birds in grazed and ungrazed habitats. Sample size is the number of occurrences in which a nest layer was represented in each treatment. Original data were taken from nine studies conducted in western riparian habitats (see Table 12-1). Cavity-nesting species were placed in the canopy layer and Brown-headed Cowbirds were excluded from the analysis. Groups of species in each nest layer whose abundances differ significantly (paired *t*-test, $P < 0.05$) between treatments are indicated by an asterisk.

Nest Layer	Sample Size	Standardized Mean		SE	P Value
		Grazed	Ungrazed[a]		
Ground	33	0.71	1.29	0.32	0.01*
Shrub	48	0.86	1.14	0.29	0.33
Subcanopy canopy	86	1.02	0.98	0.27	0.78

[a] Includes lightly grazed and fall-grazed treatments.

vegetative type, and timing and duration of grazing.

Because of differential responses by Neotropical migrants to grazing, management programs for single species would be difficult to develop and generally cost prohibitive (Knopf 1995). Management of single species would become necessary only when they become species of special concern and only for the time it takes for species recovery (Hejl et al., Chapter 8, this volume).

An alternative to single species management would be to manage for "ecological guilds." Whereas patterns of bird responses emerged using this approach, at least in riparian habitats, it would be difficult to apply in all situations and would have to be implemented on a habitat specific basis. For example, in grasslands, several species showed a graded response depending on the grassland vegetative type and grazing intensity. Some species (e.g., Upland Sandpiper, Grasshopper Sparrow, and Bobolink) tended to be negatively affected in shorter grasslands (at least with heavy grazing), and responded positively to grazing in taller grasslands (at least with moderate grazing). Thus, we still need a management approach that considers individual species, but one implemented with habitat monitoring on a landscape level (Hejl et al., Chapter 8, this volume). A combination of species and vegetative community monitoring will help in determining whether population declines are caused by local perturbations. If monitoring indicates no changes in vegetation yet populations are declining, that would indicate that local habitat conditions are not responsible for those declines.

Studies of grazing effects on small landbirds have reported exclusively bird abundance data, primarily during the breeding season, and evaluated only localized, short-term consequences of grazing. Land managers' legal mandates require long-term, landscape-level considerations that allow only land-use patterns that maintain natural populations, patterns, and processes. Grazing practices may not cause a great short-term change in some bird populations but we do not know the long-term consequences or whether there are widespread ramifications over the landscape. Data on long-term reproductive success, in a variety of vegetative cover types, over a broad scale would be a better index of the health of bird populations than abundance data in the short term (e.g., Van Horne 1983).

Species habitat requirements may be different from those predicted by information gathered from a limited area and time of the year, where only part of a species' life-history requirements are met (Mosconi and Hutto 1982). The ideal study would include large replicated areas totally protected from grazing for long periods. We need to assess the influence of livestock grazing in areas that are used for migration and overwintering, not merely for breeding habitats. This is particularly important in riparian habitats because they are critical for bird migration corridors (Stevens et al. 1977, Szaro 1980, Knopf et al. 1988a).

Clearly most plant and animal communities in the western United States, excluding some grasslands, have not evolved with widespread grazing repeated annually in the same locations. Thus, heavy grazing is likely to harm many species over the long term. Objectives for public lands must consider many other resources. Recently, the needs of wildlife, recreation, water quality, exemplary natural communities and biodiversity have been incorporated into management plans. For any given management unit, the objectives are apt to be specific to that geographic area, being both more complex and more detailed than was the case historically.

Given the ubiquity of livestock in much of the American West, plants and animals intolerant of activities by livestock grazing have relatively few places left to inhabit. This is undoubtedly true for birds and their habitats, which evolved in the absence of large herbivorous mammals. Protection and restoration of ungrazed habitats resembling their prehistoric counterparts must be fundamental to any conservation plan for Neotropical migrants and all other plant and animal species in western North America.

MANAGEMENT RECOMMENDATIONS

Livestock management decisions about western habitats will affect many Neotropical migrants significantly. Despite limitations to our current knowledge, we offer the following general management recommendations.

Grasslands

First, substantially increase the amount of public rangeland from which all livestock are permanently excluded (Bock et al. 1993). Of particular importance on the Great Plains are the US National Grasslands, which include more than 1.5 million ha presently managed by the US Forest Service largely for livestock production (Lewis 1989, West 1990). Many public rangelands presently are managed by applying some sort of rotational grazing strategy. However, the frequency of rotation is far too high to permit postgrazing ecological succession to proceed to the point where habitat is created for those Neotropical migrant birds (or any other species) generally intolerant of the impacts of grazing mammals. Furthermore, rotational livestock management fails to create heavily grazed habitats that may be required by some species. Therefore, we recommend establishing a system that creates a mosaic of heavily grazed habitats mixed with large (at least 1000 ha), permanent livestock exclosures, which would include a significant portion (perhaps 20%) of public lands presently dedicated to livestock production (Bock et al. 1993).

We are aware of the difficulties involved in designating public land as biological preserves and we recognize the competing interests involved. However, it is also important to recognize the declining agricultural value of many of these lands and their likely increase in value to the public as natural landscape (Popper and Popper 1991). We call only for an effort to restore to these lands something resembling their prehistoric condition. The obvious first step should be to free a portion of these lands from the controlling influence of domestic grazers.

Our second recommendation is to continue a modified version of the Federal Conservation Reserve Program (CRP), to encourage landowners to convert and maintain formerly tilled croplands as grazing lands planted to native vegetation. CRP lands remain vulnerable to recultivation and this decision rightly is in the hands of the landowners. However, it would be ecologically unfortunate if CRP lands were tilled (setting ecological succession back to zero), only to be returned to grassland when crop prices or future government incentives once again make it economically attractive. From the standpoint of indigenous flora and fauna, it would be much better to find ways of making the CRP grasslands valuable to landowners, perhaps by encouraging moderate amounts of livestock grazing. This strategy would be doubly valuable if it could somehow be coupled with creation of permanent livestock exclosures on the public rangelands, including especially the National Grasslands on the Great Plains.

Third, fire should be reintroduced to many grasslands from which it has recently been excluded, and where it is a natural ecological process.

