SUBURBAN AND EXURBAN INFLUENCES ON WILDLIFE AND FISH

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TABLE OF CONTENTS

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"As the years roll on, the tribute which we extract from the American landscape to sustain our prodigal society increases. There is ever growing demand for more agriculture crops, more livestock, and more wood products, while the acres for production are shrinking from urbanization and other causes. How is wildlife faring in this changing world?"

A. Starker Leopold (1978)

LIST OF TABLES

EXECUTIVE SUMMARY

Urbanization is the primary cause of species endangerment and a leading threat to biodiversity in the contiguous U.S. Unfortunately, urban sprawl has only recently been addressed as a serious issue. Although Montana is geographically the 5th largest state in the U.S. and has <1,000,000 people, no one believes it will be immune to the negative issues that are facing wildlife due to development along the rural-urban gradient. As a consequence, Montana Fish, Wildlife & Parks is concerned about suburban and exurban growth and how it will influence wildlife and their habitat. Our objective was to review relevant literature that addresses how suburban and exurban growth influences fauna in Montana.

We reviewed >400 articles and cited >350 of them in this review. Of those cited, 53% were related to game animals, 21% to non-game, 9% to fish, and 16% to management and other topics. Most (96%) were peer-reviewed and either descriptive (67%) or gradient studies (17%). Only 7% used treatment and controls. Most study periods were short: <1year (25%), 2-5 years (38%), and >5 years (8%). Over 25% of studies did not specify the length of their study. Our review concentrated on fish, reptiles and amphibians, avifauna, and mammals.

Fish populations have been seriously impacted due to reduced water flow, passage across fragmented habitats, contaminants, impervious surface cover, and other anthropogenic impacts from development. The challenges are complex and the best efforts of mitigating for urban development will not preclude additional losses to fish populations and their habitat.

Reptiles and amphibians face similar challenges. Lizards, snakes, turtles, salamanders, and anuruns have all decreased in the face of anthropogenic activities. Biologists and managers concerned with amphibian and reptile conservation need to be aware that relatively limited

development of watersheds alone, not necessarily in the riparian area itself, or even directly upstream, can influence reptiles and amphibians.

Many of the avifauna studies examined were designed to measure various characteristics associated with avifauna communities and altered landscapes from urbanization and most covered large areas. In all cases, native avifauna benefited from native vegetation in undeveloped land or surrounded by undeveloped land. The problems with urbanization and avifauna have been clearly established but few recommendations have been made to mitigate for urban, suburban, and exurban centers.

Most mammals do not benefit from urbanization (but some do) and serious problems arise due to larger species (e.g., coyotes [*Canis latrans*], American black bears, [*Ursus americanus*], ungulates) using urban areas. There is an abundant source of information about the biology of these species, documentation of the conflicts when associations with humans occur, and recommendations that have been suggested to minimize wildlife-human conflicts. Management plans and mitigation for conflict resolution between humans and American black bears and coyotes have been successful in some areas because information about the animal, habitat, and human attitudes were considered. The public involvement associated with human-wildlife conflicts is a critical component of successful management.

As with predators, the biggest concern between humans and ungulates revolve around damage and human safety along rural-urban gradients. Some authors have indicated the problems with human land uses and ungulate habitat but research explaining how to enhance and maintain habitats for ungulates is not well developed. However, it is critical that the public be involved in all decisions. Because people have such a large investment in managing wildlife in urban areas, their attitudes have to be considered when formulating management plans.

Purchasing wildlife habitat for conservation is likely the most important way to minimize the anthropogenic effects of urbanization on populations. The arena of understanding wildlife and fish among the rural-urban gradient is relatively new to research. The conflicts created for wildlife along the gradient are human caused and will only be addressed properly by collaborative cooperation that leads to clear objectives and mechanisms to meet them. The problem has to be addressed or the chaotic and planned consumption of habitat for wildlife will continue decreasing the quality of life for all organisms.

Urbanization, Missoula, Montana. Photo courtesy of J. Wright.

INTRODUCTION

Urbanization is the primary cause of species endangerment and a leading threat to biodiversity in the contiguous U.S. (Czech et al. 2000). Rural, suburban, and exurban development in the U.S. consumes ~ 1,000,000 ha of land /year (Milder et al. 2008) and contributes to increased anthropogenic mortality of wildlife from collisions with vehicles and windows, dog and cat predations, malicious gun shot, monofilament line injuries, tar, oil, fly paper contact and pesticide toxicities (Burton and Doblar 2004). Diseases also contribute to mortality factors of urban wildlife (Krausman 2002): house finch conjunctivitis, trichomoniasis, salmonellosis, sarcastic mange and notaedric mange and rabies (Krausman 2002, Burton and Dobler 2004). Cities, towns, and villages in the U.S. make up approximately 5% of the land mass but nearly 80% of the population lives in them and the surrounding suburbs (United States Census Bureau 2001). This anthropogenic development results in the modification of landscapes so severe that native habitat for wildlife is altered and eliminated (McKinney 2002). Most alteration occurs around cities and the influences of development decrease as human activity decreases from the core through suburban and into rural landscapes. This expansion of residential areas into rural landscapes is urban sprawl (Lindstrom and Bartling 2003), which creates new habitats from the native landscapes: cemeteries, parks, gardens, golf courses, lakes, and tons of asphalt and concrete. Although some species adapt well to urban sprawl (e.g., some amphibians, reptiles, small mammals, birds), many more are negatively impacted (Randa and Yunger 2006) due to abrupt habitat boundaries, road construction (Hawbaker et al. 2006), introduction of exotic flora and fauna, degradation of landscapes by humans causing long term habitat loss and increased extinction rates (McKinney 2002) leading to international biotic homogenization (McKinney 2006). As “cities expand across the planet, biological homogenization increases because the same ‘urban-adaptable’ species become increasingly widespread and locally abundant in cities across the planet” (McKinney 2006: 247). These “global homogenizers” combined with native species that adapt well to suburban environments (i.e., early successional plants and edge animal species, ground-foraging, omnivorous and frugivorous birds that use gardens, forest fragments and other suburban landscapes) provide some developed areas a rich assemblage of flora and fauna even though many native species declined. This has likely received little attention because many humans that live in suburban and exurban habitats (of all income levels) become increasingly disconnected from local indigenous species and their roles in the natural ecosystem (McKinney 2006).

In the U.S. and Canada, on average, the greatest levels of biotic homogenization were predicted for plants (22%) and fishes (14%), followed by reptiles and amphibians (12%), mammals (9%), and birds (8%). Homogenization is predicted to be greatest for fish in the southwestern and northeastern U.S., highest in eastern North America for plants, greatest for birds and mammals along the west coast of North America, and peaks in the Southeast for reptiles and amphibians (Olden et al. 2006).

Similar effects occur as rural landscapes are altered with livestock and agriculture. In both situations the concentration of anthropogenic influences is not generally beneficial to wildlife and is creating serious challenges for wildlife biologists and managers. Wildlife that are able to tolerate humans within their habitat are often undesirable to many citizens leading to human-wildlife conflicts; these conflicts can range from minor annoyances to property damage and the loss of life. Until planners consider wildlife at the development stage, habitats will continue to be altered, populations will be reduced and eliminated, and the overall quality of life will be reduced. The area of the western U.S. dominated by anthropogenic features is ~13%.

Areas by rivers are generally influenced more by the human footprint than lakes (Leu et al. 2008). The disproportional influence of humans on the western landscape creates a challenge to biologists and managers.

Unfortunately, urban sprawl has only recently been addressed as a serious issue. Although Montana is the 5th largest state in the U.S., there are <1,000,000 people, and no one believes it will be immune to the negative issues that are facing wildlife throughout North America. Exurban development will likely persist in the Rocky Mountains as people search for scenic and secluded lifestyles (Romme 1997). As a consequence, Montana Fish, Wildlife & Parks is concerned about suburban and exurban growth. How does this growth affect fish and wildlife, their movements, habitats, and connectivity? When negative impacts are identified, what practices are being applied to avoid, minimize, or mitigate the impacts? How effective are the mitigations? These data will provide an understanding of the current state of how urban sprawl is influencing wildlife. Plant communities have been examined with gradient analysis for decades (Whittaker 1967) and the technique has been used with avifauna, mammalian, and aquatic studies. Most recently it has been used to examine the complexity of urban ecology (McDonnell et al. 1993).

Development of Wildlife Conservation Along the Rural-Urban Gradient

In the United Kingdom, the shift to urban lifestyles began earlier than in the U.S. and ecologists began studying urban ecology in the early 1900s (Shenstone 1912, Shaw et al. 2009). Fitter (1945) published the first book on urban ecology as development in and around London progressed (Shaw et al. 2009). Urban ecology became popular in the U.S. in the 1960s and 1970s associated with the environmental movement (Shaw et al. 2009). However, Leopold

(1933) actively searched for ways to minimize anthropogenic influences on wildlife (Miller and Hobbs 2002). Leopold's efforts marked a shift in wildlife management to include non-game species and a conservation-based approach to management (Hadidan and Smith 2001). The shift was accepted by the urbanizing public because they were interested in attracting wildlife to their homes and understanding the distribution of animals in developed areas (DeStefano and DeGraaf 2003, Shaw et al. 2009).

With the shift from "old conservation" (i.e., exploitation) toward "new conservation" (i.e., clean water, air, open space, outdoor recreation, quality of human environments; Dasmann 1966, Shaw et al. 2009), a suite of laws was enacted (e.g., Endangered Species Act) and non-game management programs were established in state fish and wildlife agencies (Shaw et al. 2009). The first conference on the urban environment was sponsored by the U.S. Fish and Wildlife Service in 1968 and the National Institute for Urban Wildlife was founded in 1973 (Adams 2005). These activities and others led to an explosion of interest and research in attracting wildlife to backyards (Shaw et al. 2009), how wildlife coexisted with humans (Destefano and DeGraff 2003), bird-habitat relationships (Emlen 1974, Campbell and Dagg 1976, Lancaster and Rees 1979, Beissinger and Osborne 1982, DeGraff 1991, Blair 1996, Germaine et al. 1998, Melles et al. 2003), values of wildlife watching, surveys of public attitudes, and planning and management (Lyons and Leedy 1984, Shaw et al. 1985, Shaw et al. 2009). However, urban ecology remained at the fringes of mainstream ecology because many considered anthropogenic alterations to areas as biological degradation and suggested that studies in undisturbed natural areas were of more value (Miller and Hobbs 2002). During the 1980s, urban wildlife research concentrated on ways to mitigate human-wildlife conflicts (Loker et al. 1999, Shaw et al. 2009) and throughout the 1980s and 1990s, interest in urban ecology surged with growing interest from management agencies, universities, working groups, and professional societies (Shaw et al. 2009). In the 1990s the significance of urban and exurban landscapes to the conservation of biodiversity was widely recognized; conservation that focused primarily on wildlands was not adequate to maintain a full range of biodiversity (Shaw et al. 2009). Furthermore, urbanization was fragmenting habitats and urban “ecology moved quickly from the periphery of ecological science to the mainstream, as concerns about unprecedented growth of suburban areas and corresponding loss of open space motivated ecologists to develop research programs to inform regional planning and conservation initiatives (DeStefano et al. 2005)” Shaw et al. 2009:xx). Universities hired urban wildlife specialists (Decker et al. 2001) and Baltimore, Maryland and Phoenix, Arizona were added to the National Sciences Foundation’s Long-term Ecological Research programs (Kingsland 2005). By 2000, there were non-game wildlife programs in every state wildlife agency (Shaw et al. 2009).

Restoring and maintaining habitats in urban areas to conserve indigenous species by reducing the impacts of urbanization and reducing human-wildlife conflicts in urban areas are the dominant fields of study in urban ecology (McKinney 2002, DeStefano and DeGraff 2003, Shaw et al. 2009). Both arenas have involved extensive involvement with passionate and often controversial conflicts; most urban centers are on public lands dominated by humans. The public has to be part of any method used to enhance wildlife or wildlife habitat, minimize habitat alteration, or reduce human-wildlife conflicts. “In urbanized areas, developing a more ecologically informed public could be the most effective way to promote the conservation of native species and reduce human-wildlife conflicts (McKinney 2002)” (Shaw et al. 2009:xx).

Our objective is to review relevant primary and gray literature that addresses how suburban and exurban growth influences wildlife and fish. Because this is a relatively new area of study, there are few examples from Montana. We will present examples from other areas that may be applicable to assist managers in Montana so planning can progress with minimal negative influences to wildlife.

SCOPE OF REVIEW AND METHODS

We used numerous search engines (e.g., web of science, wildlife and ecology studies worldwide, proquest digital dissertations) at the Science Library, University of Arizona and the Mansfield Library, University of Montana to locate articles related to wildlife and suburban and exurban development. The literature cited in these articles were also searched for other relevant references. Additional material was obtained through personal communication, Montana Fish, Wildlife & Parks, and from biologists around the country.

We were primarily interested in the influence of the human footprint on Montana. However, data from Montana are limited so we used other studies that are applicable to the state. We categorized animals as fish, reptiles and amphibians, avifauna, and mammals and delineated the major problems development were creating, and the problems and mechanisms of establishing policy. There are 4 key elements that influence the development and implementation of public policy regarding wildlife management: biophysical animal behaviors, social-structural, valuation, and institutional-regulatory forces (see definitions). "The multi-dimensional, interactive and dynamic characterization of the wildlife policy process suggests its extreme complexity and subtlety" (Kellert 1994:43). The entire process is further complicated by growing diversity of land uses, agency missions, different human attitudes, and when wildlife issues occur where humans and wildlife share fragmented habitats (Peine 2001). When enough data were available to examine each of these forces, we attempted to demonstrate how they were instrumental in formulating wildlife policy related to human-wildlife interactions.

DEFINITIONS

Because urbanization is a relatively new field in wildlife management (Krausman 2002), we present definitions of terms used in this review. We recognize that others may use different definitions.

1. Biophysical-behavioral forces.--Biological and ecological properties associated with wildlife including population dynamics, reproduction, habitat, and behavior (Keller and Clark 1991).
2. Exurban development.--Approximately 6 – 25 homes/km² and includes urban fringe development on the edge of cities and rural residential developments in rural areas that have natural amenities (Hansen et al. 2005). "… Exurban development occurs beyond incorporated city limits, and the surrounding matrix remains the original ecosystem type" (Odell and Knight 2001).
3. Institutional-regulatory forces.--These are factors that allow wildlife management to operate successfully: financial base, law enforcement, states' rights for residential wildlife, expansion of wildlife values, public trust, federal regulation of migratory species, and inclusion of habitat when defining wildlife (Kellert and Clark 1991).
4. Rural areas.--An area with < 193 people/km² (U.S. Census Bureau 2002).
5. Social-structural forces.--The role of power versus property relationships that reflect the rights and privileges to use and control wildlife resources (Kellert and Clark 1991). Wildlife in the U.S. is common property but access can become complicated on private lands, especially where federal and private lands merge.

6. Suburbs.--The patchwork of residential, commercial, municipal, and industrial land uses and related transportation and utility corridors often adjacent to urban centers (Knuth et al. 2001).
7. Synurbization.--Adjustment of wild animal populations to specific conditions of the urban environment (Luniak 2004).
8. Unnatural food.--Any food for wildlife made available by humans (Peine 2001).
9. Urban areas.--An area with a high-density core of ≥ 386 people/km² (U.S. Census Bureau 2002).
10. Urbanization.--Changes in landscape caused by urban development (Luniak 2004).
11. Urban stream syndrome.--The consistently observed ecological degradation of streams draining urban lands. Symptoms “include a flashier hydrograph, elevated concentrations of nutrients and contaminants, altered channel morphology, and reduced biotic richness, with increased dominance of tolerant species” (Walsh et al. 2005:706).
    a. Urban wildlife.--Species that find habitat in urban and suburban landscapes (e.g., deer [*Odocoileus* spp.], raccoons [*Procyon lotor*], coyotes [*Canis latrans*]; Krausman 2002).
12. Valuation forces.--The general manner in which society considers wildlife. These forces can be categorized as economic, ecological, and social-physiological (Kellert and Clark 1991). Valuation forces are essentially the manner in which society views wildlife (Peine 2001). For example, tourist operations may use wildlife to entice customers (economic), biologists encourage biodiversity for the necessary roles in ecological processes (ecological), while Native Americans revere some wildlife spiritually (social-physiological).

RESULTS

We found relevant material in >350 articles: 192 (53%) were related to game animals, 77 (21%) to non-game animals, 32 (9%) to fish, and 58 (16%) to management and other topics. The majority (96%) of the articles were peer-reviewed. Most studies were descriptive (67%), gradient studies (17%), surveys (4%), used treatment and controls (7%) and 3% were analyzed differently. Eleven papers (3%) were reviews. Most sample sizes were <100 and study periods were from < 1 year (25%), 2 – 5 years, (38%), and > 5 years (8%). Many study periods (29%) were not specified. Studies were sponsored by universities (55%), governments (31%), NGO (6%), or private sources (3%). Several authors (5%) did not indicate the source of support for their work.

Fish

The Clean Water Act was passed in 1972 and since then the U.S. Government has increased attempts to reduce the threats to rivers, lakes, and wetlands from pollution. There have been numerous successes but nearly half of the nation's surface waters are not capable of supporting basic aquatic values or water quality that is even safe for recreational swimming (Bohn and Kershner 2002). Urbanization has contributed to the contamination of the nation's riparian systems and related flora and fauna (McDonnell and Pickett 1990). Urbanization has significantly altered fish passage, containments, impervious surface cover, ecology, and fish assemblages of riparian systems. Management will have to be incorporated into the planned and chaotic urban experiment to restore aquatic systems and associated flora and fauna. Walsh et al. (2005:706) called the "consistently observed ecological degradation of streams draining urban land"…the 'urban stream syndrome'."