Fourth, caution should be taken in implementing short-duration grazing as a grassland management tool. Short-duration grazing is advocated as a means of increasing livestock production, while improving rangeland condition (Savory 1988). Most field tests of this grazing system have failed to support either claim (e.g., Weltz and Wood 1986, Heitschmidt et al. 1986, Taylor et al. 1993). Furthermore, we found no studies evaluating the impacts of such high-intensity grazing on ground-nesting birds.

Shrubsteppe

First, exclude or significantly reduce livestock grazing. Although we have not documented likely responses of bird populations to this management change, avian communities are expected to respond positively in a landscape that resembles historical conditions. Where livestock are grazed, the short-term goal should be to maintain adequate herbaceous cover to conceal nests through the first incubation period. This could be accomplished by maintaining current season growth through 15 July, and allowing more than 50% (see Pond 1960) of the annual vegetative growth of perennial bunchgrasses to persist through the following nesting season.

Second, restore perennial bunchgrasses.

forbs, shrubs and plant-species diversity to historical levels. Seedings of native species, prescribed burns, and fall–winter grazing must be more carefully controlled to ensure the maintenance of residual plant cover.

Third, avoid fencing and water developments in circumstances where protection is needed for maintenance of plant communities and for population sources of species of special concern. This could result in the concentration of livestock in some areas while creating de facto protected areas in other locations.

Fourth, eliminate the conversion of shrubsteppe habitats to seedings of exotic grasses for the purpose of livestock grazing. Attempts should be made to restore burned areas of shrubsteppe to native vegetation rather than exotic seedings for livestock.

Fifth, determine methods for recovering soil cryptogam layers to increase soil moisture, increase seedling germination, reduce soil erosion, and enhance soil productivity.

Sixth, initiate long-term research that will help us understand the following problems: the direct effects of grazing on shrubsteppe avifauna, the indirect effects on the avifauna mediated through changes in vegetation, the influence of livestock on the distribution of Brown-headed Cowbirds, and the effects of cowbird parasitism on the productivity of breeding birds.

Riparian

First, the condition of riparian areas must be considered critically when implementing grazing systems. Given their scarcity, fragility, and importance to neotropical migrants and other wildlife, western riparian ecosystems should be excluded from livestock grazing wherever possible. Managers should evaluate how local activities alter potential dispersal opportunities for riparian species (Knopf and Samson 1994). Season of use, livestock numbers and livestock distributions must be strictly controlled within riparian zones to implement grazing programs that are compatible with riparian avifauna.

Where livestock must have access to riparian zones for water, restricted-access fencing can localize and minimize their impacts on streambanks and riparian vegetation. Development of alternate water sources also could help reduce concentration of livestock in riparian zones. When the cost of fencing is prohibitive, uplands and riparian zones must be managed together and grazing strategies should be keyed to the condition of the riparian vegetation.

Second, when riparian systems are grazed, moderate use during late fall and winter, or short-term use in spring, will be less damaging than continuous or growing-season grazing. Nevertheless, fall–winter grazing should be carefully controlled to ensure the maintenance of residual plant cover.

Third, degraded riparian habitats may require complete rest from livestock grazing to initiate the recovery process. Four years after cattle removal from riparian habitat in Arizona, understory vegetation and Neotropical migrants showed dramatic increases in abundance (Krueper 1993). In systems requiring long-term rest, the necessary period will be highly variable depending upon the extent of damage and growth rate of regenerating plant species (Clary and Webster 1989). Damaged riparian areas should be rehabilitated by revegetating with native species.

Coniferous Forest

Land managers and field biologists have an unparalleled opportunity to provide information where none currently exists concerning the impacts of grazing on Neotropical migrants in western coniferous forests. Monitoring of migratory landbirds both during the breeding season and in migration, with attention to matched forested habitats differing in grazing regimes or grazing histories, could supply much-needed data. Explicit quantitative assessment of grazing pressure and grazing histories, in conjunction with the collection of appropriate vegetation data will be critically important for assessing the relationships between grazing and Neotropical migrants. For breeding season studies, emphasis should be placed on species that nest and/or forage on or near the ground. Migration-period studies should be focused more broadly on the entire suite of species that utilize coniferous forest habitats as staging areas and for foraging activities.

Table 12-7. Standardized relative abundances of seven foraging guilds representing 68 species of Neotropical migrants in grazed and ungrazed habitats. Sample size is the number of occurrences in which a guild was represented in each treatment. Original data were taken from nine studies conducted in western riparian habitats (see Table 12-4). Guilds whose abundances differ significantly (paired t-test, $P < 0.05$) between treatments are indicated by an asterisk.

Foraging Guild	Sample Size	Standardized Mean		SE	P Value
		Grazed	Ungrazed[a]		
Aerial insectivore	24	1.04	0.96	0.43	0.83
Bark insectivore	4	0.89	1.11	1.30	0.88
Carnivore	4	0.58	1.42	0.71	0.21
Foliage insectivore	49	0.91	1.09	0.31	0.44
Ground insectivore	24	1.21	0.79	0.30	0.03*
Nectarivore	5	0.41	1.59	0.66	0.06
Omnivore	64	0.82	1.18	0.25	0.07

[a] Includes lightly grazed and fall-grazed treatments.

Ground insectivores were better represented in grazed areas. Over half the species in this guild are represented by birds (e.g., American Robin, Killdeer) that are well known for their adoption of human-altered habitats. Ground and aerial insectivores were the most commonly found guilds in grazed riparian habitats at various elevations in the Southwest (Szaro and Rinne 1988).

As a group, long-distance migrants (Status A in Table 12-4) appeared more susceptible (Standardized Means 0.86 vs. 1.13 [grazed vs. ungrazed], $T = -1.64$, $P = 0.10$) to disturbances by livestock grazing than short-distance migrants [status B in Table 12-4; standardized means 0.98 vs 1.01 ± 0.20 (grazed vs ungrazed), $T = 0.10$, $P = 0.92$]. One explanation for this result could be that many short-distance migrants are cavity nesters, whose nesting sites appear not affected by grazing in the short term. Long-distance migrants also might be more energy stressed upon arrival at the breeding grounds, and thus more vulnerable to human-related disturbances.

MONTANE CONIFEROUS FOREST HABITATS

Characteristics of Forested Habitats

Montane coniferous forests of western North America vary in species composition over broad geographic areas in response to the complex interactions produced by climate, elevation, latitude, soils, and the temporal and spatial pattern of disturbance factors such as fire. A highly simplified characterization for the general pattern of coniferous forest distributions would place juniper and xeric-adapted pine (*Pinus*) woodlands at lower elevations, ponderosa pine (*P. ponderosa*) savannahs at moderate elevations providing slightly less xeric conditions, and Douglas-fir (*Pseudotsuga menziesii*) forests and mixed-conifer associations at higher elevations that typically provide more mesic conditions. Throughout the West, lodgepole pine (*Pinus contorta* or *P. murrayana*) forests occur over a wide range of elevations, typically occupying areas following disturbance.

Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) are dominant tree species at high-elevation forests throughout the Rocky Mountains (Peet 1988). Mixed cedar-hemlock-pine (*Thuja–Tsuga–Pinus*) forests and grand-fir (*Abies grandis*) Douglas-fir forests also are common at lower elevations in the northern rockies and interior Northwest.

Jeffrey pine (*Pinus jeffreyi*) dominates mid-montane and lower montane forests in the eastern Sierra. Conifer associations of white fir (*Abies concolor*) incense-cedar (*Libocedrus decurrens*), sugar pine (*P. lambertiana*), ponderosa pine, and Douglas fir comprise the forests along the western slopes of the Sierra (Verner 1980).

Historical Perspective of Coniferous Forests

Many forests of western North America were maintained historically by frequent, low-intensity fires carried by fine herbaceous

Utah

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Vertebrate Information Compiled by the Utah Natural Heritage Program: A Progress Report

State of Utah Department of Natural Resources

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Greater Sage-grouse
Centrocercus urophasianus

TAXONOMY AND NOMENCLATURE

CLASS: Birds (Aves)

FAMILY: Grouse, Quail, and Pheasants (Phasianidae)

SUBSPECIES: The subspecies that occurs in Utah is *Centrocercus urophasianus urophasianus*.

CONSERVATION STATUS DESIGNATIONS

This species is included on the UDWR Sensitive Species List (UDWR 2003). Utah Partners in Flight identifies it as a Priority Species (Parrish et al. 2002). A management plan (UDWR 2002a) has been developed to facilitate recovery efforts.

DISTRIBUTION AND ABUNDANCE IN UTAH

Scattered populations occur throughout much of the state excluding the southeastern quarter of the state, being absent from most of the Colorado Plateau of southeast Utah (Fig. 68) (UDWR 2002a). Within this range, the extent of occupied habitat has declined by an about 60% from the historical extent (Beck et al. 2003). Populations occur primarily in habitat dominated by sagebrush (*Artemisia* spp.), especially big sagebrush (*A. tridentata*). Other habitats, such as wet meadows, may be of high importance seasonally.

The size of the Utah breeding population has been estimated to comprise 13,000 adults (UDWR 2002a). Population data collected since the late 1960s indicate statewide population declines (UDWR 2002a, Beck et al. 2003). Several factors may contribute to population declines (UDWR 2002a). For example, anthropogenic disturbance at lek sites may affect reproductive success rates. However, the primary factor affecting population levels is thought to be habitat loss. Although urban expansion and the conversion of native habitat to agricultural purposes may account for some habitat loss, especially historically, declines of populations have been largely attributed to decreasing suitability of sagebrush steppe habitat, which has resulted in the loss and fragmentation of sage-grouse habitat. Invasive non-native plants, particularly cheatgrass (*Bromus tectorum*), have resulted in dramatic changes to habitat structure and species composition in many areas. This grass is also involved in altered fire cycles and the associated conversion of large areas from shrub steppe habitat to nonnative grassland. Changes to sagebrush steppe habitat are also a result of overgrazing by livestock.