Fish Passage Structure

Roads throughout the U.S. have altered landscapes and highway culverts often impede or block fish movements (Belford and Gould 1989). To minimize obstructed fish passage through culverts, engineers should consult with fish biologists so that all fish passage considerations are recognized including sex, age, species of fish, swimming speed, water depth, role of migration in their life history process, control of sediment, inlet drops, culvert alignment, water velocity, and culvert outfall barriers (Belford and Gould 1989, Votapka 1991, Fitch 1996, Hegberg 2001, Blank et al. 2005, Bufford 2005). Culvert installation should not change riparian conditions that existed prior to installation. Ideally, properly constructed culverts can limit the use of bridges where culverts are appropriate and eliminate the use of culverts where restrictions to fish movements cannot be overcome (Fitch 1996). In Virginia, culverts can be the primary options for crossing streams known to provide habitat for trout if the culvert is placed on the same slope as the streambed and the slope is <3%, flow velocity does not exceed 1-2m/second under normal flow conditions and the barrel of the culvert can be properly countersunk at the outlet to prevent perching (Fitch 1996).

Small-stream fish are highly mobile and influenced by the type of structure used to allow vehicles to cross streams. In Arkansas, culvert, slab, open-box, and ford crossings are the type of structures used. Overall, fish movement and number of species was lower through culverts than natural reaches or other structures. In spring, retention of fishes was highest upstream of crossings and lowest in downstream segments for all crossing types (Warren and Pardew 1998). The road crossings could be improved by determining critical levels of water velocity through crossings (Warren and Pardew 1998). This is especially important in urban areas where urban stream design, low-flow, and the aquatic habitat channel are forgotten (Hegberg 2001).

In Montana, Belford and Gould (1989) and Blank et al. (2005) examined passage of rainbow trout (*Oncorhynchus mykiss)*, brown trout (*Salmo trutta)*, cutthroat trout (*O. clarkii)* and brook trout (*Salvelinus fontinalis)* through culverts of varying length (45 – 93m). They found similar "strenuous passage relations" among species after a passage length of about 10m when the slope of the strenuous-passage relation becomes relatively flat (Belford and Gould 1989).

Fish passage through a variety of culverts was also examined for prairie fish along 2 tributaries of the lower Yellowstone River (Rosenthal 2007). With the exception of longnose dace (*Rhinicthys cataractael*), fish movement through culverts was similar to movement in natural reaches (Rosenthal 2007).

Additional research on culvert design is needed and on-going (Blank et al. 2005). In a basin-wide study of culverts in the Clearwater River drainage near Seeley Lake, Montana, 46 culverts of different types were examined; 76 to 85% were barriers at low flow for cutthroat trout, brook trout, brown trout, and bull trout (*Salvelinus confluentus*; (Blank et al. 2005, Bufford 2005).

Contaminants

As riparian ecosystems are influenced by urbanization, there is a concentration of heavy metals in fish associated with urban runoff, roads, and other pollutant sources. In Orlando, Florida, significant concentrations of heavy metals were documented for fish living in stormwater ponds, especially in the redear sunfish (*Lepomis microlophus*). Heavy metal concentrations were influenced with bottom sediments (Campbell 1994).

There are high concentrations of metals (e.g., As, Cd, Cu, Pb, Zn) in surface waters in the intermountain western U.S. where discharges from mining activities have occurred since the late

1800s (Farag et al. 1995). In the Clark Fork River, Montana, tissue metal accumulation in brown trout were significantly higher than fish collected at reference sites resulting in impaired fish health (Farag et al. 1995). Chronic exposure (≤100 days) of rainbow trout to sublethal levels of waterborne Cd, Cu, or Zn caused alterations of appetite, reduced swimming speeds, and metabolic costs (McGeer et al. 2000).

In central Puget Sound, Washington, the Hylebos Waterway is severely contaminated by organic and inorganic contaminants. Female flatfish show evidence of precocious sexual maturation in young animals, inhibited gonadal development in older fish, and salmon are exposed to chemical contaminants that impair growth, immunosuppression, and increased mortality (Collier et al. 1998).

Sewage effluent and urbanization also negatively influence fish. In streams near Toronto, Ontario, Canada, better wastewater treatment and management allowed sensitive species to colonize (Wichert 1995).

Impervious Surface Cover

A significant, consistent, and pervasive effect of urbanization is the increasing impervious surface cover within urban catchments. Impervious surface cover alters hydrology and geomorphology, increases nutrient loading, and the addition of pesticides, metals, and other contaminants to streams. These changes result in declines in stream richness and fish communities (Paul and Meyer 2001). This is in contrast to relatively unimpacted rivers leading to a common categorization in the single category of “pool” condition (Davenport et al. 2001).

Impervious surface cover is a characteristic of urban areas and is being used as a measure of environmental quality (Arnold and Gibbons 1996). In general, impervious surface cover of

10% can be tolerated by sensitive aquatic fauna but at 25%, fauna can be impacted and >25% is often non-supporting (Center for Watershed Protection 2003). In Columbus, Ohio, the health of streams declined when the amount of impervious cover exceeded 13.8%; declining biological integrity was noted in some streams at levels of urban land use as low as 4% (Miltner et al. 2004). In Wisconsin "connected imperviousness levels between 8 and 12% represented a threshold region where minor changes in urbanization could result in major changes in stream conditions" (Wang et al. 2001:255; 2003).

Because impervious surface coverage can be measured and used as an environmental indicator in urban areas (Arenold and Gibbons 1996), more research is needed for the larger planning and regulatory framework of urban development (Wang et al. 2003, Miltner et al. 2004). The Center for Watershed Protection (2003) reviewed 225 scientific articles that addressed impervious surface cover in urban areas.

Biological and Ecological Impacts

Increased runoff from urban areas and alterations in sediment yields promotes bank erosion and channel widening (Wolman 1967) that can alter bull trout habitat in Montana (Rich et al. 2003). Bull trout have increased resistance to competition by brook trout in streams with high habitat complexity and connectivity (Rich et al. 2003).

Other factors including temperature (LeBlanc et al. 1997), hydrology lowering water tables (Graffman et al. 2003), nutrients and contaminants (Myer et al. 2005) alter the ecology of riparian systems including the biology of species and fish assemblages. Black-nose dace (*Rhinichthys atratulus*) populations accompanying watershed urbanization in Maryland had increased growth rates of young but decreased age and size at maturity (Fraker et al. 2002). In

Wisconsin and Michigan, bluegill (*Lepomis macrochirus*) populations were less productive in highly urbanized areas than in undeveloped lakes (Schindler et al. 2000). Ecological parameters are correlated with human activity and studies are needed to indentify the underlying mechanisms (Limburg and Schmidt 1990).

Numerous researchers have examined how fish assemblages change or are reduced due to urbanization across the U.S. (Helms et al. 2005, Moscrip and Montgomery 1997). Agriculture (Fitzgerald et al. 1998, Helms et al. 2005), barriers (Fitzgerald et al. 1998), the frequency of floods (Moscrip and Montgomery 1997), habitat alteration (Scott et al. 1986, Rahel and Hubert 1991, Moscrip and Montgomery 1997, Walters et al. 2003, Meador et al. 2005), impoundments (Fitzgerald et al. 1998), and invasive species (Meader et al. 2005) are factors that influence species assemblages of urban aquatic landscapes. Habitat alteration was caused by cattle grazing (Rahel and Hubert 1991), impervious surface cover (Wang et al. 2000), large woody debris (Larson et al. 2001), nutrient enrichment (Scott et al. 1986, Meador et al. 2005), pollution (Tramer and Rogers 1973, Kemp and Spotila 1997), riparian loss (Weaver and Garman 1994), stream flow (Rahel and Hubert 1991, Roy et al. 2005), temperature (Rahel and Hubert 1991, Seilheimer et al. 2007), water quality (Seilheimer et al. 2007), and water velocity (Booker 2003).

Management

Because nearly every aspect of urbanization influences riparian systems and their flora and fauna, management will be challenging for restoration of these important areas. Some attempts have been made to restore marsh lands for spawning and feeding by aquatic fauna. However, the potential exists for chemical bioconcentration and biomagnifications through the aquatic community. Aquatic wetlands in urban areas may not be suited to the dual purposes of aquatic habitat enhancement and water quality improvement unless upstream controls and reduction of contaminant inputs are put into place (Helfield and Diamond 1997). Restoration of urban wetlands is a relatively new and ongoing process. Their success will depend on adequately determining the types and intensities of urban influences and to assess functional performance (Ehrenfield 2000). In urban areas, restoration of wetlands is complex because of the multiple uses that must coexist. Recreation, restoration, and research can be compatible uses of urban wetland habitats (Zedler and Leach 1998). Passive recreation would contribute to public support for habitat modification and restoration. The restoration activities would enhance the appearance of degraded sites and by designing the work as a scientific study, will add knowledge for successful coexistence of multiple users (Zedler and Leach 1998).

Quick fixes of treating the worst degraded sites are often unsuccessful. Managers should use an ecosystem approach to determine appropriate land uses and effective habitat improvement measures. As such, restoration opportunities can be identified in a multidisciplinary and interagency environment. “The focus of restoration is to improve ecosystem function rather than single resource management. Once the casual mechanisms of disturbance are identified, the appropriate restoration practices can be planned that address the limiting factors for an aquatic species or community” (Bohn and Kershner 2002:355).

Effective management of urban wetlands will require the elimination of point sources of pollution, physical channel reconstruction resulting in the duplication of undisturbed channels, and the provision of habitat for self-sustaining biotic communities (Booth 2005). Unfortunately, even the best efforts of mitigating for urban development, the sheer magnitude of anthropogenic activities suggest that there will be even further resource loss. It is imperative that ecologists understand the critical processes that cause degration (Booth and Jackson 1997). This challenge is certainly an unexploited opportunity for ecology (McDonnell and Pickett 1990).

REPTILES AND AMPHIBIANS

Most of the available data on anthropogenic influences on reptiles and amphibians is categorized under biophysical-behavioral forces. An early paper (Minton 1968) summarized the importance of altered aquatic habitats. Between 1949 and 1958, 2 species of salamanders, 6 species of anurans, 6 species of turtles, and 7 species of snakes were recorded on the boundary of Indianapolis, Indiana. Six years (1963 - 1964) later only 2 species of anurans, 1 species of turtle, and 4 species of snakes were recorded. In the earlier survey ≥ 11 species bred within the area but in the 1960s there was no evidence of amphibians breeding (Minton 1968).

> "Insofar as amphibians and reptiles are concerned, clearing of the land with removal of ground cover and underbrush affects all terrestrial species but is particularly severe for salamanders and some snakes. Modification of aquatic habitats by drainage, dredging, pollution, or removal of vegetation has serious effects on all amphibians except those whose egg and larval stages are spent on land or in small or transient collections of water. Aquatic turtles are more or less severely affected, whereas snakes suffer indirectly through deprivation of aquatic refuge and prey. Road construction and subsequent traffic can be devastating when it denies a population free access to its hibernating or breeding areas; otherwise traffic kill has little effect, particularly on smaller species. Direct killing by man affects chiefly the larger snakes and occasionally turtles. It can be a major factor in extermination of species that aggregate for breeding or hibernation, but more frequently it delivers the *coup de grace* to a population severely endangered by alteration of its habitat. Changes in food supply and in predator-prey relationships incident to urbanization are difficult to evaluate. They apparently cause some small species of amphibians and reptiles to increase in numbers in urban and suburban sites, at least temporarily. *Natrix kirtlandi* and *Thamnophis butleri* are good examples in the Midwest. On the other hand, urbanization may deprive other species of suitable prey or leave them vulnerable to excessive predation. Pesticides undoubtedly have the potential for doing severe harm to amphibian and reptile populations. Evaluation of their effect is difficult without toxicologic studies on exposed populations. (Milton 1968:115)"

The results are consistent with other early reports on the influence of development on reptiles and amphibians in the Midwest (Seibert and Hogen 1947, Conant 1951) and in San Francisco County, California (Banta and Morafka 1966). These results also reflect the negative consequences of continued urbanization on reptiles and amphibians throughout the U.S. Lizards, snakes, turtles, salamanders, and anurans have all decreased in the face of anthropogenic activities.

Lizards were surveyed along a gradient ranging from a mean density of 3.26 homes/ha to native, undisturbed landscapes in Tucson, Arizona (Germaine and Wakeling 2001). Low-moderate development densities supported lizard abundance, species richness, and evenness but as house density and paving increased beyond moderate levels, lizard assemblages decreased rapidly and the tree lizard (*Urosaurus ornatus*) was dominant (Germaine and Wakeling 2001).

In New Hampshire, snake abundance, occupancy, and richness was examined on habitat patches (range = 0.2 – 120.0 ha) in a landscape undergoing substantial land use (Kjoss and Litvaitis 2001). The species-area hypothesis (MacArthur and Wilson 1967) suggests that populations on small patches would be volatile and characterized by few species relative to larger patches. This hypothesis was supported in the study of Kjoss and Litvaitis (2001). Because populations that are dependent on early successional and shrub-dominated habitats have declined to levels where they are subjected to regional extinction, Kjoss and Litvaitis (2001) advocate the maintenance of patches >10 ha.

Suburban and urban developments are also detrimental to turtles, especially those developments with dense road networks and abundant populations of generalist predators (e.g., raccoon). Reductions in turtle populations are likely caused by reduced recruitment caused by habitat alterations (e.g., dense road networks) that will reduce or eliminate local populations (Marchand and Litvaitis 2004).

Similar results are documented for the southern two-lined salamander (*Eurycea cirrigera*) and northern dusky salamander (*Desmognathus fuscus*) in North Carolina (Price et al. 2006), spotted salamanders (*Ambystama moculatum*) in Rhode Island (Skidds et al. 2007), and California newts (*Taricha torosa*) in California (Riley et al. 2005). In North Carolina, the southern two-lined salamander and northern dusky salamanders have decreased 21 – 44% from 1972 to 2000 as urban lands have influenced stream ecosystems (Price et al. 2006). Salamanders in Rhode Island and central Pennsylvania are negatively influenced (i.e., reduced reproduction) by urbanization (Skidds et al. 2007).

Simply maintaining habitat components for salamanders is not enough. Connectivity to other populations is important. Allelic richness and heterozygosity were lower in isolated populations of eastern red-backed salamanders (*Plethodon cinereus*) due to urbanization in southern Québec compared to populations located in continuous habitat (Noël et al. 2007). In an urbanized area north of Los Angeles, California, streams in more developed watersheds have fewer native species such as California newts compared to streams not influenced by urbanization (Riley et al. 2005).

More research has been conducted on urban influences on anurans than other reptiles and amphibians. Unaltered landscapes without exotic fish that include aquatic resources and associated uplands are important for the maintenance of many amphibians (Pearl et al. 2005). However, urbanization alters these landscapes that may make them unsuitable for some species. For example, eastern spadefoot toads (*Scaphiopus holbrookii*) require wetlands and upland habitat to complete their life cycle; wetlands for reproduction and larval development and uplands for feeding and burrowing (Jansen et al. 2001). When urbanization alters the characteristics of burrowing sites, adequate habitat is eliminated and precludes the persistence of the eastern spadefoot toad in Florida (Jansen et al. 2001).

Other anurans are at risk due to the creation of road networks in and adjacent to their habitat. Vagile species like the northern leopard frog (*Rana pipens*) were negatively affected by traffic density within a 1.5 km radius of their habitat (Carr and Fahrig 2001) in Ontario, Canada. Roads serve to fragment habitats for anuran communities. When urbanization occurs, it is important to consider interconnected wetlands and upland habitats to avoid isolation and potential population reduction or local extinction (Pillsbury and Miller 2008). In central Iowa, all 7 species of anurans studied exhibited a negative association with urbanization (Pillsbury and Miller 2008) due to roads, alteration of upland habitats, predation, and increased hydroperiod. Anuran species' richness was also significantly lower in breeding ponds in urban landscapes compared to forested and agricultural landscapes in and around Ottawa, Ontario and Gatincau, Québec, Canada (Gagné and Fahrig 2007). These studies are consistent with the findings of Lehtinen et al. (1999), Ficetola and DeBernardi (2004), Riley et al. (2005) and Rubbo and Kiesecker (2005). In an urban area north of Los Angeles, California, when development exceeded 8% of the watershed, exotics increased and native species like California tree frogs (*Hyla cadaverina*) decreased (Riley et al. 2005).

Although each of these studies related to reptiles and amphibians and urbanization are categorized under biophysical-behavioral forces, the social-structural and valuation forces have led to detailed management plans in at least 1 area in the U.S., Pima County, Arizona. The county wants to minimize impacts of earth-moving, alteration of seasonal waters where amphibians breed, and other negative influences associated with urbanization. To that end, the county surveyed amphibians in the Tucson, Arizona metropolitan area including distribution and life history characteristic data. When urbanization was to occur, translocation was used to mitigate its negative effects. The suite of 13 amphibian species was translocated in a salvage-rescue translocation project for impacted areas. Over 600 amphibians of 4 species (i.e., Couch's spadefoot [*Scaphiopus couchii*], Mexican spadefoot [*Spea multiplicata*], Great Plains toad [*Bufo cognatus*], and Sonoran Desert toad [*B. alvarius*]) were salvaged and translocated.

The management plan also reviews mosquito ecology and biological control relevant to amphibian conservation to determine how urban wetland communities might be structured to avoid public health hazards. Two approaches are suggested: incorporate mosquito-eating native fish into summer rain-pool ecosystems, and manage for populations of beneficial mosquito-eating tadpoles and aquatic invertebrates.

The program in Pima County, Arizona also suggests techniques to enhance ecological restoration in infrastructure and parks and stresses the importance of monitoring (Rosen and Funicelli 2008). This is a comprehensive management plan that if implemented will minimize the influence of urbanization on this sensitive wildlife assemblage.

Biologists and managers concerned with amphibian and reptile conservation need to be aware that relatively limited development of watersheds (≥8%) can alter the habitat and viability for some species; development in the watershed alone, not necessarily in the riparian area itself or even directly upstream can negatively influence amphibians. Monitoring for amphibians and exotics should be a common practice in landscape development (Riley et al. 2005).

AVIFAUNA

Most of the research on the influence of urban and exurban development related to avifauna is categorized under biophysical-behavioral forces. However, studies range from small areas of interest (Smith and Sharp 2005) to anthropogenic influences across countries (Pidgeon et al. 2007) and continents (Clergeau et al. 2008). Most of the studies examine various aspects of community composition and density along the rural to urban gradient and others relate to specific impacts (i.e., noise) related to development. Fewer studies provide specific management recommendations (Marzluff and Ewing 2001).

Many of the studies were designed to measure various characteristics associated with avifauna communities and altered landscapes from urbanization and most covered large areas. In all cases, native avifauna benefited from native vegetation in undeveloped land or surrounded by undeveloped lands. In Portland, Oregon (Hennings and Edge 2003), Tucson, Arizona (Germaine et al. 1998), Pitkin County, Colorado (Odell and Knight 2001), Jackson Hole, Wyoming (Smith and Wachob 2006), and Tulsa, Oklahoma (Boren et al. 1999) urbanization influenced avifauna communities.

In Oregon, breeding bird and plant communities were surveyed along 54 streams in the Portland metropolitan region. Total and exotic bird abundance was higher in narrow forests and native bird abundance was greater in narrow forests surrounded by undeveloped lands. Native species richness and diversity were greater in less-developed areas (Hennings and Edge 2003). Neotropical migrant abundance, richness, and diversity were greater in areas with fewer roads and open-canopied landscapes (Hennings and Edge 2003). The negative influence of roads on avifauna was also documented in Spain by Palomino and Carrascal (2007).

In a gradient study in Tucson, Arizona (Germaine et al. 1998), residential development was beneficial to non-native birds. Non-native birds are at a disadvantage when competing for resources in natural habitats with native birds (Green 1984) and their resource requirements are best met in urban environments (Emlen 1974). Native breeding birds increased with increases in the array of land cover types but Germaine et al. (1998) did not determine how birds were distributed among cover types or how cover types met the resource needs of birds. However, they did determine that anthropogenic influences altered physiognomic, floristic, and spatial habitat alterations that influence bird assemblages (Germaine et al. 1998). If some species (e.g., loggerhead shrikes [*Lanius ludovicianus*]) are left unmolested and sufficient open areas are maintained within the urban environment, the species may not be negatively influenced by urbanization (Boal et al. 2004).

In Pitkin County, Colorado, the avian densities reported were not statistically different between high (1.04 ± 0.67 houses/ha) and low (0.095 ± 0.083 houses/ha) density development but were statistically different from undeveloped sites. The American robin (*Turdus migratorius*), black-billed magpie (*Pica pica*), brown-headed cowbird (*Molothrus ater*), European starling (*Sturnus vulgaris*), house wren (*Troglodytes aedo*), and mountain bluebird (*Sialia currucoides*) had higher densities in developments of high housing density and 8 species had significantly reduced densities in high-density housing densities: black-caped chickadee (*Poecile atricapillus*), blue-gray gnat catcher (*Polioptila caerulea*), black-headed grosbeak (*Pheucticus melanocephalus*), dusky flycatcher (*Empidonax oberholseri*), green-tailed towhee (*Pipilo chlorurus*), orange-crowned warbler (*Vermivora celata*), plumbeous vireo (*Vireo plumbeus*), and Virginia's warbler (*Vermivora virginiae*).

In another study conducted in the west, Smith and Wachob (2006) sampled bird community parameters and habitat variables at microhabitat, macrohabitat, and landscape scales along a residential development gradient within the Snake River riparian corridor in Jackson Hole, Wyoming. Overall, species richness and diversity declined as residential development increased. The most negatively impacted species were Neotropical migrants that declined in proportional representation on forested plots as residential development densities increased (Smith and Wachob 2006).

Changes in vegetation cover also altered avian communities in Tulsa, Oklahoma (Boren et al. 1999). The loss of Neotropical migrants and increased number of generalist species in high density rural populations was related to decreased native vegetation, road development, and increased landscape fragmentation. Other studies that documented avifauna changes related to urbanization were consistent with the above works.

In Colorado, grasslands along an urbanizing region at the western edge of the North American Great Plains, rough-legged hawk (*Buteo lagopus*) populations declined by nearly 75% between 1971 and 2003 and red-tailed hawk populations (*B. jamaicensis*) nearly tripled as the human population steadily increased over the 33 years. Rough-legged hawks avoided human development in preference for treeless grassland. Red-tailed hawks adjusted to utility poles and areas closer to buildings and roads, which allowed an increase in population size. The rough-legged hawk and other grassland species such as mountain plover (*Charadrius montanus*), long-billed curlew (*Numenius americanus*), burrowing owl (*Athene cunicularia*), common night hawk (*Chordeiles minor*), loggerhead strike, and lark bunting (*Calamospiza melanocorys*) declined as a result of urbanization and represent the challenge facing managers as landscapes are manipulated (Schmidt and Bock 2005).

Open space grasslands can support sizeable populations of diurnal raptors (e.g., red-tailed hawks, Swainson's hawks [*B. swainsoni*], American kestrel [*Falco sparverius*], bald eagles [*Haliaeetus leucocephalus*], golden eagles [*Aquila chrysaetos*]) if prey populations are available. In Denver, Colorado, black-tailed prairie dog (*Cynomys ludovicianus*) towns received heavy use by wintering raptors (Weber 2004). However, some species are highly sensitive to landscape urbanization (e.g., bald eagle, ferruginous hawk [*B. regalis*], rough-legged hawk, prairie falcon [*Falco mexicanus*]) when <8% is urbanized (Berry et al. 1998). Raptor numbers and the number of species were lowest in a study in Washington near Spokane, but both consistently increased as distance from the city center increased (Ferguson 2004). Bird species also declined in riparian woodlands along a gradient of urbanization in Santa Clara Valley, California. Decreases were attributed to increased development (i.e., buildings, bridges), decreased width of riparian habitat, and decreases in native vegetation (Rottenborn 1999). Landscapes directly influenced by urbanization caused a decline in bird density but lands adjacent to urbanized areas were influenced also (Rottenborn 1999). These responses were documented also in Santa Clara County, California where bird ecology was examined across a gradient of undisturbed land to highly developed landscapes including biological preserves, recreational areas, golf courses, residential neighborhoods, office parks, and business districts. Birds in undisturbed sites were predominately native species but were replaced by exotic and invasive species in the business district (Blair 1996). These patterns were related to shifts in habitat structure that occurred along the gradient. The area had been dominated by oak (*Quercus* spp.) woodlands. The patterns described by Blair (1996) as woodlands were transformed to urban sites and suggests that any land use development is detrimental to avifauna. "Even minor perturbations, such as the grazing that formerly occurred in the open-space recreation area, apparently lead to a loss of species. Moreover, species that disappear at the lightly disturbed sites do not reappear at some more highly disturbed site" (Blair 1996:517).

Contrasting bird abundance across continents also demonstrated the influence of human development. Avifauna were compared in the cities of Quebec, Canada and Rennes, France that had landscapes that changed from rural to urban (Clergeau et al. 1998). Diversity of avifauna declined across the gradient as development increased in both cities. Because the surrounding landscape did not explain the variation of species in the city, Clergeau et al. (1998) suggests that urban environments be viewed as new ecological landscapes rather than degraded environments. If that approach is adopted, birds could be regrouped into 2 major categories: omnivorous species adopted to the urban environment and its food resources (i.e., garbage), and the species that find resources in the urban environment they normally exploit in their usual habitat (Clergeau et al. 1998).

Most of the researchers that examined bird communities across the rural to urban gradient used a variety of plots randomly placed within the gradient and systematically documented bird composition and density within plots to obtain their data. These designs are appropriate for relatively small study areas but other techniques are used for larger landscapes.

To examine forest bird species richness associated with housing across the United States, Pidgen et al. (2007) used the North American Breeding Bird Survey (Sauer et al. 2003) as their data source for bird distribution and abundance. This survey consists of an annual monitoring system that censuses birds on permanent monitoring plots administered by the U.S. Fish and Wildlife Service, and provides data on the relative abundance of birds across the 48 conterminous U.S. and southern Canada. The survey has been conducted since 1966. Data on humans and development were determined by examining the bounding rectangle that encompassed the survey route. Bounding rectangles were centered on survey routes and developed by extending half the length of the route to define 1,200 km^2 landscapes. The human development within these boundaries was determined from the 1990 and 2000 U.S. Decennial Census (i.e., human density, housing density, household density, seasonal housing density) and the National Land Cover Data (i.e., landscape composition). From these data, Pidgeon et al.

(2007) investigated species richness of all forest birds versus the predictor variables of housing density in 2000 and the abundance of forest, seminatural, and intensive use land cover. These data were used to develop models that showed housing density and residential land cover as significant predictors of forest bird species richness. Results were the same for smaller scale studies; urbanization decreased species richness.

Other approaches have been used to examine avifauna related to urbanization (i.e., capture-recapture models; Cam et al. 2000) but the results are generally the same; human settlement at some levels may limit avifauna by reducing resources, increasing nest predation, competition for resources, and brood parasitism (Marzluff and Ewing 2001).

However, in exurban situations some species increase in abundance from increased environment heterogeneity and biotic resources (Fraterrigo and Wiens 2005). More urban studies provided evidence of negative relationships between human development and species richness (Batten 1972, Emlen 1974, Huhtalo and Järvinen 1977, Beissingeo and Osborne 1982, Bezzel 1984, Rapport et al. 1985, Jakimäki and Suhonen 1993, Zalewski 1994, Clergeau et al. 1998). Others provide evidence that intermediate levels of urban development cause peaks in avifauna species richness (Jakimäki and Suhonen, 1993, Blair 1996). In the urban gradient studies, the decline in avifauna was often associated with broad-scale vegetation patterns and ground level habitat patterns that were not associated with the development gradient studied in an exurban landscape (Fraterrigo and Wiens 2005). The bird community results of the study in exurban development of north-central Colorado (Fraterrigo and Wiens 2005) were consistent with the concept that "…human settlement can act as an intermediate disturbance on the landscape, … and that habitat heterogeneity can enhance avian diversity (Fraterrigo and Wiens 2005:271)". Species that increase with increases in building density are habitat generalists (i.e., house sparrows [*Passer domesticus*], common grackles [*Quiscalus quiscula*]) that may benefit from resource supplements provided by development. Habitat specialists did not increase with the exurban environments (Fraterrigo and Wiens 2005).

Urbanization influences life history characteristics of wildlife in numerous ways besides abundance and distribution. Several authors have examined the influence of urban noise (Wood and Yezerimcec 2006), nest predation (Blair 2004), and reproduction (Millsap and Bear 2000, Morrison and Bolger 2002, Thorington and Bowman 2003, Phillips et al. 2005) in relation to urbanization and birds.

Sound pressure levels from anthropogenic activities have the potential to mask bird songs, which could alter the ability of males to attract mates. Song sparrows (*Melospiza melodia*) singing at locations with higher sound pressure levels exhibited higher-frequency low notes and had less amplitude in the low frequency range of their songs (1-4 KHz) where most anthropogenic sound pressure levels occurred. How this will influence the population is under study but is an environmental variable that warrants further study (Wood and Yezerinac 2006).

Urbanization also influences nest predation rates. Predation causes most nest failures in birds (Thorington and Bowman 2004). In south-central Florida, Thorington and Bowman (2003) examined the influence of nest predation using artificial nests in a suburban matrix. Nest predation was highest at high housing density (>40 houses/40/ha) and lowest at low density (<20houses/40h); nest predation may increase with human housing density (Thorington and Bowman 2003). Others have documented predation decreasing on artificial nests with increasing urbanization (Blair 2004). However, these data did not reflect the nesting success of birds that did not increase with urbanization. Blair (2004) classified birds along the urban gradient as urban exploiters or urban avoiders from the individual, species, community, landscape, and global perspective. Urban exploiters successfully reproduce, invade locally, have multiple broods, use heterogeneous patches, and ubiquitous species invade, respectively leading to homogenization of communities. On the other hand, urban avoiders do not successfully reproduce, become locally extinct, have single broods, require homogeneous patches, and require maintenance of native species, respectively, leading to conservation.

Blair's (2004) generalization needs to be considered cautiously as species and habitats are important considerations when discussing or predicting fragmentation effects. For example, in Southern California, Morrison and Bolger (2002) examined whether rufous-crowned sparrows (*Aimophila ruficeps*) were subjected to higher nest predation at the edge of urban sites compared to sites within the habitat interior from 1997 to 1999. Total reproductive output did not differ between edge and interior in any year. However, wood thrushes (*Hylocichla mustelina*) in southern Ontario (Phillips et al. 2005) and burrowing owls in Florida (Millsap and Bear 2000) had reduced reproductive rates to urbanization. Wood thrushes breeding in woodlots with embedded houses experienced higher rates of parasitism by brown-headed cowbirds than wood thrushes in woodlots with homes ≤ 100m of the forest edge, or undeveloped woodlots. However, nest predation did not increase in developed woodlots in Ottawa (Phillips et al. 2005). The effects of housing developments on wood thrushes may be region specific or depend on cowbird density.

Burrowing owl populations benefited from high prey densities around homes in Florida but this advantage was offset by increased human-caused nest failures and declines in the number of young fledged. Nest site density of owls increased until 45 – 60% of lots were developed before decreasing and the number of young fledged decreased as development increased above 60% (Millsap and Bear 2000).

Threshold distances from urbanization that avifauna can tolerate without alterations in their life history characteristics would be useful in urban planning but this has not been studied by many. However, results from a study in central Spain (Palomino and Carrascal 2007) suggested that “As a general rule, the significant threshold distances in the models averaged 400 m for cities, and 300 m for the roads, although these figures varied among different bird populations” (Palomino and Carrascal 2007). These buffer distances in a densely developed landscape matrix suggest severe fragmentation of suitable habitat for native avifauna.

It is not surprising that habitat alteration influences bird communities even in small patches like gardens. Gardens were examined in 10 suburbs of Hobart, Tasmania, Australia (Daniels and Kirkpatrick 2006) and although use varied, native birds used exotic plants but exotic birds largely used exotic plants. Because gardens can be designed and managed to favor particular species, gardeners have a potential role in the conservation of native birds. However, larger landscape alterations such as golf courses may not be as beneficial for native birds. Number of species, number of Neotropical migrant species, and degree of conservation concern of the species present were higher in less altered golf courses in coastal South Carolina and were significantly influenced by percent of forested area (Jones et al. 2005). Enhancing avian habitat is possible through increasing the amount of forest and reducing the amount of managed turf grass (Jones et al. 2005). Others have also demonstrated that the size of the area considered, amount of natural vegetation, and percent of urbanization and natural land are all important factors in explaining winter bird use in Ontario, Canada (Smith 2007). Unfortunately, estimates of threshold sizes for habitat islands have not been determined (Beissinger and Osborne 1982). It has been established, however, that the habitat structure and population-supressing factors in developed areas creates habitat for only a few bird species (Beissinger and Osborne 1982).

Fortunately, when some altered landscapes (i.e., grasslands) are restored, the habitats they provide contain bird assemblages similar to those in native prairie habitats, suggesting that restored grasslands may provide similar habitat for most grassland birds (Fletcher and Koford 2002).

Management for Avifauna in Altered Landscapes

The problems with urbanization and avifauna have been clearly established but fewer recommendations have been made as to how urban, suburban, and exurban centers can exist and minimize their influence on birds. Many authors that address avifauna in urban landscapes provide suggestions to enhance native species but they are unproven suggestions that may, or may not improve avian assemblages in the face of landscape alteration. Suggestions for management include educating residents about their impacts (Millsap and Bear 2000, Fraterrigo and Wiens 2005), developing buffer zones around sensitive areas (Millsap and Bear 2000, Rottenborn 1999), maintaining native plant communities (Germaine et al. 1998, Boren et al. 1999), maintaining and enhancing riparian corridors (Germaine et al. 1998, Hennings and Edge 2003), maintaining native patches of vegetation >1ha, and managing for sensitive species (Germaine et al. 1998). Because natural landscapes are important, Blair (1996) and Odell and Knight (2001) recommend that cluster developments will have less of an impact than development that traverses the landscape. Clustered development has been suggested as an important conservation tool to maintain and enhance wildlife habitats. Theobald et al. (1997) conducted a study in Summit County, Colorado, which demonstrated that clustered developments reduce the negative impacts on wildlife habitat. However, this is a new concept that has received limited attention. Others contrasted the influences of clustered and dispersed housing (1house / 2-16 ha) with undeveloped areas in Boulder, Colorado (Lenth et al. 2006). They contrasted densities of songbirds, nest density, and survival of ground-nesting birds, presence of mammals, and percent cover and proportion of native and exotic flora as indicators to assess conservation value. Both types of housing had significantly higher densities of exotic and human-commensal species (e.g., dogs, common grackles, European starlings, American robins, red-winged blackbirds, doves, killdeer [*Charadrius vociferus*]) and significantly lower densities of native and human-sensitive species (e.g., western meadowlarks [*Sturnella neglecta*], vesper sparrows [*Pooelcetes gramineus*], field mice [*Peromyscus* spp.]) than undeveloped areas. Additional research is needed to determine the ecological value of clustered housing on broader scales (Lenth et al. 2006).

Because the majority of the nation's forests are in private ownership, national conservation plans need to incorporate housing in their management strategies. "The recent development of several conservation plans at the national and international scale demonstrates the general buy-in of the conservation community to this idea" (Pidgeon et al. 2007:2008).