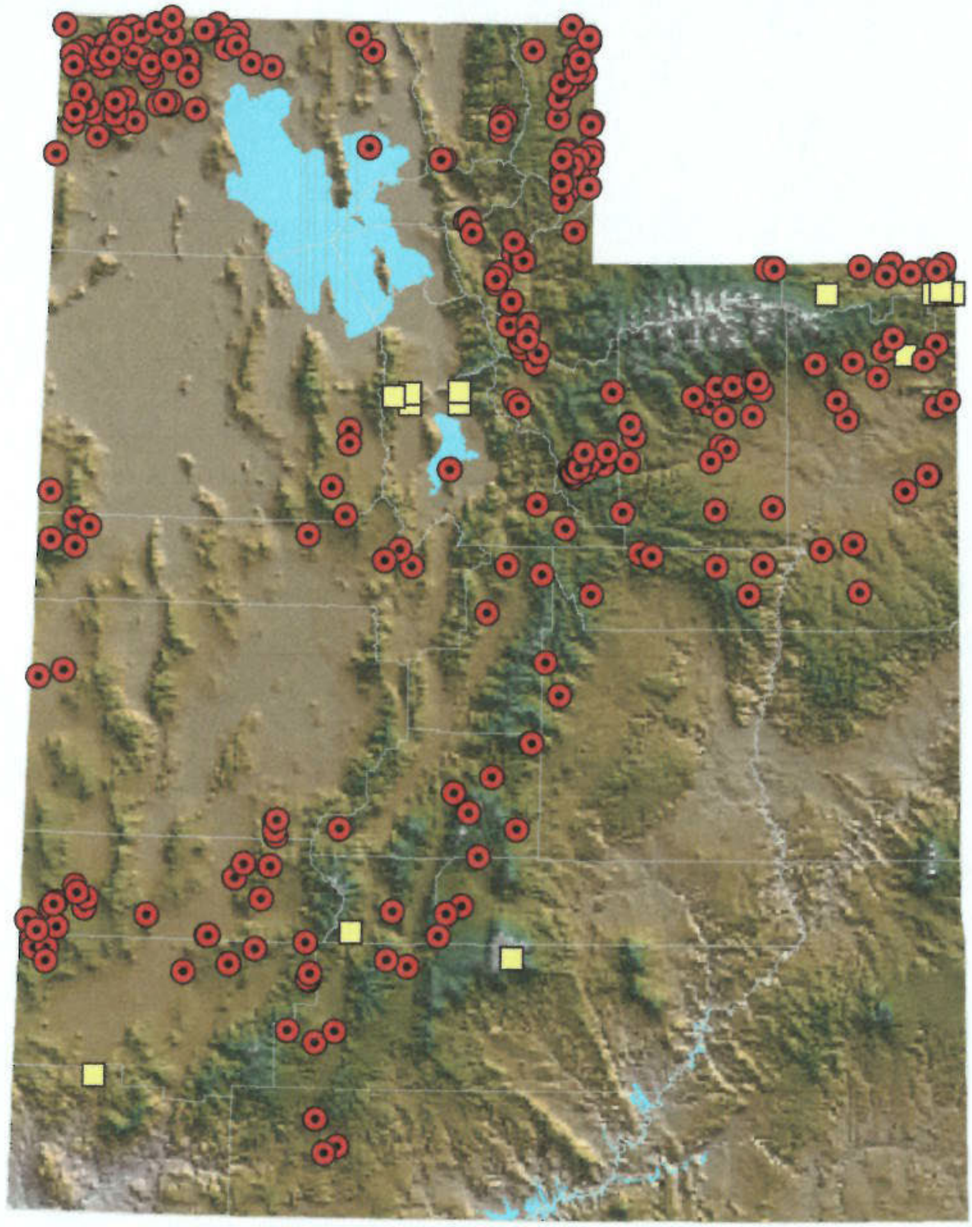

Figure 68. The distribution of records of the greater sage-grouse (*Centrocercus urophasianus*). Red circles represent records since 1983, inclusive, and yellow squares represent records before 1983.

Long-billed Curlew
Numenius americanus

TAXONOMY AND NOMENCLATURE

CLASS: Birds (Aves)
FAMILY: Shorebirds (Scolopacidae)
SUBSPECIES: Subspecies are not currently recognized.

CONSERVATION STATUS DESIGNATIONS

This species is included on the UDWR Sensitive Species List (UDWR 2003). Utah Partners in Flight identifies it as a Priority Species (Parrish et al. 2002).

DISTRIBUTION AND ABUNDANCE IN UTAH

This species breeds in scattered localities throughout the state, primarily in northern Utah, but also in the west and southwest (Fig. 72) (Behle et al. 1985). Cook (1984) presented evidence of nesting in Uintah County, but confirmation of breeding in northeastern Utah is lacking. This species occurs as a migrant throughout most of Utah. Arid grasslands, grassy shorelines, and agricultural areas are favored nesting habitats of this species (Walters and Sorensen 1983).

Populations are thought to have declined from historical levels, but few data are available to estimate the size of the historical or current population. Hayward et al. (1976) wrote that in Utah this species is "[a] fairly common summer resident and migrant ...." Behle et al. (1985) considered it to be a "[c]ommon summer resident in localized areas ...." Several authors have mentioned the decline of populations (Hayward et al. 1976, Behle et al. 1985). Loss of nesting habitat and disturbance to nest sites are suspected factors causing population declines (Hayward et al. 1976, Parrish et al. 2002). Increased predation rates associated with growing red fox populations are also of probable importance.

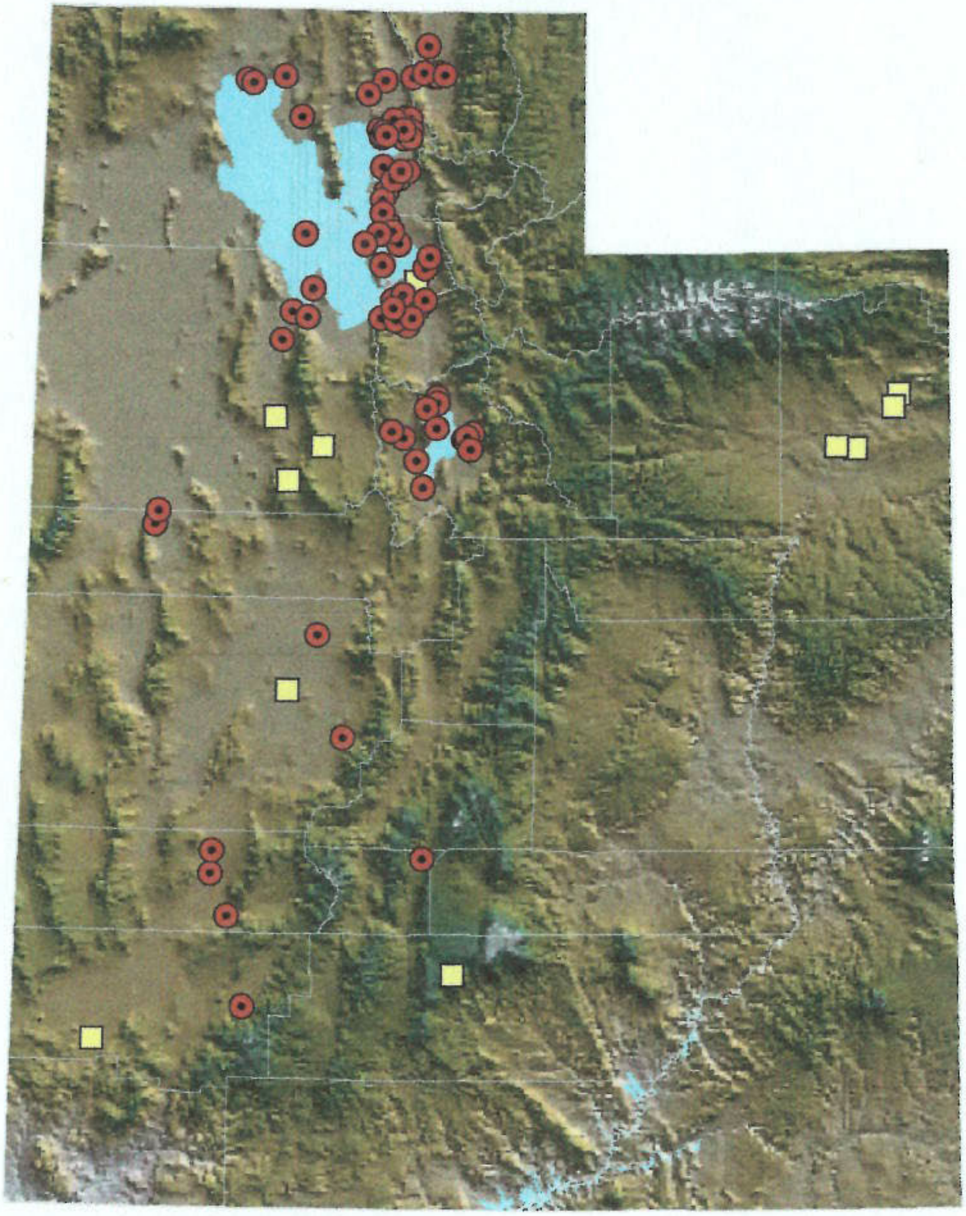

Figure 72. The distribution of records of the long-billed curlew (*Numenius americanus*) that represent probable or confirmed breeding activity. Red circles represent records since 1983, inclusive, and yellow squares represent records before 1983.

Burrowing Owl
Athene cunicularia

TAXONOMY AND NOMENCLATURE

CLASS: Birds (Aves)

FAMILY: Owls (Strigidae)

SUBSPECIES: The subspecies that occurs in Utah is *Athene cunicularia hypugea.*

OTHER NAMES: This species was formerly considered to be a member of the genus *Speotyto.*

CONSERVATION STATUS DESIGNATIONS

This species is included on the UDWR Sensitive Species List (UDWR 2003).

DISTRIBUTION AND ABUNDANCE IN UTAH

This species occurs statewide in scattered localities (Fig. 76). Nesting sites occur in a variety of shrub-dominated habitats, including sagebrush steppe and desert scrub, often in sparsely vegetated areas. An important component of the habitat is the presence of abandoned animal burrows in which the burrowing owl nests. In eastern and southern Utah, prairie-dogs (*Cynomys* spp.) create burrows that are often used by owl populations (Hayward et al. 1976). In western Utah where prairie-dogs are absent, vacant badger (*Taxidea taxus*) or ground squirrel (*Spermophilus* spp.) burrows may be used.

Populations may be locally common but are irregularly distributed (Hayward et al. 1976, Behle et al. 1985). Population declines and loss have been reported in some areas, particularly along the Wasatch Front where habitat loss to urbanization and agriculture has been severe (Hayward et al. 1976). Declining prairie-dog populations may affect owl populations (Evans 1982). Haug et al. (1993) also identified the use of pesticides (insecticides and rodenticides) and vehicle collisions (road mortality) as significant threats to populations.

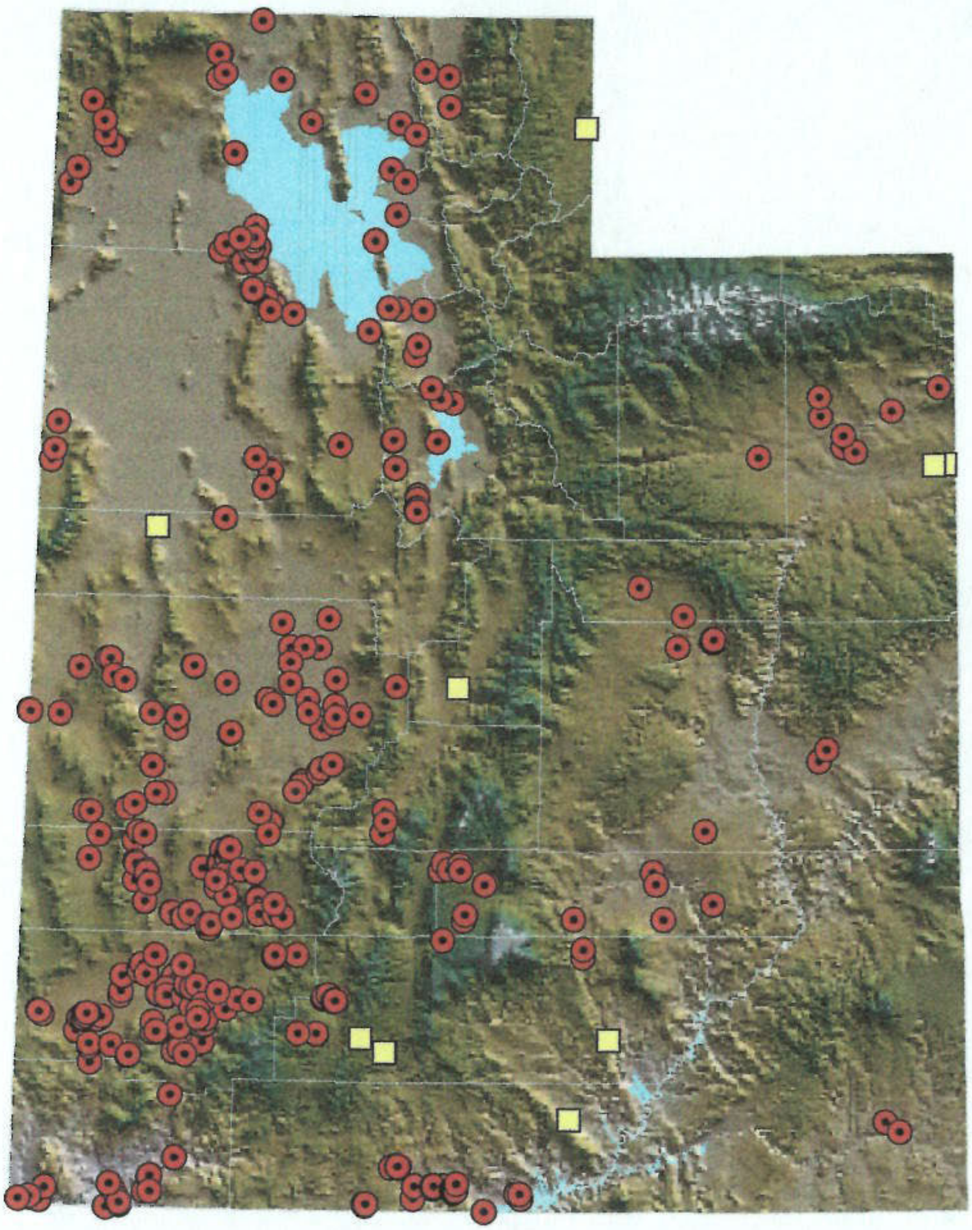

Figure 76. The distribution of records of the burrowing owl (*Athene cunnicularia*) that represent probable or confirmed breeding activity. Red circles represent records since 1983, inclusive, and yellow squares represent records before 1983.