To maximize golf courses as habitat for avifauna, wildlife should be considered before development begins to: increase forest patches, reduce managed turf grass, maintain as much native vegetation as possible, plant native vegetation in disturbed sites, reduce mowing, and maintain natural waters (Jones et al. 2005).

Clearly, management needs to be considered and Marzluff and Ewing (2001) offer 15 specific recommendations that would improve the suitability of reserves for birds.

1. Increase the foliage height diversity within fragments.
2. Maintain native vegetation and deadwood in the fragment.
3. Mange the landscape surrounding the fragment (matrix), not just the fragment.

4. Design buffers that reduce penetration of undesirable agents from the matrix.
5. Recognize that human activity is not compatible with interior conditions.
6. Make the matrix more like the native habitat fragments.
7. Actively manage mammal populations in fragments.
8. Discourage open lawns on public and private property.
9. Provide statutory recognition of the value of complexes of small watersheds.
10. Integrate urban parks into the native habitat reserve system.
11. Anticipate urbanization and see creative ways to increase native habitat and mange it collectively.
12. Reduce the growing effects of urbanization on once remote natural areas.
13. Realize that fragments may be best suited to conserve only a few species.
14. Develop monitoring programs that monitor fitness.
15. Develop a new educational paradigm.

Eight research needs are suggested by Marzluff and Ewing (2001) to guide the way toward enhancing habitats for birds in urban areas.

1. Are corridors used by dispersing birds and do they facilitate the functioning of metapopulations?
2. Does increasing native vegetation of the matrix help?
3. How does the pattern of housing affect avian population viability in surrounding fragments?
4. How do we design effective buffers that shield birds in fragments from the disturbance of the matrix?
5. How does urbanization affect insect communities?

6. Is it possible to use some non-native plant species to reduce invasions by species known to be disruptive to ecosystem function?
7. Can fragments of native habitat in urbanized landscapes make tenable contributions to avian conservation?
8. What are effective means of encouraging citizens to conserve birds and their habitats and reduce their impacts?

MAMMALS

Small Mammals

Urbanization and exurbanization vary in their influence on small mammals. Cottontail rabbits (*Sylvilagus audubonii, S. floridanus*) were counted in plots grazed by livestock embedded in housing developments, or both, or neither in southwestern Arizona (Bock et al. 2006*a*). The number of cottontails documented was positively correlated with the number of homes near plots. Cottontails benefited from exurban development due to increased cover provided by structures and landscaping especially in open grasslands with limited natural cover (Bock et al. 2006*a*). Rodents however, were negatively influenced by livestock grazing or its effects on vegetative ground cover. Exurban development did not influence rodent abundance or diversity. If housing densities are low and embedded in a matrix of natural vegetation with limited grazing, exurban development can maintain rich assemblages of grassland and savannah rodents (Bock et al. 2006*b*). In contrast, native rodents as a group were captured more often in interior plots compared to edge plots in Boulder, Colorado. Plots that had the highest capture rates of native species were in landscapes <10% suburbanized. The authors concluded that proximity to suburban landscapes had a negative influence on the abundance of native rodents in open-space grasslands (Bock et al. 2002).

The study was not designed to understand why rodents were relatively scarce near suburban edges on open-space grasslands. However, they did eliminate habitat, competition, and patch size as likely causes. Likely causes for the differences reported is predation by domestic cats (Bock et al. 2002).

Species abundance and richness has been the most common metric used to document how urbanization influences small mammals. Behavior has also been used to measure separate and combined impacts of anthropogenic actions at the individual and population levels. With the use of giving up densities, Bowers and Breland (1996) determined that gray squirrels (*Sciurus carolinensis*) living close to humans were more limited by food or less sensitive to risk of predation than squirrels in more natural areas.

In Pennsylvania, small mammal (e.g., eastern chipmunks [*Tamias striatus*], white-footed mice [*Peromyscus leucopus*], deer mice [*P. maniculatus*], woodland jumping mice [*Napaeozapus insignis*]) species richness was lowest in parks containing manicured landscapes surrounded by human-modified landscapes. Only 1-2 species of small mammals were in mowed landscapes (Mahan and O'Connell 2005). These authors recommended leaving 10-15 m strips that are not mowed along streams and planting native trees along stream corridors to encourage small mammals in suburban and urban parks. Similar results have been reported world wide. For example, in Melbourne, Australia, small ground-dwelling mammals were the group of mammals most negatively affected by urbanization; only 2 out of 15 species has a >10% probability of persisting (van der Ree and McCarthy 2005). Because Melbourne will continue to expand, van der Ree and McCarthy (2005) recommend that state and local governments design and adopt a comprehensive strategy to manage habitat networks with cross jurisdictional boundaries and include Melbourne to enhance viability of mammalian populations.

Chiroptera

Many of the studies designed to examine the influence of human development on bats have approached the question by contrasting bat use of habitat within rural to urban gradients. In residential areas in California, Myotis (*Myotis yumanensis*) preferred large trees ($\bar{x}$ diameter = 115 cm) for roosting, close to water with forest cover in the surrounding 100 m radius. As urbanization encroaches into these areas, managers need to preserve large trees and forested parkland, especially along stream corridors to help maintain bat populations in urbanizing landscapes (Evelyn et al. 2004). Indiana bats (*Myotis sodalis*), a federally listed endangered bat, also prefers woodlands over developed habitats for roosting and foraging (Sparks et al. 2005).

Urban habitats were not used as much compared to agricultural landscapes for foraging by evening bats (*Nycticeius humeralis*) or big brown bats (*Eptesicus fuscus*) in Indiana. However, big brown bats roosted in man-made structures (i.e., urbanization) but evening bats roosted in tree cavities in woodlots. Evening bats are likely more sensitive to suburban development near roosts than big brown bats (Duchamp et al. 2004).

Urban settings do not always represent negative habitats for bats. Five species of bats and the *Myotis* group were monitored in the Chicago metropolitan area. All bats identified exhibited positive relationships within the urban marix (Gehrt and Chelsvig 2004). They were also detected more frequently in urban areas than in more rural habitat fragments. “Urban areas may represent islands of habitat for some bats within larger landscapes dominated by intensive agriculture. Thus, the nature of the relationship between urbanization and bats is probably dependent on context at the macrogeographical scale and local habitat quality within the urban matrix" (Gehrt and Chelsvig 2004:625).

Mesopredators

Mesopredators are common urban inhabitants but little is known about their demographic response to urbanization (Prange and Gehrt 2004). Because raccoons exploit anthropogenic resources efficiently, they have become common residents of urban areas (Prange and Gehrt 2004). Opossums (*Didelphis virginiana*) and striped skunks (*Mephitis mephitis*) were also common in developed areas. In north eastern Illinois, density of raccoons was greater in urban plots than rural plots throughout the year and at urbanized sites raccoons may have had larger litter sizes. Adult female survival was also higher in urban sites in the absence of disease (Prange et al. 2003). Raccoons in urban areas had fewer mortality sources (disease was greatest) and those residing in suburban and rural sites had the most (road-kills were the greatest). There also may have been greater site fidelity at urbanized sites (Prange et al. 2003). Because raccoons are capable of quickly repopulating an area after the resident population has been reduced, control methods will have to be continuous.

As raccoons, and other mesocarnivores, become abundant, society often views them as pests. They create nuisance-related problems and may transmit diseases and parasites to humans and domestic animals. Increased survival, higher annual recruitment, and increased site fidelity contribute to increased densities of raccoons in urbanized areas. To efficiently manage the population, direct control measures (e.g., trapping and removal) and reduction or elimination of anthropogenic food sources will be required (Prange et al. 2003). However, direct removal of overabundant species may not be economically feasible and could be socially unacceptable (Goodrich and Buskirk 1995).

Domestic cats are an often overlooked influence on fauna in urban, suburban, and exurban landscapes, but they have a significant impact on native rodents and birds (Hawkins et al. 2004). In one study in southeastern Michigan, 25% of landowners owned outdoor cats, which killed ≤1.4 birds/cat/week. Over 23 species were killed including species of conservation concern (Lepczyk et al. 2004). There were ≤3,100 cats across the landscape which killed ≤47,000 birds during the breeding season (~1 bird killed/km/day). Even taken conservatively, these data (Lepczyk et al. 2004) suggest that cat predation plays an important role in fluctuations of bird populations.

Devices have been developed that can reduce the efficiency of domestic cats as predators (i.e., collar-mounted pounce protectors, the Cat-Bib™). However, they need to be worn consistently by cats and most cat owners were reluctant to maintain this cat apparel (Calver et al. 2007).

Overabundant wildlife and cats in urbanized landscapes are clearly an increasing problem. More effective programs have to be developed for the management of urbanized species (DeStefano and DeGraaf 2003).

Larger Predators

The problem

Addressing the influence of suburban and exurban development on other large predators (e.g., wolves [*Canis lupus*], mountain lions [*Puma concolor*], wolverines [*Gulo gulo*], grizzly bears[*Ursus arctos horribilis*]) has not generated as much concern as other predators because they are primarily wilderness species (Leopold 1933) that do not benefit from close associations with humans. However, as the human population increases and continually alters landscapes, habitats for larger predators will be influenced. Successful translocations of larger predators into historic habitats will be challenging especially when they encroach upon exurban developments. Successful conservation is compounded because larger predators require large landscapes that often traverse public and private lands and international borders (i.e. gray wolf, jaguar [*Panthera onca*]). Maintaining habitats and connectivity between habitats is challenging (Harrison and Chapin 1998). For example, a highway in Spain is a significant barrier to wolf movement, which may be isolating 2 subpopulations (Rodriguez-Freire and Cricente-Maseda 2008). An interstate freeway through Tucson, Arizona also serves as a barrier for mountain lion movement (K. Nicholson, graduate research assistant, and P.R. Krausman, unpublished data). Mountain lions in California that dispersed, used corridors associated with natural travel routes, cover and underpasses to avoid high-speed road crossings and limited artificial lighting with < 1 house / 16 ha (Beier 1995).

When examining resiliency of larger predators, Weaver et al. (1996) examined plasticity in diet and food availability, demographic compensation to mitigate increased exploitation, and dispersal to maintain connectivity with fragmented populations. Wolves "possess resiliency to modest levels of human disturbance of habitat and populations" (Weaver et al. 1996:964). Grizzly bears and wolverines possess less resilience because of requirements for forage, low productivity, and philopatry of females to maternal home ranges (grizzly bears). Humans have altered these life history characteristics causing widespread declines and as exurban development spreads, conflicts will increase.

Controversy escalates as predators kill livestock or threaten human safety (Geist 2008). As a result, numerous authors call for management by maintaining refugia on public and private lands (Carroll et al. 2003) that encompass the full spectrum of required habitats that are connected to other refugia through landscape linkages (Weaver et al. 1996).

Valuation and institutional-regulatory forces have provided the mechanism for restoration of larger predators and Rasker and Hackman (1996) proposed the hypotheses that the protection of refugia that sustains wild carnivores does not have a detrimental effect on local or regional economics. Environmental protection and economic development are complementary goals (Rasker and Hackman 1996). Agencies, organizations, and the public need to work together for successful conservation (Harrison and Chapin 1998), especially of larger carnivores. Influences of exurban development have not been examined as thoroughly with larger predators but lessons learned from other conflicts can certainly be used as models as these species and human habitats merge on the landscapes.

Coyotes

The problem

Coyotes have established populations in habitats dominated by humans across North America (Atkinson and Shackleton 1991) likely because of habitats and unnatural food provided. Establishment and increases in population sizes are specially prevalent in suburban and exurban areas that contribute to habitat fragmentation. Unfortunately life history characteristics of coyotes in human dominated landscapes are not well documented but when coyotes and humans use shared habitats, attacks on humans increase creating concern for human safety and property. Similar problems exist for American black bears (*Ursus americanus*) and other carnivores often because of human encroachment into their habitat. Anthropogenic development often creates habitat for coyotes and serves as an undesired attractant.

Much of the available literature on coyotes addresses and contrasts their life history characteristics with coyotes in rural areas and wildlands and addresses problems coyotes cause in human dominated landscapes. Few studies have thoroughly investigated mechanisms to minimize coyote-human interactions.

Biophysical-behavioral forces

Information about coyotes in urban settings has increased from limited descriptions and observations (Gill 1965, Andelt and Mahan 1980) over 3 decades ago to more detailed studies of habitat, activity, survival, diet, and reproduction in human dominated habitats. Most of these studies are strictly biological and provide limited data as to how problems between coyotes and humans can be minimized or mitigated.

Home-range sizes.--As with most studies of home-range size in animals, results vary due to techniques used to calculate sizes, habitat, sample size, and other factors (Table 1). The 95% minimum convex polygon home-range size of resident coyotes in western North America varies from 1.1 km^2 in urban areas, Los Angeles, California (Shargo 1988) to 118 km^2 in Washington (Springer 1982). Mean home-range sizes for coyotes in habitats that include suburban and exurban landscapes ranged from 7.7 ± 4.2 (SE) km^2 in Canada (Atkinson and Shackleton 1991) to 26.8 ± 5.1 km^2 in Tucson, Arizona (Grubbs and Krausman 2009*a*).

Habitats in urban areas.--Although coyotes continue to increase in urban areas and expand their range, they still heavily rely on cover. Of the several habitat studies of coyotes in urban areas conducted, most have involved monitoring radiocollared animals but others have

Table 1. Mean home-range size of resident coyotes that use suburban and exurban landscapes in Canada and the U.S.

Home-range size (km^2)	Method of calculation[a]	Sex	Location	Landscape	Source
7.7±4.2(SE)	95% MCP	M	Lower Fraser Valley, BC, Canada	Rural-urban gradient	Atkinson and Shackleton (1991)
17.0±20.7	95% MCP	F	Lower Fraser Valley, BC, Canada	Rural-urban gradient	Atkinson and Shackleton (1991)
10.8±11.2	95% MCP	M/F	Lower Fraser Valley, BC, Canada	Rural-urban gradient	Atkinson and Shackleton (1991)
12.6±3.5	95% MCP	M/F	Tucson, Arizona	Rural-urban gradient	Grinder and Krausman (2001)
26.8±5.1	95% FK	M/F	Tucson, Arizona	Urban	Grubbs and Krausman (2009*a*)
15.4±6.3	MCP	M/F	Tucson, Arizona	Suburban	Bounds and Shaw (1997)
2.97-23.48	ADK	M/F	West-central Indiana	Rural-urban gradient	Atwood et al. (2004)

[a] MCP = minimum convex polygon, FK = fixed kernel, ADK = adaptive kernel method.

used public observations and scent stations to evaluate habitats. Broader studies have examined carnivores across a spectrum of habitats that essentially constitute an urban to rural gradient (Crooks 2002, Randa and Yunger 2006).

In an examination of 29 urban habitat fragments in coastal southern California ranging from those surrounded by human-modified landscapes to mesa-top habitat and others dominated with ornamental plants Crooks (2002), examined carnivore distribution based on track plots. Coyotes were present in 26 of 29 sites. The estimated area for detecting a 50% probability of occurrence was 1 ha; larger bodied carnivores require areas that will eventually disappear if not connected to other patches (Crooks 2002). The only patches examined that did not have coyotes were 12 and 2 ha in size. Due to their behavioral plasticity, allowing coyotes to exist in a variety of disturbed sites limits their utility as an indicator of connectivity (Crooks 2002).

In a similar study, Randa and Yunger (2006) examined carnivores across 47 sites in a rural-urban gradient in the Chicago metropolitan area counting carnivore tracks at scent stations. Sites varied from those with higher amounts of human influences (e.g., people, industry, commercial, high road densities; urban) to those further removed from the city (e.g., agriculture; rural). Coyotes were common across the entire gradient: in 6 of 18 urban sites and 18 of 29 rural sites and in all habitats (i.e., mowed lawn, non-native grassland, old field, prairie, row crops, shrubland, woodland, woodland edge) except lawn (Randa and Yunger 2006). However, coyotes should occupy more rural than urban sites. The ability of coyotes to make large scale movements has likely usurped the effects of habitat fragmentation (Randa and Yunger 2006).

Results of these broad scale studies (Crooks 2002, Randa and Yunger 2006) are generally supported by specific examinations of coyote habitats in urban areas. In a series of papers Quinn (1991, 1995, 1997) reported that coyotes in Seattle, Washington preferred undisturbed habitats and used forest and shrub habitat for hiding cover and used densely mixed vegetation (with forest and shrub vegetation) more than other habitats. These areas were adjacent to rural areas, provided escape, prey, and cover. Quinn (1991, 1995, 1997*b*) used radiocollared coyotes and public sightings to describe coyote habitat associated with urbanization due to the abundant patches of vegetation.

In west-central Indiana, Atwood et al. (2004) examined habitat use of coyotes along a rural-exurban-suburban gradient consisting of forest, grassland, fencerows, grassy drainage ditches, agriculture matrix, and human development including commercial and residential. Twelve percent of the 250 km^2 study area included human developments with 70% in agriculture production. Within their home ranges, coyotes used fence, ditch, and grassland. Forested habitat was used more than the agriculture or urban matrix. All coyotes preferred corridors when present (Atwood et al. 2004).

Habitat of coyotes in Tucson, Arizona was examined in 2 different studies (Grinder and Krausman 2001, Grubbs and Krausman 2009*b*). In the first study, coyotes were captured and radiocollared throughout the city including areas adjacent to wildlands (Grinder and Krausman 2001). In this situation, coyotes concentrated in >30% of at least 1 of 3 habitats: natural areas (i.e., state and federal parks, private open space, and cropland; <1 house/ha), parks (i.e., schools, military grounds, cemeteries, zoos, golf courses, small parks, stables), and residential areas (i.e., >1 house/ha). These patch types contained food, cover, and breeding sites.