TAXONOMY AND NOMENCLATURE

CLASS: Birds (Aves)
FAMILY: Owls (Strigidae)
SUBSPECIES: The subspecies that occurs in Utah is *Asio flammeus flammeus*.

CONSERVATION STATUS DESIGNATIONS

This species is included on the UDWR Sensitive Species List (UDWR 2003).

DISTRIBUTION AND ABUNDANCE IN UTAH

This species breeds across the northern two-thirds of the state (Fig. 78) (Walters and Sorensen 1983) and occurs throughout the state during non-breeding periods. Locally, breeding status is often difficult to evaluate because this species may breed opportunistically and sporadically in response to rodent density. It is said to be less common in eastern Utah (Hayward et al. 1976) and dramatic population decline has been noticed along the Wasatch Front (Behle et al. 1985). Such declines are the result of urban and agricultural encroachment on its habitat, threats that are likely driving declines range-wide.

Walters and Sorensen (1983) listed the habitats in Utah where this species is known to nest as marshes and wet hummocks, agricultural croplands (non-woody), arid grasslands; they listed other habitats utilized during the breeding season as cold desert shrub (including saltbrush and greasewood) and sagebrush-rabbitbrush. They considered all of these habitats to be utilized during winter.

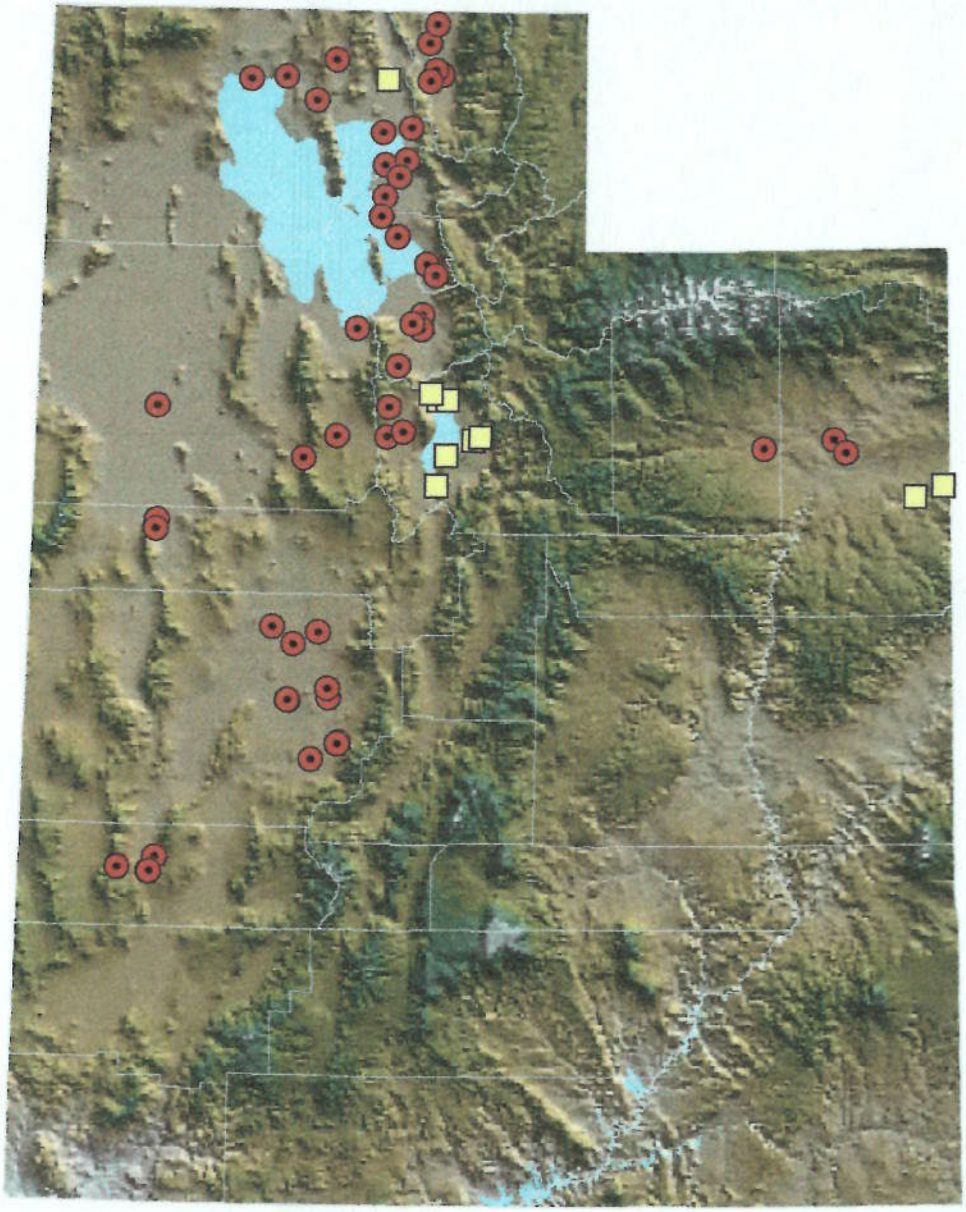

Figure 78. The distribution of records of the short-eared owl (*Asio flammeus*) that represent probable or confirmed breeding activity. Red circles represent records since 1983, inclusive, and yellow squares represent records before 1983.

Bendire's Thrasher
Toxostoma bendirei

TAXONOMY AND NOMENCLATURE

CLASS: Birds (Aves)
FAMILY: Thrashers (Mimidae)
SUBSPECIES: No subspecies are recognized.

CONSERVATION STATUS DESIGNATIONS

No conservation status is currently applied by management agencies.

DISTRIBUTION AND ABUNDANCE IN UTAH

Few data are available regarding the status of this species. It appears to breed sparsely in the state. Breeding occurs in open sagebrush and sagebrush-juniper habitat (England and Laudenslayer 1993) and has been definitively documented in Utah and Uintah counties (Fig. 84) (Bee and Hutchings 1942, White et al. 1983). Nesting is also suspected to occur in Washington, Iron, Garfield, Kane, San Juan, Grand, and Tooele counties (e.g., Hayward et al. 1976), but locations and breeding status have generally been poorly documented.

No information is available to suggest the size or trend of the Utah population. Degradation of habitat associated with introduced plant invasions, disturbance to sagebrush steppe habitat from livestock use, and altered fire regimes have the potential to affect the viability of breeding populations.

Figure 84. The distribution of records of Bendire's thrasher (*Toxostoma bendirei*) that represent probable or confirmed breeding activity. Red circles represent records since 1983, inclusive, and yellow squares represent records before 1983.

Pygmy Rabbit
Brachylagus idahoensis

TAXONOMY AND NOMENCLATURE

CLASS: Mammals (Mammalia)
FAMILY: Rabbits and Hares (Leporidae)
SUBSPECIES: No subspecies have been recognized.

CONSERVATION STATUS DESIGNATIONS

This species is included on the UDWR Sensitive Species List (UDWR 2003).

DISTRIBUTION AND ABUNDANCE IN UTAH

Populations are distributed in western Utah, primarily in areas within the Bonneville Basin (Fig. 101). Populations occur in areas having dense, tall stands of sagebrush (*Artemisia* spp.), especially big sagebrush (*Artemisia tridentata*). Local distribution is also correlated with soil characteristics that are conducive to burrowing. Populations generally occur in areas having sandy soils or in association with deep alluvial deposits.

Few data are available to indicate the size or trend of populations. Population density varies greatly spatially, evidently in response to habitat quality, but temporal variability is poorly understood (Green and Flinders 1980). Pygmy rabbits have not been detected at some of the sites recorded by Janson (1946) (UDWR unpublished data), which suggests a decline in the area of occupancy from historical levels. Apparent declines are thought to be related to the decline of range conditions, specifically the degradation and loss of sagebrush steppe habitat. Habitat loss has been the result of altered fire regimes, development, and agricultural conversion. Livestock overuse and weed invasions also number among the important factors contributing to the degradation of sagebrush habitat.

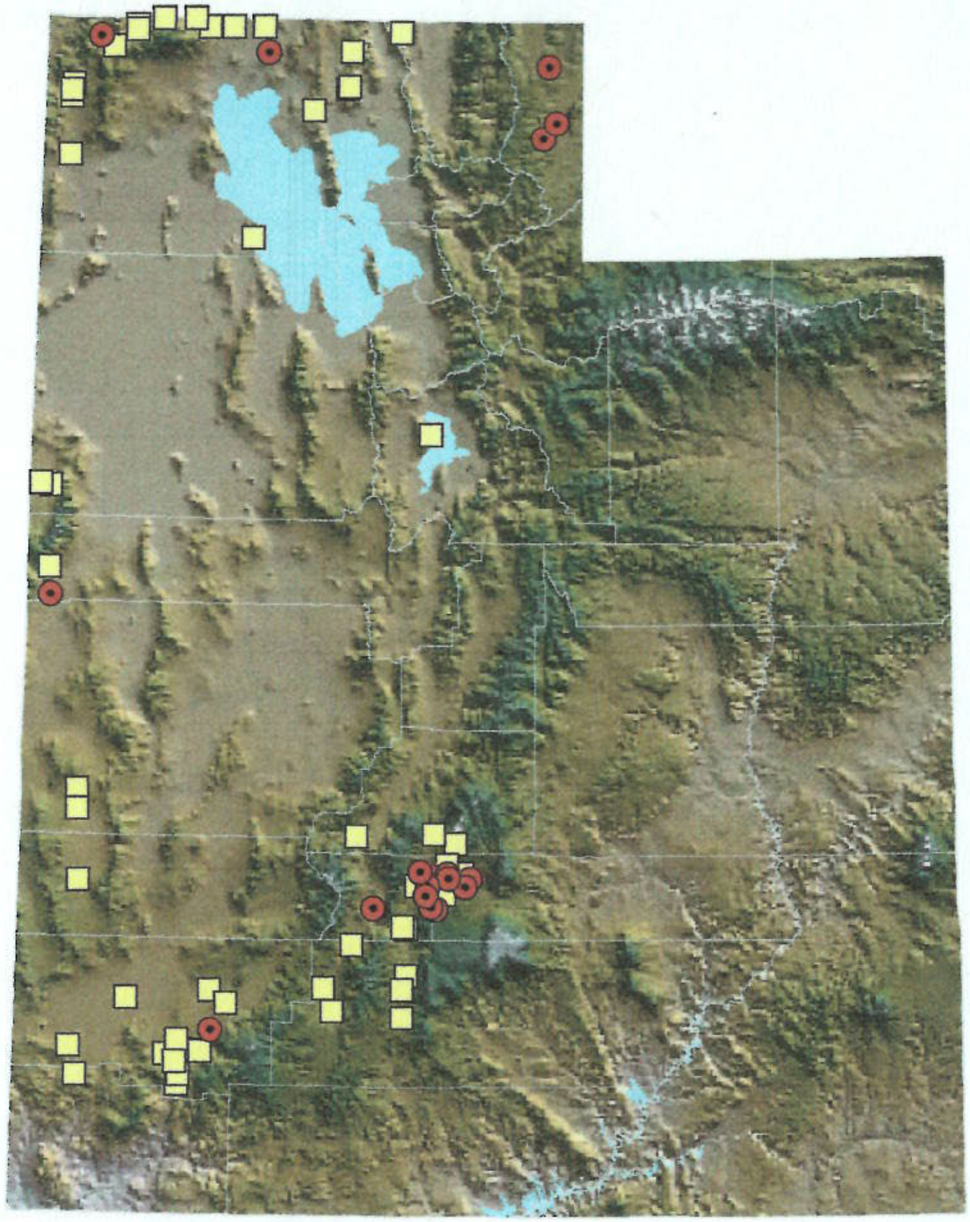

Figure 101. The distribution of records of the pygmy rabbit (*Brachylagus idahoensis*). Red circles represent records since 1983, inclusive, and yellow squares represent records before 1983.