The study of Grubbs and Krausman (2009*a*) used different habitat classifications because all collared animals were entirely within the city and did not use adjacent wildlands. These coyotes used washes, medium (2-7 residences/ha), or low density (< 1 residence/ha) residential areas more than expected by chance alone. All coyotes avoided high density residential areas

(>7 residence/ha), commercial areas, and roads. Washes and medium density residential areas offer an abundance of shade and cover; important items in times of high human activity. The presence of washes throughout Tucson, and especially in dense residential areas, is likely a reason coyotes persist throughout the city (Grubbs and Krausman 2009*a*).

Diet.--Coyotes in urban areas exhibit the same catholic diets as animals in wildlands. In the Lower Fraser Valley, British Columbia, coyotes primarily consumed small rodents (70.2%), lagomorphs (8.12%), and other mammals (4.7%) (i.e., raccoon, opossum, muskrat [*Ondatra zibethicus*], and black-tailed deer [*Odocoileus hemionus columbianus*]). They also consumed birds (2.0%), and domestic stock (4.3%) (i.e., sheep, cattle, pigs, chickens). Plant material (10.3%) included plums, apples, grasses, and holly. Other items documented in the diet were insects, paper, cloth, plastic, and rubber (Atkinson and Shackleton 1991).

Across a rural to urban gradient in Western Washington, Quinn (1997*a*) reported changed foods, particularly mammals, as human caused land use patterns changed. Fruits and mammals represented the largest classes of food items in all habitats. Voles (*Microtus* spp.) were the most abundant mammalian food item (41.7%) in mixed agricultural-resident habitat, which shifted to house cat (13.1%) and squirrel (7.8%) in residential areas. Cats were a common source of food for coyotes in Tucson, Arizona also (Grubbs and Krausman 2009*b*).

Coyote diets were also examined across a rural-urban gradient in the Santa Monica Mountains, California (Fedriani et al. 2001) and their data supported results of Quinn (1997*a*); consumption of anthropogenic foods by coyotes varied according to human density. The area with more people provided 14-25% of diet to coyotes as anthropogenic foods (i.e., trash, livestock, and domestic fruit), the area with the fewest people provided 0-3% of items and the intermediate level of human activity provided 4-6% of anthropogenic foods. Similar findings were reported by McClure et al. (1995) in Tucson, Arizona; anthropogenic foods replaced natural foods. Others also documented increases in anthropogenic foods in coyotes using urban areas (MacCracken 1982, Shargo 1998).

The diets of coyotes in urban areas have been studied rarely, possibly because tools to discriminate coyote feces from those of domestic dogs have not been available. Criteria that have been used to discriminate between dog and coyote scats include size (i.e., scats >5 mm in diameter as coyote; Atkinson and Shackleton 1991) and collecting scats in areas of relatively high coyote density (Quinn 1992, 1995, 1997*a*). However, these techniques are not accurate (Krausman et al. 2006) and to ensure accuracy, DNA analysis should be used. The role that available food plays in foraging strategies of coyotes needs to be examined thoroughly to understand its ecology and relationships to dogs and humans in urban areas. DNA analysis used for scat identification provides a mechanism to distinguish canid scats (Krausman et al. 2006).

Activity.--Activity patterns are plastic and not catholic. Some researchers documented coyote activity to be crepuscular with extensive nocturnal movements (Gipson and Sealander 1972, Andelt and Gipson 1979, Laundre and Keller 1981, Woodruff and Keller 1982). Others have documented coyote activity to peak at different times of day and night (Major and Sherburne 1987, Morton 1988, Brundige 1993, Patterson et al. 1999). Activity patterns are dependent on a variety of factors; Atwood et al. (2004) suggested diel activity was typical of unexploited coyote populations. Others studying coyotes in urban areas, documented a shift in activity to periods when prey is more active (Atkinson and Shackleton 1991; i.e., nocturnal). This was also observed in Jackson Hole, Wyoming when coyote activity was contrasted between suburban and agricultural landscapes and undeveloped areas. Coyotes in the suburban and agricultural areas reduced activity during diurnal periods and increased activity during crepuscular and nocturnal periods (McClennen et al. 2001). Nocturnal activity increased in Saguaro National Park, Tucson, Arizona likely due to easy access to anthropogenic foods (Bounds and Shaw 1997). The general shifts in activity in urban areas compared to rural landscapes have been suggested as a response to human activity. Humans are generally more active during the early hours of evening than at other times of the night and remain inactive longer than coyotes in more rural areas to avoid contact with humans (Grinder and Krausman 2001, McClennen et al. 2001, Atwood et al. 2004, Grubbs and Krausman in press). For example, in Tucson, Arizona, when human activity was high and coyote activity low in mid-day, coyotes were in areas of low human activity with cover. As evening approached and human activity decreased, coyotes moved to golf courses, washes, and residential areas. As night progressed, coyote activity increased and they moved through high density residential and commercial areas. As dawn approached, human activity increased, coyote activity slowed and they returned to familiar day resting spots (Grubbs and Krausman 2009*a*). Others however, have suggested that coyotes in Banff, Alberta, Canada used habitats available to them regardless of human activity (Gibeau 1998). Gibeau (1998) concluded that coyotes were not attracted to urban environments.

Survival.--Very few studies have examined survival of coyotes in exurban-suburban landscapes. In Tucson, Arizona, the annual survival rate of coyotes was 0.72. They were exposed to viral, bacterial, and parasitic infections common to many coyote populations, but humans were the major source of mortality (95%): vehicles (n = 16), trapping (n = 2), unknown (n = 1); Grinder and Krausman 2001). Human caused mortality was also common in a later study of coyotes in Tucson, Arizona. Of 8 monitored animals, 3 were killed by vehicles, 1 drowned, and 2 died of unknown causes (Grubbs and Krausman 2009*a*). In Southern California, the survival rate for coyotes was 0.742 and did not vary with urban association (Riley et al.

2003). Vehicles, other carnivores, and anticoagulant rodenticides were the main causes of death for coyotes. Toxin-caused mortality was related to urban association. Rat poison was also responsible for the death of coyotes in the north edge of Boston, Massachusetts (Way et al. 2006).

Human-coyote conflicts

As urbanization encroaches into coyote habitat and vice versa, an undesirable human interface with coyotes has resulted (Howell 1982). Numerous attacks on adults, children, and pets have been reported from homes, yards, parks, and golf courses (Baker and Timm 1998). The most severe was in August 1981 when a 3 year-old girl in Glendale, California was killed in her front yard by a coyote. These attacks and tragic deaths, plus hundreds of observations of coyotes in urban areas, caused valuation forces to demand action. The City of Glendale, California provides a successful plan of action to minimize coyote-human interaction (Howell 1982).

Case study; Glendale, California.--The incidence of coyotes in urban areas is enhanced by humans. Humans modify habitats to provide prey for carnivores and subsidize diets with anthropogenic foods (Baker and Timm 1998) and provide cover and den sites (Grubbs and Krausman 2009*a*). When coyotes and humans clashed in Glendale, California, the County Board of Supervisors authorized the Agricultural Commissioner to contract with the City of Glendale to selectively trap and remove coyotes where conflicts occurred. In addition to direct removal, the commissioner initiated a public education program to minimize human-coyote conflicts. Furthermore, an ordinance was passed that prohibited feeding predatory animals and rodents in unincorporated areas of Los Angeles County.

The program had early success. In 80 trap days within a 0.8 km radius of the residence where the child was killed, 55 coyotes were trapped or shot (Howell 1982). The City of Glendale still has one of the best deterrent programs in the country. It is currently administered by the police department based on co-existence as the only long-term solution for coyote-human conflicts. When problems are identified, citizens are educated about fencing, habitat modification, human and wildlife behavior, coyote biology, city wildlife anti-feeding ordinances, and the use of pepper spray. Trapping and killing coyotes are used when all recommended solutions fail. Success is attributed to reinstating the fear of humans and urban areas into coyotes (Baker and Timm 1988). All trapped coyotes (No. 3 Victor Soft Catch) are euthanized; relocation is not biologically sound, humane, or sometimes legal (Baker and Timm 1998). When translocation is attempted, it is usually to pacify opposing views of control: those who demand that something be done and those who are against control. Regardless, the issue with translocation is that the problem is also being translocated. Furthermore, translocated animals may create intraspecific conflict and disease transmission. The program in Glendale, California has been successful; attacks and pet losses have been reduced.

The City of Glendale also informs citizens about coyote behavior characteristics and of human safety risks they should be aware of (Baker and Timm 1998).

1. An increase of lost pets at night.
2. An increase in observations of coyotes on streets and yards.
3. Crepusular observations of coyotes on streets, parks, and yards.
4. Observations of coyotes chasing or killing pets.
5. Attacking leashed pets, chasing joggers, bikers, and walkers.
6. Observing coyotes in and around play areas and parks in mid-day.

Management

The emerging picture of coyotes in urban areas suggests they fulfill their daily requirements by shifting activity periods to times when humans are least active and by using areas where coyotes can avoid humans. Unfortunately, many urban dwellers enjoy seeing coyotes as part of their daily life (Kellert 1985) and often feed them directly or indirectly. Feeding causes attacks. Even the small girl killed by coyotes in California had been encouraged by her father to feed coyotes (Grinder and Krausman 1998). From a survey of 188 U.S. National Parks, Bounds and Shaw (1994) reported that where aggressive coyotes were reported, feeding by humans was significantly more commonplace than in parks without aggressive coyotes. Altering human behavior (Peine 2001) and public education is necessary to prevent wildlife-human conflicts (Timm et al. 2004).

Coyotes likely benefit from habitat alteration of urban areas (Grinder and Krausman 2001). Thus, in fragmented landscapes, cover likely facilitates occupancy of areas with relatively high human activity (Atwood et al. 2004). For example, forested habitat provides cover, forage, and travel corridors (Atwood et al. 2004). Society needs to understand how habitat modification to suburban development can progress without increasing mortality or dispersal rates (if that is a desired objective; McClennen et al. 2001). When the decision is made to co-exist with coyotes, society will need more information about coyotes in urban areas so conservation can be promoted by preserving open space, reducing use of rodenticides, providing usable crossing points under freeways and roads, and driving more slowly (Riley et al. 2003).

However, when problems arise, lethal control, education, hazing, and other methods have reduced conflicts (Timm et al. 2004). The Arizona Game and Fish Department recommended 12 ways to discourage coyotes in urban areas.

1. Do not feed coyotes.
2. Eliminate sources of water for coyotes.
3. Remove bird seed.
4. Edible garbage should not be available.
5. Do not keep pet food where coyotes have access.
6. Trim and clean shrubbery to the ground that provides cover for coyotes.
7. Fence the yard.
8. Install a battery-operated electric fence.
9. Do not leave small children outside unattended.
10. Keep pets restrained.
11. Assertively discourage coyotes by making loud noises and throwing rocks when they appear.
12. Ask neighbors to follow the same steps.

A further step could be with broader scale educational programs. In Vancouver, British Columbia, Canada, the Stanley Park Ecology Program (http://www.stanleyparkecology.ca/) has been a useful tool to reduce human-coyote conflicts.

Coyote in front yard of a Tucson, Arizona resident. Photo courtesy of S. Grubbs.

American Black Bears

The problem

As society encroaches further into habitats of American black bears, there are numerous opportunities for human-bear conflicts because bears are common in the U.S. and are increasing in many areas, can tolerate humans, and are attracted to human food (Spencer et al. 2007). As a result, American black bears have presented a serious challenge to biologists, especially where anthropogenic activities occur within bear habitat. An overview about bear populations, levels of complaints about bears, the type of human-bear interactions, management strategies, and documentation was provided by Spencer et al. (2007) via a mail survey completed by American black bear biologists. The survey was sent to wildlife agencies throughout North America. Although few details were provided about the survey design, Spencer et al. (2007) received responses from all 39 states surveyed (states without bears were not surveyed and 8 of 12 Canadian provinces and Mexico responded to the survey). Most responding agencies (44 of 48; 91.6%) provided estimates of black bear population size (n = 747,000; Montana reported 16,500). All agencies reported bear complaints; 43,237 annual complaints with 250/year in Montana and increasing). Garbage and food attractants were the most common type of human-bear conflict followed by general sightings. Agriculture damage (i.e., apiary, orchard, crops) and human encounters were similar in rank. Attacks by bears on humans and livestock were the least common type of conflict reported. In Montana, human-black bear conflicts decreased from garbage and food attraction to human attacks: garbage and food attraction, human-black bear encounters, agriculture damage, general sightings, campsite encounters, livestock attacks, and human attacks. In Montana, 67% of human-black bear conflicts occurred in spring and summer.

Most agencies (77%) did not have a damage fund to pay for the losses due to black bears (Spencer et al. 2007).

Most agencies (89%) had defined protocols for field personnel when responding to human-bear conflicts: site visit, capture and relocation, and euthanasia. Kill permits and use of hunters were ranked the lowest means in protocols. When trapped, culvert traps were the most common traps used followed by leg-hold snares and use of dogs.

Most agencies (75%) relocate problem bears; public pressure drives this decision many times (44%). Few agencies (15%) reported that relocation was the best management approach. On site release was used by 42% of respondents; of these 65% of respondents maintained a database of released bears.

Most agencies marked some captured bears but only 50% marked captured bears all the time to monitor results of aversive conditioning and relocation. Adversive conditioning (i.e., rubber bullets and loud noises) was a common tool used on bears involved in conflict (64%). Bear-resistant containers were used by 50% of respondents and 8 of 24 respondents used agency funds to purchase them.

States, policies, or laws allowing fines for creating depredation situations were reported by 47% of respondents. Garbage management and fines were preferred deterrents to black bear problems over aversive conditioning and relocation. Education programs (i.e., brochures, press releases, radio, TV, workshops) were reported for 81% of agencies.

The survey was instrumental in providing an overview of American black bear issues in North America. Based on these data, Spencer et al. (2007) recommended that agency responses to human-black bear conflicts can be strengthened $\geq$3 ways.

1. Develop protocols for marking and monitoring all bears captured related to conflicts and maintain a database.
2. Develop proactive garbage management.
3. Develop effective education programs.

Deterrents to reduce human-black bear conflicts

Removal of unnatural food.--Garbage and food attractants were the most common type of human-black bear conflicts reported in North America (Spencer et al. 2007). Most wildlife biologists agree that by eliminating unnatural foods, human-black bear conflicts will be minimized (McCullough 1982, Breck et al. 2006). Bear-proof garbage cans, sanitation, and only putting garbage out on collection day are obvious solutions to minimize this problem but the problems persist. As a result, other deterrents have been developed to keep bears from concentrated sources of food.

Translocation.--Capturing and translocating problem bears is a common wildlife management practice in North America (Spencer et al. 2007). However, the process is generally expensive, time consuming, not always successful, and does not address the situation that caused the nuisance behavior (McArthur 1981). In a casual review of 179 American black bears (≥2 years old) translocated in 11 states and provinces, Rogers (1986) reported that 20% of bears returned to the capture site (within 8 – 20 km) when translocated >220 km. When translocated <64 km 81% returned, when translocated 64-120 km 48% returned, and when translocated >120-220 km 33% returned to the capture site. Of documented sexes, 54% (n = 36) of males and 70% (n = 32) of females returned. Translocation did not greatly increase natural mortality among translocations summarized by Rogers (1986).

An advantage of translocating nuisance bears was the removal of the offending individual. These same individuals resumed their negative activities ≤ 65% of the time but usually the following year. This allowed some animals to breed and preserved many for fall hunting. Bears shot during the season were used for food or trophies but few bears killed as nuisances were used for either.

In 1967-1977, 112 American black bears were captured and translocated in Glacier National Park (McArthur 1981). Transplants were considered unsuccessful if the bear returned to the capture area. Other results were recurring problems by the translocated bear elsewhere or being killed outside the park. Thirty-nine bears were transplanted successfully on the first attempt and 19 on subsequent transplants. Females were more likely than males to return to the capture area and males were more likely to create problems elsewhere (McArthur 1981). Factors that contributed to successful transplants included timing, distance moved, number of ridges between the capture and transplant site, elevation, and barriers to movement. Transplants were moderately effective in addressing nuisance black bears in Glacier National Park (58 of 112 bears were successfully translocated). However, how transplanted bears influenced population dynamics and community ecology at the capture or release site are largely unknown.

Education.--Educating the public about nuisance American black bears is another common wildlife management practice in North America (Spencer et al. 2007). Any means possible to heighten the public awareness of the human-bear conflict issue also provided factual information for public debate was instrumental (McCarthy and Seavoy 1994, Tennent and Garshelis 1999) in enhancing human understanding of bears.

Shocking.--Shocking devices have been successful in preventing American black bears from obtaining unnatural concentrated food sources. One device, the Nuisance Bear Controller

(NBC), operates on 2 6-volt batteries wired to a disk that emits 10,000-13,000 volts. Activation occurs when a bear (or another animal) contacts the disk (Breck et al. 2006). The NBC was tested in Minnesota at 10 independent sites in 2004. The NBC was placed on protected and unprotected birdfeeders; no protected feeders were disturbed or destroyed but 40% of the unprotected feeders were robbed or destroyed. The NBC can be useful to deter bears from concentrated forage (Breck et al. 2006).

Taste-aversion.--Taste aversion techniques have also been helpful in minimizing nuisance black bear behavior. Thiobendazole (72-165 mg/kg bear) was administered to 5 nuisance bears in central Minnesota so they would avoid pre-packaged military food. In the next 122 days, they ignored 6 (15%), approached but did not taste 12 (29%), tasted but did not consume 14 (34%), partially consumed 9 (22%), and did not totally consume any of the 41 offered pre-packaged military foods (Tennent and Garshelis 1999). The authors concluded that taste-aversion could minimize nuisance bear behavior toward target foods if alternate unnatural foods were minimized. In Saskatchewan, thiobendazole used around bee yards reduced black bear depredations (Polson 1983). However, in Alaska, bears simply learned how to avoid thiobendazole placed in trash cans and still managed to obtain garbage (McCarthy and Seavoy 1994).

Dogs, pepper spray, crackers, and bullets.--Deterring bears from urban areas has been attempted with a variety of other measures including dogs, pepper spray, crackers, and rubber slugs and buckshot (McCarthy and Seavay 1994, Beckmann et al. 2004). None of the techniques were effective in the long-term (Beckmann et al. 2004), and without addressing the bear attractants congruently, will have limited success at minimizing future conflicts (Leigh and

Chamberlain 2008). Beckmann et al. (2004) suggested they not be used when attempting to alter bear behavior for > 1 month.

Alternative view.--Because many nuisance black bears are associated with unnatural food, attempts have been directed toward eliminating the food or deterring bears from the food. However, McCullough (1982) maintains that it is more than the food that bears become conditioned to. The stimuli for unnatural foods include human scent, human presence, human structures and equipment among others. So even if bears do not detect food, they are still attracted by related stimuli and the conditioned behavior will remain. Furthermore, in many situations (e.g., parks) there are no opportunities for reinforcement of fear (i.e., hunting) in bears toward humans. To avoid human-bear conflicts (McCullough 1982) suggests a model of bear management be developed that instills fear of humans into bears. “Humans and bears have coevolved as adversaries; to expect peaceful coexistence is both unnatural and unwise (McCullough 1982:32).

Biophysical-behavioral forces.--Much of the research related to American black bears within suburban and exurban areas concentrated on deterrents to nuisance behavior with little attention paid to life history characteristics. An early study examined nuisance bears relying on unnatural food by describing life history characteristics of 126 bears (Rogers et al. 1976). The study was conducted in northern Michigan. Most captured bears (67%) using garbage were males as reported by others (Erickson et al. 1964, Rogers 1970). Males have larger home ranges than females and have more opportunities to encounter unnatural food. Number of cubs from females consuming unnatural foods was higher than for females using natural foods (3.1 versus 1.9). Bears using unnatural food were also heavier than those using natural food.

As a resource, black bears were not used for food or trophies when destroyed as nuisance bears. As with other studies, Rogers et al. (1976) recommends that garbage be minimized by prompt removal, bear-proof garbage cans, and garbage dumps located ≥1.6 km from campgrounds or residential areas (Rogers 1970).

More recently, Beckmann and Berger (2003*a*) contrasted bear ecology at the urban-wildland interface and in wildlands at the interface of the Sierra Nevada Range and Great Basin Desert, western North America. Bears in the urban areas had densities 3 times as high as historic values, sex ratios were 4.25 times more skewed toward males, body mass was 30% larger, home range size was reduced 90 and 70% for males and females, respectively, and bears entered dens later than those in wildlands and remained in them for fewer days (Beckmann and Berger 2003*a*). Bears in the urban-interface also shifted activities to nocturnal periods (Beckmann and Berger 2003*b*). Black bears also shifted activity patterns from crepuscular and diurnal to nocturnal when using unnatural food in Sequoia National Park, California (Ayres et al. 1986). The shift to nocturnal activity in campgrounds (with unnatural food) was related to the reduced human activity at night.

Females in the urban-interface had more potential reproductive years and gave birth to 3 times the number of cubs as wildland females. American black bears in the Lake Tahoe basin experienced a distribution shift rather than a demographic increase in response to unnatural food.

Although generally fecundity and age at first reproduction are higher and earlier in urban areas, elevated mortality rates also exist. Subsequently, bear populations associated with urban areas may function with source-sink dynamics, where the urban areas act as sinks to bears from urban and wildland areas (Beckmann and Lackey 2008).

Regulating human-related mortalities of American black bears is another important aspect of adult survival based on modeling populations on the southeastern Coastal Plain (Freedman et al. 2003). Regulations will require limitations to legal and illegal harvests, habitat connectivity, construction of highway underpasses and speed reduction on highways during peak bear activity. Each of these will gain stature with increasing habitat fragmentation and subsequent anthropogenic mortality.

To counteract these changing dynamics and reduce human-black bear interactions, (Beckmann and Berger 2003*a b*) recommended extensive public education about how anthropogenic activities affect bears; laws, ordinances, and regulations against feeding wildlife unnatural foods; and landowners and businesses should obtain and use bear-proof garbage containers.

"Populations located on large tracks of public land, and in particular those with access to bear sanctuaries, may be buffered from increasing human impacts. Nevertheless, other populations will be less fortunate as conditions fostering positive population growth become increasingly rare. The continued subdivision of prime bear habitat and accompanying reductions in sub-adult and adult survival will increase the likelihood of local extinctions and make recovery of at-risk populations more difficult to achieve" (Freedman et al. 2003: 61). Although this statement relates to black bears in the southeastern Coastal Plain, it will also be applicable to Montana if appropriate management is not applied.

Community case studies

When problem black bears are in urban settings, human safety can be a serious concern. Peine (2001) used Kellert and Clark's (1991) framework to describe the evolution of public policy related to black bears for 4 communities in the U.S.

Juneau, Alaska.--Juneau is an ideal area to examine American black bears on the urban interface. It extends for 54 km with about half the area's concentrated development adjacent to prime American black bear habitat. Most areas of human habitation are within 0.4 km from prime black bear habitat. Development of policies and programs concerning nuisance bears took 4 years and went through several stages.

1. Research established density of 3 – 7 bears/1.6 km^2 associated with unnatural food. Researchers (McCarthy and Seavay 1994) concluded that garbage removal was the best solution to the problem and they attempted to alter bear behavior (i.e., a key biophysical-behavioral force) via physical and chemical aversive conditioning (i.e., 12-gauge shotgun slugs, 12-gaage explosive cracker shells, hand thrown seal control bombs, thiobendazole). These methods failed to alter behavior.
2. In 1987 there were numerous complaints and 14 bears had to be killed. Media coverage led to public demands for non-lethal solutions. Non-Alaskans also influenced policy by voicing their protectionist views and threatened to demonstrate at terminals for cruise ships in Juneau, which constituted a major contributor to the community's tourist-based economy (i.e., valuation forces of environmental stewardship and economic community development). These valuation forces led to the formulation of community policy.
3. In 1987, the Juneau City Assembly drafted an ordinance defining guidelines for storage and collection of garbage to discourage wildlife from getting into the city's garbage.

Once the ordinance was passed, the local police served as the institutional force to facilitate the policy.

4. This was followed by education that emphasized the link between unnatural foods and bears including graphic scenes of bears being shot. The education (via various media) included warnings of fines, videos, radio jingles, coloring books, bumper stickers, buttons, pins, and fliers to get the word out.

Voluntary compliance was not widespread and garbage cans were not bear-proof. In 1991, complaints of nuisance-bears increased and 15 bears were killed and 2 people were injured. This led to increased emphasis on limiting unnatural foods and through public education led to public acceptance of a revised ordinance requiring the use of bear-proof garbage containers causing some convenience and financial-related sacrifices. The evaluation of policy driven by economic stability, public safety, and wildlife stewardship was successfully implemented through education (McCarthy and Seavoy 1994, Peine 2001).

Mammoth Lakes, California.--This community is surrounded by the Sierra National Forest, east of Yosemite National Park and is another example of the development of a community-based wildlife policy that took years to evolve. The initiative began in 1996 when up to 3 generations of American black bears entered town for unnatural food and also followed people, some that were pushing baby carriages (Peine 2001). As a result, the community passed an ordinance to ban feeding and hunting within the city limits and provided procedures to minimize nuisance wildlife complaints (e.g., fines from $100.00 - $500.00 and 6 months in jail). Garbage had to be indoors until the day of pickup and bear-proof dumpsters were provided for commercial businesses and a common community dump. These regulations were driven by valuation forces (e.g., public safety, protection of property, and protection of wildlife).

The institutional forces used involved the police department that could apply aversive conditioning to chase the bear or kill the animal and a single private wildlife management consultant who developed an aversive conditioning program. The latter was designed to influence the behavioral forces through aversive conditioning (i.e., capture, immobilizing, removal of blood and a tooth) for first time offenders. This "trauma" often discourages the bears from returning. If they do return for unnatural food, the animals are attacked with liquid-filled and rubber bullets, buckshot, pepper spray, audio devices, pyrotechnic devices, flash-bang devices, and Karelian dogs trained to chase bears. In Mammoth Lakes, policy was influenced by this consultant working with local police and the California Fish and Game Department. This is one of the few communities in the area that has addressed the issue of nuisance bears around the park (Peine 2001). No information was provided on the duration of the project or the overall success.

West Yellowstone, Montana.--Located at the western entrance to Yellowstone National Park, this community is surrounded by American black bear and grizzly habitat. Valuation forces driving public policy for this community were public safety and protection of property. As a result, a comprehensive garbage disposal ordinance was passed that required "storing garbage, refuse and other food of any type what-so-ever edible by bears" in a secure building or bear-proof garbage can. Feeding, approaching, or harassing bears is also prohibited.

The Chief of Police constitutes the institutional forces and approves garbage cans used, schedules garbage collection, and penalizes offenders ($500.00 and/or 3 days in jail). This ordinance has been enforced for over a decade, is the most comprehensive community ordinance designed to minimize human-bear conflicts, and can serve as a successful model (Peine 2001).

Gatlinburg, Tennessee.--Gatlinburg is the primary gateway to Great Smokey Mountains National Park, which receives more visitors than any other park in the U.S. As a result, there are numerous tourist activities and venues available and anthropogenic development is rapid. The park provides habitat for bears but the animals have accessed unnatural food in the community for >25 years. Biophysical-behavioral factors were responsible for policy to reduce this activity. In the early 1990s, the bear population rapidly increased due to an abundant source of natural food and there was an active plan to reduce poaching. In 1997, the natural foods were reduced due to an unusual frost and drought causing a higher dependence on unnatural foods in Gatlinburg. The result was higher human-bear conflicts (Peine 2001).

Due to social-structural forces, there was resistance toward developing wildlife policy to address the problem. Gatlinburg was the prime area to hunt bears in the county and harvesting bears using unnatural food was a tradition among locals. Tennessee allows baiting to entice bears up to 10 days prior to and during the hunting season. Members of the City Council were protective of hunters' rights and if they were not hunters, they knew hunters, or had relatives that hunted (Peine 2001). In 1997, a record 81 bears were legally shot in Seiver County, most within the city limits of Gatlinburg.

In 1997 and 1998, this record harvest during the peak of the autumn season and the increased number of bears wandering in search of unnatural foods was instrumental in bringing media attention to Gatlinburg. In addition, a tourist witnessed a bear being shot at a dumpster in town and the largest bear on record from the park was found dead; it had been shot searching for garbage. These valuation forces clashed and the national media asked why the City of Gatlinburg did not have a policy to deal with these conflicts. Clearly the city benefited from the presence of bears as they are a tourist attraction and the city also began to have concerns about liability due to potential personal injury. This concern intensified when a person was killed by bears in the park near Gatlinburg. Institutional forces began. Some homeowners obtained bear-proof garbage containers. Once the unnatural food was secure in these areas, bears moved to restaurants and hotels and foraged in their trash leading to a task force in 1989 established by the city to investigate the problem. They reported 5 key findings.

1. Garbage control was not adequate.
2. Some citizens wanted the bears, others did not.
3. Sevier County would not let bears be translocated outside the county.
4. U.S. National Park policy did not allow bears to be translocated into the park.
5. Trained personnel to deal with the problem were limited.

A second group was established in 1997 and met until 1999 to further study the problem. They recommended garbage control and prohibitions on feeding bears with $500.00 fines for violations. Still some business owners failed to comply because pan-handling bears attract customers. The hunting issue was not resolved because that was controlled by the Tennessee Wildlife Resources Agency (Peine 2001). This conflict is another that demonstrates the complexity of establishing wildlife policy in a timely fashion.

In each of the examples cited (Peine 2001), human-bear conflicts involved unnatural food-conditioned bears, communities were reluctant to develop policy, policy initiation was triggered by specific events (e.g., economy, human safety, property protection, protection of bears), and after various attempts, bear-proof garbage containers were accepted (with the exception of Mammoth Lakes). Policy formulation occurred over 10-25 years. The time lags were due to human behavior, which is reluctant to behavioral changes unless they recognize the consequences of their collective actions on resources in specific situations that are valued (Peine 2001).

Developing ordinances are important to minimize human-American black bear conflicts. As in the case studies presented by Peine (2001), the California Department of Game and Fish recognized that most human-bear problems were generated from improper garbage storage in the San Gabriel Mountains. Policy was established to educate and recommended that unnatural food had to be removed, garbage should be kept inside until the pick-up day, pet food should be kept inside, barbecue grills should be cleaned, and ripened and dropped fruit picked up. However, none of this was enforced and even if some citizens complied, the problems persisted (Lyons 2005). The cost of converting to bear-proof cans was the main reason the policy was not enforced. Human behavior has to be adjusted if human-bear conflicts are to be minimized (Lyons 2005). Adjusting trash cans and collection methods will likely be significantly cheaper than losing multi-million dollar, out-of-court settlements due to bear attacks (Peine 2001).

Ungulates

The problem

Deer in urban areas are widespread throughout the United States, and deer-human conflicts are occurring at much higher rates with deer expansion and increasing human populations (Conover 1995, DeNicola 2000). These conflicts include human health concerns associated with Lyme disease, deer-vehicle collisions (DVC), and economic damage to landscaping and crops from herbivory (Conover 1995).

There are several factors that enhance deer populations in urban areas. In rural landscapes, deer depend on transitional zones between forests, grasslands, and agricultural areas where combinations of food and cover are available. Similar edge habitats occur in suburban areas, which provide high-quality food in the form of ornamental plants, gardens, or fertilized lawns (Swihart et al. 1995) while adjacent woodlands or natural patches provide daytime cover (DeNicola et al. 2000). Additionally, many suburban and exurban areas have firearm restrictions or protective laws that limit hunting throughout most of the year and lack natural predators, leading to behavioral adjustments (Conover 1995, DeNicola et al. 2000). Finally, residential homes provide shelter from severe weather (Grund et al. 2002) and suburban residents often feed deer (Swihart et al. 1995, DeNicola et al. 2000).

White-tailed deer (*Odocoileus virginianus*) are adaptable species in diet breadth and habitat use and have broadened their ranges to include expanding urban and suburban environments (DeNicola et al. 2000). In areas where deer herds in urban areas are not actively managed, these populations are becoming locally overabundant, (i.e., deer numbers approach or exceed human tolerance levels; DeNicola et al. 2000). The restrictions for deer management in urban areas include real and perceived safety concerns, conflicting perceptions and social attitudes towards wildlife, restrictions on hunting and firearm-discharge, and public relations and liability concerns (DeNicola et al. 2000). Clearly, deer in urban areas are becoming an increasingly important issue for state wildlife agencies and continuing management and monitoring is needed to sustain healthy deer populations and mitigate human-wildlife conflicts.

A thorough understanding of the ecological factors concerning deer populations in urban areas should be incorporated into management decisions to determine the scope of the solution and likelihood of success. The current boom in exurban development also raises issues in deer management as many of the conflicts and resulting mitigation for deer in urban, suburban, or rural areas differ in exurban areas. Also, most research concerning deer ecology in urban areas focuses on white-tailed deer in the eastern and mid-western U.S. Similarly, most research regarding control and management of deer in urban areas is also geared toward white-tailed deer. Because of the varying biophysical-behavioral forces between white-tailed deer, mule deer (*O. hemionus*), and elk (*Cervus elaphus*), each will receive separate treatment concerning the biophysical-behavioral characteristics of the species and the cumulative effects of urbanization on them.

White-tailed deer

Densities.--Deer densities vary across studies and locations but tend to be higher in urban landscapes, ranging from 19.8 - 105.6 deer/km^2 (Table 2). Throughout a 5 year culling and contraception program in Rochester, New York, that reduced the population from 99% carrying capacity to 48%, deer densities decreased from 19.8 deer/km^2 to 9.6 deer/km^2 (Porter et al. 2004). Beringer et al. (2002) measured deer densities in a rural area to be 4.0/km^2. White-tailed deer densities ranged from approximately 2.5-5.9 deer/km^2 in rural Illinois (Nixon et al. 1991). However, on 2 study areas of the Lower Yellowstone River in Montana, deer densities were higher, averaging 34.3 deer/km^2 and 39.3 km^2 (Dusek et al. 1989).

Home ranges.--Home ranges of white-tailed deer in urban areas vary across studies, states, and housing densities but are generally smaller than home ranges in rural areas (Table 3). Kilpatrick and Spohr (2000) recorded a 43.2 ha annual home-range size for female white-tailed deer in suburban Connecticut. Piccolo et al. (2001) found female home range sizes averaging 25.8 ha and 60.8 ha in 2 forest reserves in urban Illinois. Beringer et al. (2002) examined post-translocation home-range sizes for deer in urban areas in Missouri and found home ranges for males and females increased 170 ha post-translocation.

Table 2. White-tailed deer densities across rural-suburban gradients, 1989 - 2004.

Density (deer/km^2)	Location	Landscape	Source
34.3-39.3	Lower Yellowstone River, MT	Rural	Dusek et al. (1989)
2.5-5.9	Illinois	Rural	Nixon et al. (1991)
72.7	Bridgeport, CT	Suburban	Swihart et al. (1995)
28	Groton, CT	Suburban	Kilpatrick and Walter (1999)
31	Town and Country, MO	Suburban	Beringer et al. (2002)
4	Huzzah Conservation Area, MO	Rural (translocation area)	Etter et al. (2002)
0-105.6	Chicago, IL	Suburban	Porter et al. (2004)
9.8 9.6	Rochester, NY	Suburban Post-culling/ contraception	

White-tailed deer, Butler Creek, Missoula, Montana. Photo courtesy of C. L. Krausman.