Taxonomy and Nomenclature

Class: Mammals (Mammalia)
Family: Squirrels, Chipmunks, and Prairie-dogs (Sciuridae)
Subspecies: No subspecies have been proposed.

Conservation Status Designations

The Utah prairie-dog was classified by USFWS as an endangered species under the federal Endangered Species Act during 1973 (38 Federal Register 14678). It was later downlisted to threatened status during 1984 (49 Federal Register 22330-22334). Conservation efforts are currently guided by a conservation strategy (Utah Prairie Dog Recovery Implementation Team 1997). This species is included on the UDWR Sensitive Species List (UDWR 2003).

Distribution and Abundance in Utah

This species is endemic to southwestern Utah, occurring in the southern Bonneville Basin and the high-elevation plateaus of central Utah (Fig. 107). Collier (1975) found that several habitat factors were important for this species: elevation below 9,000 ft, the availability of water in addition to precipitation, heterogeneity of plant community, less than 10% of the vegetative cover composed of "tall" (12 in. or 31 cm) vegetation, and non-alkaline soils. Crocker-Bedford and Spillett (1981) stated that historically "[p]rime habitat would have been below 2,200 m in elevation and would have had much cool season palatable forage.... [M]ost Utah prairie dogs now inhabit either densely populated colonies which have alfalfa, or sparsely populated colonies on high plateaus. Permanent Utah prairie dog colonies always are associated with areas that provide moist vegetation throughout the summer.... The nutritious, succulent plants found in such areas are crucial for Utah prairie dogs: colonies without such vegetation are decimated by drought, and higher moisture content in the vegetation allows greater population density ...."

Populations have declined dramatically from historical levels (e.g., Collier and Spillett 1972). The total number of Utah prairie-dogs has fluctuated between 3,500 and 6,000 adults since 1991 (e.g., McDonald 1996, Bonzo and Day 2002). Habitat loss arising from development and agricultural uses is the primary threat to populations. Intentional control efforts, including poisoning and shooting, have also been of importance in some areas. Sylvatic plague is an introduced disease that is, in part, responsible for tremendous fluctuations in population size.

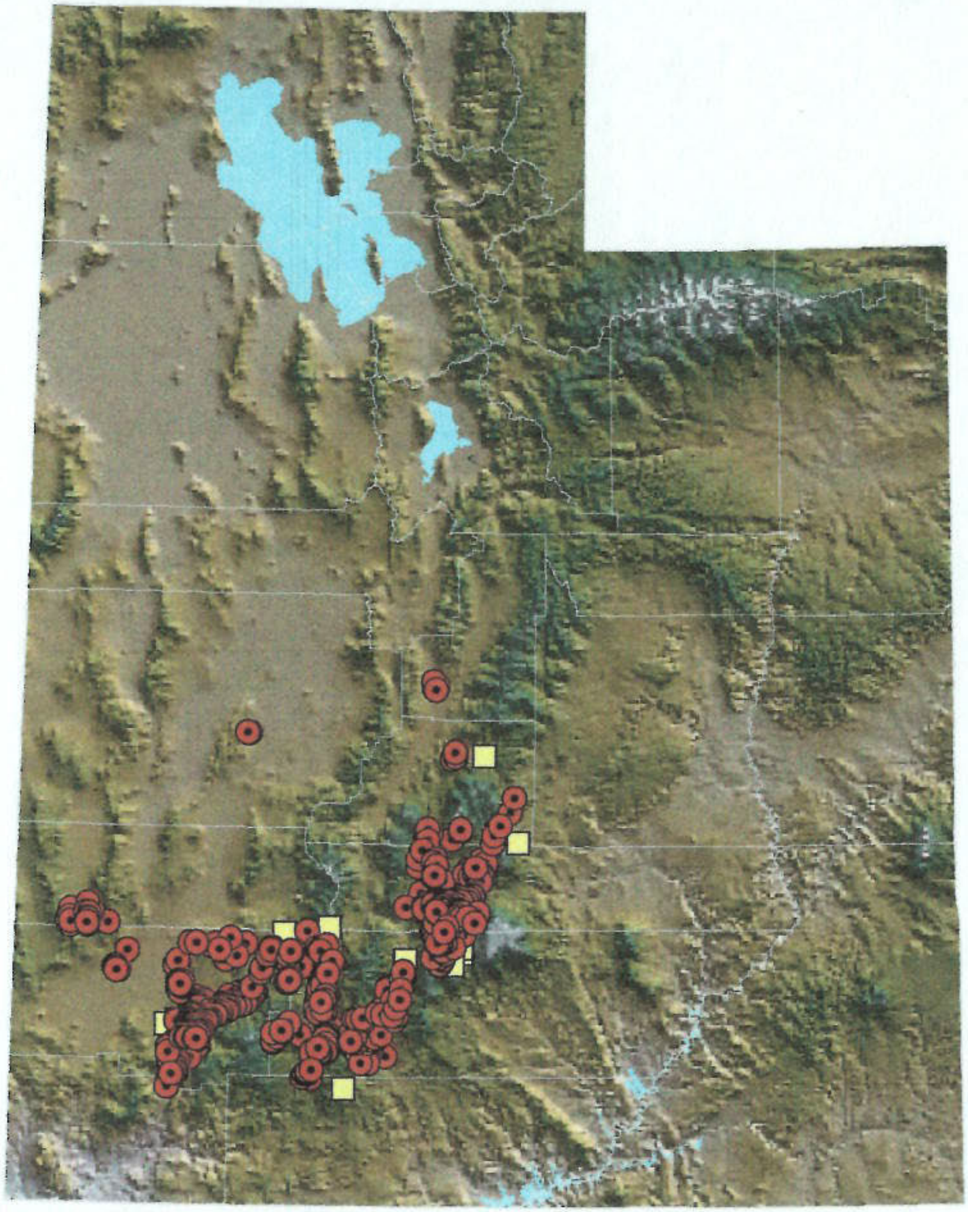

Figure 107. The distribution of records of the Utah prairie-dog (*Cynomys parvidens*). Red circles represent records since 1983, inclusive, and yellow squares represent records before 1983.