Several studies also report a seasonal variation in home-range size, with home ranges being smaller in the summer at the onset of fawning season and largest during winter and early spring (Swihart et al. 1995, Grund et al. 2002, Etter et al. 2002, Storm et al. 2007). This seasonal behavior has also been reported for white-tailed deer in rural areas (Nixon et al. 1991, Sparrowe and Springer 1970). For example, home ranges in suburban Minnesota increased from 50.4 ha to 85.3 ha summer to winter (Grund et al. 2002). Kilpatrick and Spohr (2000) and Porter et al. (2004), however, reported no difference between summer and winter home-range size in Connecticut or New York. This might be due to the limited availability of undeveloped areas within deer home ranges and habitat fragmentation that restricts home ranges from expanding (Kilpatrick and Spohr 2000).

Home ranges in suburban areas tend to be smaller than in exurban areas (Cornicelli 1992, Cornicelli et al. 1996, Storm et al. 2007). Summer and winter exurban ranges of females in Illinois averaged 53.0 ha and 90.6 ha, respectively (Storm et al. 2007).

Survival.--Survival of white-tailed deer in urban areas is higher than for deer in rural areas (Table 4; Etter et al. 2002) although Storm et al. (2007) reported higher survival rates for white-tailed deer in exurban environments (87%) than in suburban (62 - 82%) and rural (57 - 76%) areas. Porter et al. (2004) recorded survival rates of 89%. Deer-vehicle collisions are the primary source of white-tailed deer mortality in many urban and suburban areas (Etter et al. 2002, Nielson et al. 2003, Porter et al. 2004), whereas recreational hunting tends to be the primary source of mortality in rural areas (Nixon et al. 1991, Brinkman et al. 2004). The primary source of mortality for Florida Key deer (*Odocoileus virginianus clavium*) fawns was drowning in ditches and death due to collision with vehicles (Peterson et al. 2004). In contrast, there was 67% mortality of white-tailed deer fawns for the initial 8 weeks of life in an exurban area in Alabama. Coyotes were responsible for 41% of mortality and starvation due to abandonment was responsible for 25%. Vehicle collisions were not an important cause of fawn mortality in Alabama (Saalfeld and Ditchkoff 2007). Fewer DVC occurred in exurban areas due to fewer roads (1.5 km/km^2; Storm et al. 2007). Additionally, hunting in exurban areas is not permitted on all properties (19%; Storm et al. 2007).

Habitat use.--Habitat use of white-tailed deer in urban areas varies across studies and locations as white-tailed deer are widespread and topography and vegetation vary throughout their range. Studies of white-tailed deer in urban areas indicate that deer readily habituate to human development, and although they sometimes appear to avoid residential areas when possible (i.e., summer fawn-rearing season), will exploit these areas if little other choice is available and there is sufficient cover, especially during winter (Swihart et al. 1995, Kilpatrick and Spohr 2000, Grund et al. 2002). Females in exurban Illinois selected grasslands outside of zones of human influence during summer fawning (Storm et al. 2007). In Connecticut residential areas were important foraging sites for deer during winter (Swihart et al. 1995).

In Connecticut, housing density decreased from 48.3 houses/deer home range to 30.2 houses/deer home range from winter to the fawn-rearing season and distance from core area to residential development increased from 41.4 m to 99.4 m from winter to summer (Kilpatrick and Spohr 2000). Others also found a decrease in the number of homes/ha within deer home range and core areas from winter to summer in an exurban area. Dwelling density in home ranges of deer and core areas averaged 0.13 ± 0.03 dwellings/ha and 0.14 ± 0.05 dwellings/ha, respectively, during fawning and 0.18 ± 0.02 dwellings/ha and 0.16 ± 0.03 dwellings/ha in winter home and core ranges, respectively (Storm et al. 2007).

Table 3. Annual and seasonal home ranges for white-tailed deer across the rural - urban gradients, 1970 - 2007.

Home range (SE) ha			Method	Sex	State	Landscape	Source
Annual	Winter	Summer					
	699.3 (440)	259 (129.5)	100% MCP	M/F	SD	Rural	Sparrowe and Springer (1970)
	132 (18.3)	221 (19.0)	Hand-drawn	F	NY	Rural	Tierson et al. (1985)
	150 (18.3)	233 (23.4)		M			
1,630			100% MCP	F	MT	Rural	Vogel (1989)
340						Exurban	
	177 (14)	55 (7)	MMA[a]	F	IL	Rural	Nixon et al. (1991)
	440 (19)	323 (49)		M			
45			100% MCP	F		Enclosure[b]	Beir and McCullough (1990)
142				M			
158			100% MCP	F	IL	Exurban[c]	Swihart et al. (1995)
67						Suburban[d]	
560			100% MCP	M/F	MT	Rural	Dusek (1987)
	42	58		F			
	32	255		M			
400 (50)	130 (3)	230 (3)	100% MCP	F	MT	Rural	Dusek et al. (1988)
	75 (17)	45 (19)			MN	Urban	Grund (1998)
	28.5 (2.8)	32.4 (2.7)	95% MCP	F	SC	Suburban	Henderson et al. (2000)
43.2 (2.7)	35.7 (3.2)	32.9 (3.2)	95% AK	F	CT	Urban-suburban	Kilpatrick and Spohr (2000)
60.8 (13.1)			95% MCP	F	IL	Urban	Piccolo et al. (2001)

Table 3. continued. Annual and seasonal home ranges for white-tailed deer across the rural - urban gradients, 1970 - 2007.

Home range (SE) ha			Method	Sex	State	Landscape	Source
Annual	Winter	Summer					
25.8 (4.9)							
		86[e]	Kernal	F	MO		
51.4-72.6		250-430[e]				Suburban	Beringer et al. (2002)
	40.5-61.5	22.0-30.0	95% MCP	F	IL	Post-trans.	
	85.3 (5.8)	50.4 (6.8)	95% AK	F	MN	Suburban	Etter et al (2002)
	22.4 (2.7)	21.4 (3.4)	90% MCP	F	NY	Suburban	Grund et al. (2002)
			95% FK	F	IL	Suburban	Porter et al. (2004)

[a] = Lower-density housing, 1-2 ha lots.

[b] = Deer surrounded by high density commercial/residential housing.

[c] = Minimum mean area.

[d] = Home ranges for fawns and adults.

[e] = George Reserve.

Table 4. Annual survival of white-tailed deer across rural-suburban gradients, 1990 - 2007.

Annual Survival (proportion killed by vehicles)				Sex	State	Landscape	Source
All	Fawns	Yearlings	Adults				
			0.46	M	MN	Rural	Fuller (1990)
			0.69	F			
			0.73	M/F	IL	Suburban Post-translocation	Jones and Witham (1990)
	0.44		0.34	F	IL	Rural	Nixon et al. (1991)
	0.95	0.62	0.71	M			
	0.88	0.38	0.39	F	CT	Suburban	Swihart et al. (1995)
	0.38	0.86	0.82	M			
	0.77		0.83	F	SD	Rural	Deperno et al. (2000)
			0.57	M/F	MO	Suburban	Beringer et al. (2002)
				M/F	MO	Post-translocation	Etter et al. (2002)
0.69 (0.68)				F	IL	Suburban	
							Brinkman et al. (2004)
0.30 (0.09)				M	MN	Rural	Porter et al. (2004)
0.82 (0.66)	0.85 (1.0)	0.82 (0.80)	0.83 (0.55)	F	NY	Suburban	Storm et al. (2007)
0.83 (1.0)				F	IL	Exurban	
0.76 (0.20)	0.84[a] (0.17)	0.97[b] (0.0)	0.75[c] (0.23)	F			
0.57 (0.44)	0.60	0.89	0.62				
			0.87 (0.14)				

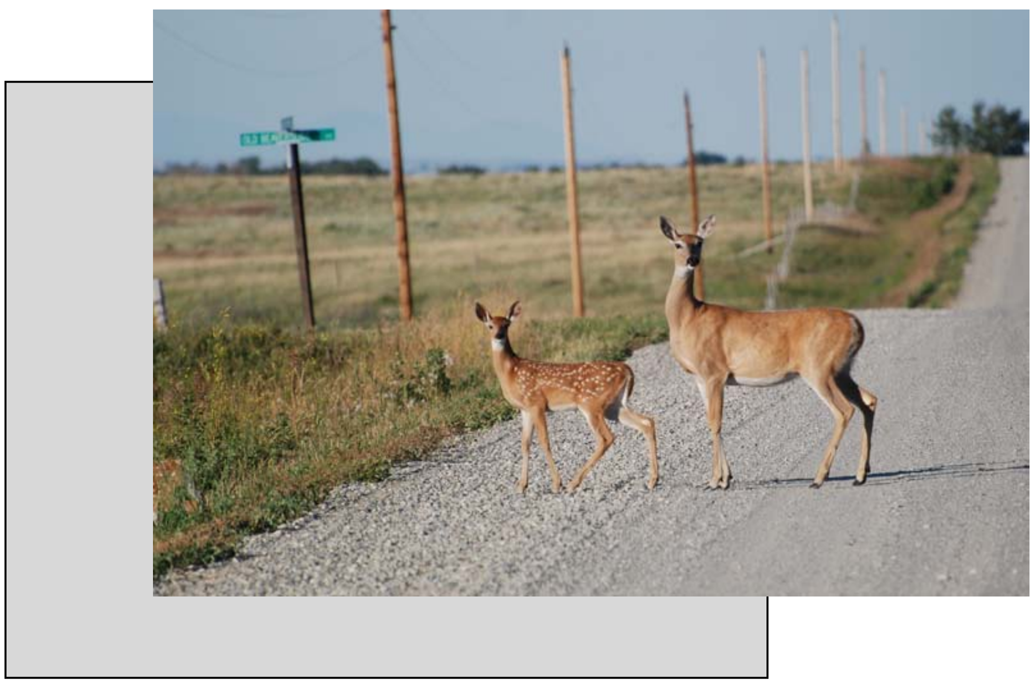
[a] = Neonate fawns.
[b] = 7-12 mo.
[c] = >12 mo.

Movements and dispersal.--Overall, seasonal movements of female white-tailed deer in suburban areas are limited, and with 1 exception, all females in a Rochester, New York study moved <1.1 km (Table 5; Porter et al. 2004). White-tailed deer in Bloomington, Minnesota did not migrate or shift their home range centers during mild winters but moved <1.2 km during an extreme winter. Less than <10% of females dispersed and <15% migrated in suburban Chicago, Illinois (Etter et al. 2002). Meanwhile, 50% of male fawns and <10% of yearling and adult males dispersed, while no males migrated (Etter et al. 2002). Dispersal and migration distances for females were 7.6 km and 4.0 km, respectively, and dispersal distances for males averaged 5.4 km (Etter et al. 2002). Porter et al. (2004) observed 56% of females migrating (although nearly all deer showed >50% overlap between summer and winter home ranges). Annual dispersal rates were 14.3% for yearlings and 8.3% for adults, with an average dispersal distance of 4.0 km (Porter et al. 2004).

Few studies of deer in rural areas report low dispersal rates, except for adult females (≤5%; Hawkins and Klimstra 1970, Tierson et al. 1985, Aycrigg and Porter 1997). Meanwhile, dispersal rates of 13 - 50% for fawn and yearling females and 37 - 80% for fawn and yearling males have been reported for rural areas (Hawkins and Klimstra 1970, Tierson et al. 1985, Dusek et al. 1989, Nixon et al. 1991, Nelson and Mech 1992, McCoy et al. 2005). Migration rates for white-tailed deer in rural areas are also low; deer on the lower Yellowstone River were generally non-migratory; 27 of 34 (79%) of adult and 13 of 19 (68%) yearlings had overlapping summer and winter ranges (Dusek et al. 1989).

Fertility.--Fertility rates in urban landscapes differ by age class but are generally lower in urban than in rural landscapes (Table 6). In suburban Connecticut, fetus:female ratios averaged 0.0, 0.60, and 1.20 for fawns, yearlings, and adult females, respectively (Swihart et al. 1995).

`Fertility rates increase after translocation and removal programs. Beringer et al. (2002) measured pre- and post-translocation fertility rates of deer in Missouri. Fawn:female ratios for fawn and yearling females averaged 0.0 and 0.86 in the suburban area and 0.14 and 1.22 post-translocation (Beringer et al. 2002). Nielson et al. (1997) measured fertility rates before and after an adaptive management program; fetus:female ratios increased from 1.33 to 1.85 post-removal. Porter et al. (2004) examined the effects of culling and contraception on fertility rates over a 5-year period. Fetus:female ratios for fawn, yearling, and adult females increased from 0.0, 1.17, and 1.41 from a high-density (99% carrying capacity) population to 0.27, 1.75, and 2.11 for a population at 48% carrying capacity (Porter et al. 2004).



White-tailed deer doe and fawn, Dupuyer Creek Road, Dupuyer, Montana.
Photo courtesy of G. H. Palmer.

Table 5a. Dispersal rates (%) of male and female fawn, yearling, and adult white-tailed deer across the rural - suburban gradients, 1970 - 2005.

Dispersal rates (SE				Sex	State	Landscape	Source
Fawn	Yearling	Adult	Distance (km)				
	80	7	1.3-7.7	M	IL	Rural	Hawkins and Klimstra
	13	0		F			(1970)
	40.5	13.5		M	NY	Rural	Tierson et al. (1985)
		3.8[a]		F			
	46		18.5	M	WY	Rural	Dusek et al. (1998)
	17		19.5	F			
51			40.9 (5.0)	M	IL	Rural	Nixon et al. (1991)
50			49.9 (4.8)	F			
	20		17.6-168.0	F	MN	Rural	Nelson and Mech (1992)
		2.7	4.0	F	NY	Rural	Aycrigg and Porter (1997)
50	9.1	5.2	2.9-8.8	M	IL	Suburban	Etter et al. (2002)
7.3	4.7	6.5	1.9-33.9	F			
	14.3	8.3	4.0 (1.3)	F	NY	Suburban	Porter et al. (2004)
	36.7		1.9-13.8	M	TX	Rural	McCoy et al. (2005)

[a] = Did not specify age-class.

Table 5b. Migration rates (%) of male and female fawn, yearling, and adult white-tailed deer across the rural - uburban gradients, 1970—2005.

Migration rates (SE)				Sex	State	Landscape	Source
Fawn	Yearling	Adult	Distance (km)				
	20.6	46.2		M/F	WY	Rural	Dusek et al. (1989)
		19.6[a]	13.0 (3.0)	F	IL	Rural	Nixon et al. (1991)
4.9	14.3	4.8	4.0	F	IL	Suburban	Etter et al. (2002)
			0.64 (0.25)[b]	F	MN	Suburban	Grund et al. (2002)
		56.3[c]	<1.1[d]	F	NY	Suburban	Porter et al. (2004)

[a] = Includes yearling females.
[b] = Seasonal shift between geometric centers consecutive seasonal HR.
[c] = Did not specify age class.
[d] = Only 1 female had completely non-overlapping seasonal HRs.

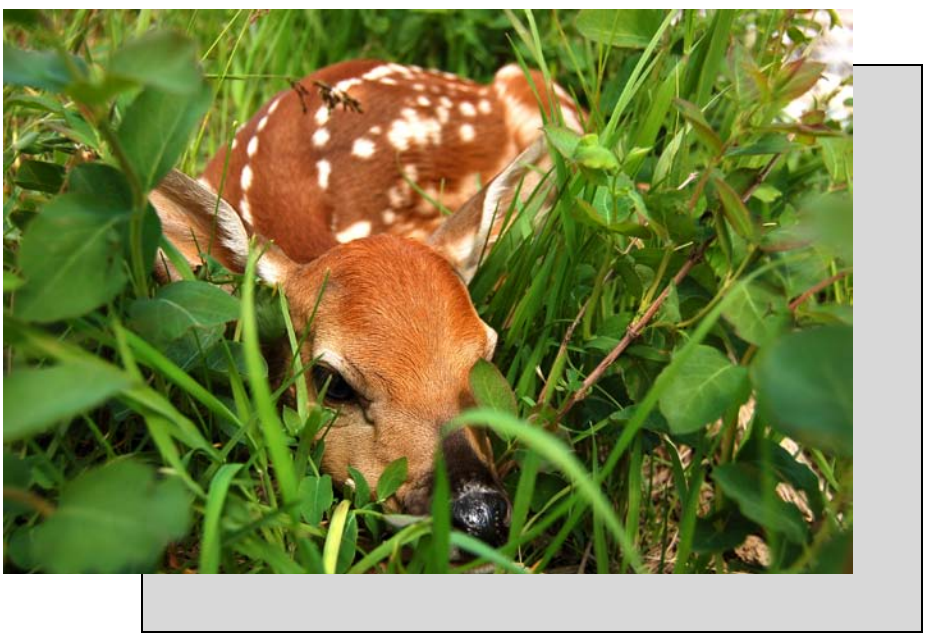
White-tailed deer, Butler Creek, Missoula, Montana. Photo courtesy of C. L. Krausman.

Table 6. White-tailed deer fertility across rural-suburban gradients, 1991 - 2004.

Fertility (fetuses: female)				State	Landscape	Source
All	Fawn	Yearling	Adult			
1.33[a]				IL	Rural	Nixon et al. (1991)
	0.0[a]	0.60[a]	1.20[a]	CT	Suburban	Swihart et al. (1995)
1.33					Suburban	Nielsen et al. (1997)
1.85					Post-AM[b]	
	0.0[a]		0.86[a]	MO	Suburban	Beringer et al. (2002)
0.96[a]	0.14[a]		1.22[a]		Post-translocation	
	0.0	1.17	1.41	NY	Suburban – high density	Porter et al. (2004)
	0.27	1.75	2.11		Suburban – low density	

[a] = Indicates fawns:female instead of fetuses:female.

[b] = Adaptive management program (culling and contraception).



White-tailed deer fawn, Theodore Roosevelt Memorial Ranch, Dupuyer, Montana. Photo courtesy of S. M. Smith

Mule deer

White-tailed deer are among the most well-known and widespread large mammal species in North America and have flourished in wilderness and metropolitan areas during the last century (Conover 1995, DeNicola et al. 2000). However, little evidence suggests that mule deer survive and thrive as well as white-tailed deer in urban environments (Reed 1981*a*, Vogel 1989). Furthermore, predictions of the consequences of fragmentation from exurban development to mule deer were in their infancy (Kucera and McCarthy 1988). In a study of the effects of housing density on white-tailed and mule deer in Gallatin County, Montana, a species composition shift occurred in which white-tailed deer increased in abundance or expanded into areas historically occupied by mule deer (Vogel 1989). This could be the product of several ecological factors between the species. For instance, white-tailed deer have higher natality rates, lower fawn mortality, and an overall younger population structure than mule deer (Vogel 1983). Disturbance is usually tolerated in species that have greater numbers of young, shorter life expectancy, wider dispersal patterns, and more nocturnal habits (Geist 1971), a description better suited toward white-tailed deer than mule deer (Vogel 1989).

Density.--Although studies that investigate the ecology of mule deer in urban areas are limited, several reports indicate that densities of mule and black-tailed deer are lower in urban areas than rural areas, unlike those of white-tailed deer (Smith et al. 1989). McClure et al. (2005) calculated densities of approximately 6.3 deer/km^2 in urban sites in Utah and 7.1 deer/km^2 in adjacent rural areas, coinciding with Vogel's (1983, 1989) observation of fewer deer observed in more developed areas in Gallatin County, Montana. Similarly, Columbian black-tailed deer densities averaged <0.3/ km^2 in urban Vancouver, Washington versus 2.7 deer/km^2 in surrounding rural areas (Bender et al. 2004*b*). In Montana, mule deer densities can be fairly high; low-density rural winter ranges in Montana contain 8.3 - 10 deer/km^2 whereas more restricted, mountainous winter range areas in Montana experience mule deer densities ranging from 31 - 50 deer/km^2 in mild winters to 83 - 250 deer/km^2 in concentrated areas during more severe winters (Mackie and Pac 1980).

Survival.--Bender et al. (2004*b*) recorded survival rates of 0.70 (n = 11) for female and 0.86 (n = 6) for male Columbian black-tailed deer in urban Vancouver, Washington. Fawn survival rates averaged 0.84 (n = 26; Bender et al. 2004*b*). The survival rates of females were low compared to Columbian black-tailed deer in rural habitats in the Pacific Northwest (i.e., 0.73 and 0.80 for females; McNay and Voller 1995, McCorquodale 1999). Male Columbian black-tailed deer survival was higher compared to rural studies, McCorquodale (1999) estimated survival rates of 0.50 while Bender et al. (2004*c*) reported rural male survival 0.50 - 0.52. Mortality of urban black-tailed deer was primarily attributed to DVCs or deer-train collisions (83.3%, Bender et al. 2004*b*). Two of the 3 deer killed by DVCs were observed being chased by domestic dogs just prior to the collision (Bender et al. 2004*b*). Primary causes of mortality for rural black-tailed deer are predation and hunter harvest (McNay and Voller 1995; McCorquodale 1999). In rural areas in Washington, hunting contributed to 56% of the male black-tailed deer mortalities (Bender et al. 2004*c*).

Habitat.--Some development affecting mule deer habitat and populations has occurred in areas used as summer and fall range, and while more is likely to occur, the greatest threat comes from development on and adjacent to major winter ranges (Mackie and Pac 1980). Because mule deer distribute themselves and exhibit fidelity to specific sites, loss of these regions can have profound implications on mule deer occurrence in different areas and other seasons (Mackie and Pac 1980, McClure et al. 2005). For example, in the Bridger Range north of

Bozeman, Montana, deer winter in densities of 25.5 - 34.0 deer/km^2 and summer at densities of 3.9 - 4.6 deer/km^2 (Mackie and Pac 1980). Thus, the loss of 1km^2 of winter range may result in a minimum loss of 26 - 34 deer in the population; the equivalent of all adult animals within 25 – 31 km^2 of summer range (Mackie and Pack 1980). Additionally, some areas of winter range are more important than others - during more severe winters, ~77 deer can concentrate on less than 1 km^2 of range; loss of this primary area could result in a significant loss in the deer population (Mackie and Pac 1980).

The influences of urban developments on mule deer habitat are also complex because many mule deer are migratory and the alteration of habitat conditions from urban developments in mule deer habitat affect population sizes and the ratio of migratory to non-migratory animals (McClure et al. 2005). McClure et al. (2005) examined the ratio of migratory versus non-migratory mule deer and assessed differences between fawn recruitment in deer herds using adjacent urban and rural winter ranges in the Cache Valley, Utah. They found that 15 of 17 radiocollared deer in urban areas were migratory, opposed to 8 of 14 deer in rural areas. Additionally, spring migration of deer in urban areas commenced an average 2 - 3 weeks sooner than deer in rural areas in both years examined (McClure et al. 2005). Deer in urban areas travelled an average 31.5 km and deer in rural areas travelled an average 14.5 km between winter and a shared summer range (McClure et al. 2005).

Winter habitat use between mule deer in urban and rural areas also differs (McClure et al. 2005). For instance, deer in urban areas selected habitats with concealment vegetation, whereas deer in rural areas exhibited more neutral behavior toward this characteristic (McClure et al. 2005). McClure et al. (2005) postulated that deer in urban areas were less likely to take risks associated with large-scale movements for better forage, which may explain their lower nutrition and recruitment, and the earlier, more pronounced migration from the herd in the urban area. The home ranges of deer in urban areas were also smaller than deer in rural areas. Activities of deer in urban areas were more aggregated than their rural counterparts (McClure et al. 2005).

In a comparison of fawn:female ratios between mule deer in rural and urban areas, fawn ratios are conspicuously lower in urban areas (McClure et al. 2005). In 1995 and 1996, the fawn:female ratios for migratory urban mule deer were 0.88 and 0.71, respectively, for non-migratory urban deer the ratio was zero in both years, indicating that non-migratory behavior is perishing within the urban deer herds (McClure et al. 2005). Fawn recruitment for mule deer in rural areas was the same for migratory and non-migratory animals at 1.0 fawn:female in 1995 and 1.25 fawn:female in 1996 (McClure et al. 2005).

Columbian black-tailed deer in Washington produced approximately 1.83 fawns:female and 1.36 fawns:female over a 2-year study period (Bender et al. 2004*b*). Due to fawn or female mortalities, recruitment rates of black-tailed deer in Washington ranged from 0.90 to 1.33 fawns:female (Bender et al. 2004*b*). For black-tailed deer in urban Vancouver, Washington, high rates of fawn recruitment compensate for lower survival rates compared to rural populations (Bender et al. 2004*b*). Productivity in this study was higher than any previously recorded level in the Pacific Northwest (Bender et al. 2004*b*).

Case study

The increasing population of deer in Helena, Montana is a good example of the interaction of biophysical-behavioral, social-structural, valuation, and institution-regulation forces. The ecology of deer is well defined and a basic principle of any management is knowledge of population levels (Krausman 2000). To supplement baseline data with population estimates for the city, Hickman (2007) conducted a street-by-street survey. He also subjectively documented deer impacts and deer defenses developed by homeowners. Observers (2 – 4) in a vehicle drove throughout the city at <40 km/hour counting deer within the city boundaries. Some areas were not included in the survey because of traffic congestion, safety considerations, time considerations, or inaccessibility (e.g., Helena airport). The vehicle stopped if numbers, sex, and age needed to be verified. Deer were recorded by street or block with GPS coordinates as male, female, or fawn. Surveys were conducted 2-3 times beginning 3 hours before sunset or 3 hours after sunrise in 7 Helena Citizens Council Districts (no. surveys = 20) in December 2006 and January 2007. The maximum number counted for the survey was 289 mule deer along the 280 km of survey routes. There were 36 male:100 females in the observational survey.

Because the deer were within the city and on private and state owned land, the social-structural forces were simplified, however, valuation forces were unclear until a series of public comments and surveys were conducted. Public meetings and comments recorded revealed an array of attitudes ranging from complete control of the deer population to no control (Helena Urban Wildlife Task Force 2007). A telephone survey revealed that citizens did not enjoy seeing deer in their neighborhoods (36.4%), deer were a problem (58.6%), and actions were necessary (78.3%). Most supported lethal (54.3%) and non-lethal (82.1%) solutions to control deer and nearly 90% said the Montana Fish, Wildlife & Parks should be responsible for deer management along with the City of Helena (72.1%). Only 2.1% of those surveyed were involved in an accident with deer within city limits. Controlling deer would enhance the health of the deer population (48.2%), reduce predator activity in the city (62%), and reduce risks to their pets from urban deer (45.5%) (Helena Urban Wildlife Task Force 2007). Deer were clearly creating concerns in the city (Table 7).

Based on public meetings and the survey, biologists, administrators, and managers had a good representation of public valuation forces. In 2003, the Montana Legislature passed HB 249 and SB 410 as amended to read: "7-31-4110. Restriction of wildlife. A city or town may adopt a plan to control, remove, and restrict game animals, as defined in 87-2-101, within the boundaries of the city or town limits for public health and safety purposes. Upon adoption of a plan, the city or town shall notify the department of fish, wildlife, and parks of the plan. If the department of fish, wildlife, and parks approves the plan or approves the plan with conditions, the city or town may implement the plan as approved or as approved with conditions… The plan may allow the hunting of game animals and provide restrictions on the feeding of game animals."

Once the institutional-regulation forces were in place, the City of Helena began to implement their plan in autumn 2008.

Elk

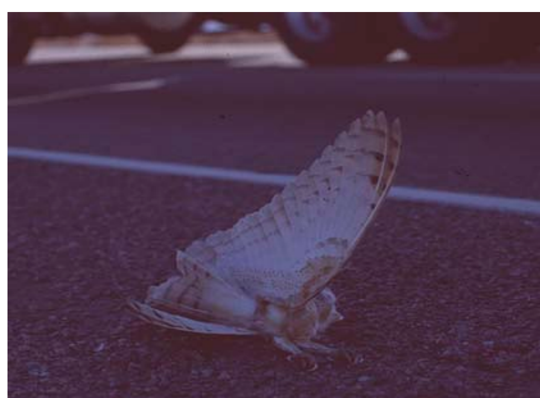
Exurbanization and its influence on white-tailed deer and mule deer has received more attention than the influences on elk but elk are clearly affected. Lessons learned from other ungulates can be applied to elk as urbanization spreads into the habitat of elk. Many of the conflicts occur because elk generally spend summer and autumn on high-elevation public land but migrate to lower elevations in winter and spring; many of these habitats are in private ownership which does not have the same level of protection as public lands (Hayden 1975, Henderson and O'Herren 1992, Haggenty and Travis 2006). All across the west, changes in use of private lands are changing the landscape and wildlife habitat (Henderson and O'Herren 1992, Wait and McNally 2004). These changes include alterations of large agricultural holdings to

Table 7. Reports to Montana Fish, Wildlife & Parks personnel about wildlife within the city limits of Helena, Montana, 2004 - 2007.

Species	Interaction report	No. interactions / year			
		2004	2005	2006	2007
Mule deer	Injured/sick	22	49	54	60
	Dead	21	26	40	41
	Aggressive	2	52	28	27
	Hanging in fence	3		6	4
	Nuisance	9	22	24	14
	Killed dog				1
	Fawns separated	1	1	7	5
	Poaching				1
	No deer in area				1
White-tailed deer	Injured		1		2
	Dead		3	1	2
Mountain sheep	Dead		1		
Moose	Sighting				1
	Nuisance				
Pronghorn	Sighting				1
American black bear	Getting: garbage				9
	fruit trees				1
	grain				1
	bird feeders				1
	dog food				1
	Nuisance	2			12
	Dead	1			
	Sightings				12
Raccoons	Nuisance				1
Bats	Nuisance			1	1
	Dead	1			
Muskrat	Nuisance	1			
Lagomarph	Nuisance	1			
Porcupine	Nuisance	1	1		
Skunk	Nuisance	1	1		
	Sick/Injured				1
Marmot	Nuisance		1		1
Squirrel	Nuisance	2	1		
	Dead		1		
Fox	Injured/sick	1			
Mountain lion	Sighting	5	5	3	4
	Chasing deer				1
	Nuisance			1	1
	Encounter	1			

Table 7. continued. Reports to Montana Fish, Wildlife & Parks personnel about wildlife within the city limits of Helena, Montana, 2004 - 2007.

Species	Interaction report	No. interactions / year			
		2004	2005	2006	2007
Birds:	Nuisance				1
Owl	Injured	1	2		1
	Dead			1	
	Injured				1
Nighthawk	Nuisance		1		
Woodpecker	Dead	1			
Finch	Dead	1			
Raven	Aggressive				1
	Injured				1
Duck	Injured			1	
Goose	Nuisance	1			
	Injured			1	
Hawk	Dead	1			
	Dead			1	
Grosbeak	Nuisance			1	
Magpie	Dead	4			
	Dead	1		1	
Flicker	Injured			1	
Falcon	Injured			1	
Seagull	Dead	1			
Crow	Injured		1		
Songbirds			1		
Deformed birds					



Barn owl casualty in Arizona. Photo courtesy of R. Hungerford.

small residential tracks or shifting priorities of using lands for wildlife instead of livestock (Haggenty and Travis 2006), stream alteration, disturbance of movements, restricting access to lands, nuisance wildlife concerns, and alterations of farmland. Their effects are increased because legislation is not adequate to compensate for the negative influences of urbanization (Henderson and O'Herren 1992).

Numerous studies have evaluated elk use in human-altered landscapes (i.e., logging, roads, fire, livestock) but how they are actually influenced by suburban and exurban development is relatively unpublished. However, these data are not new. In 1967 Klemmendson (1967:268) claimed big game winter range had been declining since the 1930s and that "…while something must be done to counteract the trend of dwindling winter habitat…it seems unlikely that existing priorities will greatly change." In the eastern U.S., big game winter range reduction has led to numerous problems. In the West, with adequate planning and foresight, the massive landscape changes may be planned effectively to benefit wildlife (Birris 1987, Wait and McNally 2004, Haggenty and Travis 2006). However, alterations in policy cannot linger. Elk, like all wildlife, belong to the public but as large landscapes are purchased in Montana and throughout elk habitat in the West, elk spend more time on private lands and are tolerated by landowners that do not tolerate elk hunting. In essence, this places large areas "out of administrative control" (Haggenty and Travis 2006). Ranch sales appear to be driving elk management in many areas as new owners consider the possession of elk as part of the transaction. "This perspective typically precludes a conceptualization of public access to private land for the purpose of harvesting elk as part of the necessary human ecology of elk management-the ecological commons in which hunters played the role of top predator has dissolved" (Haggenty and Travis 2006:828-829).

The ungulate-vehicle collision crisis

Deer-vehicle collisions are becoming an increasing problem across the U.S. and may result in reductions of deer populations, property damage to vehicles, and human injuries or fatalities (Romin and Bissonette 1996, Biggs et al. 2004). Over 1 million DVC occur annually in the U.S. (with approximately 50% going unreported; Conover et al. 1995, Romin and Bissonette 1996). Each year 29,000 human injuries and 211 human fatalities occur as a result of DVC (Conover et al. 1995). The economic losses associated with DVC may be substantial and vary state-to-state, but can exceed $2 billion annually across the U.S. in addition to losses associated with human fatalities (Romin and Bissonette 1996, Danielson and Hubbard 1999).

Although most DVC occur on rural, 2-lane, high-speed roads (Finder et al. 1999), they are becoming an issue in more urban areas that are experiencing an expansion of human and deer populations. Nielson et al. (2003) characterized DVC in urban areas and found that a higher proportion of DVC occurred in areas with fewer buildings and infrastructure, more patches and higher proportions of forest or shrub cover, more public land patches, and higher landscape diversity. These findings are similar to other studies involving DVC in rural areas (Bashore et al. 1985, Finder et al. 1999, Hubbard et al. 2000).

An overview of DVC management strategies and monitoring efforts was presented by Romin and Bissonette (1996) based on a survey distributed to state wildlife agencies throughout North America. Questionnaires were distributed to agencies regarding the number of deer killed annually on highways from 1982 – 1991, how the information was obtained, methods used to reduce deer mortalities on highways, and success of each technique based on personnel reports of scientific evaluation (Romin and Bissonette 1996).

Forty-three states responded to the survey and 35 reported yearly totals of deer mortality, although Montana was not among the states able to provide complete or accurate data (Romin and Bissonette 1996). Information was obtained for several other western states; Utah reported between 1,826 - 5,502 annual deer mortalities, Wyoming reported 987 - 1,756, California reported 15,000, and Colorado reported 5,202 - 7,296 (Romin and Bissonette 1996).

Of the 43 responding states, only Florida has not addressed deer-highway mortality (Romin and Bissonette 1996). Two states (4.7%) used highway lighting, 3 (7%) hazed deer, 6 (14%) modified habitat, 7 (16.3%) reduced speed limits, 7 (16.3%) built highway underpasses or overpasses, 11 (25.6%) used mirrors, 11 (25.6%) built fencing to deter deer from highways, 20 (46.5%) used warning whistles, 22 educated the public (51.1%), 22 installed swareflex reflectors (51.1%), and 40 (93%) used deer-crossing signs (Romin and Bissonette 1996). Between 62 - 95% percent of respondents using deer-crossing signs, public awareness programs, deer warning whistles, and swareflex reflectors reported that they had not conducted any scientific evaluation of the techniques and evaluation of success were based upon opinion (Romin and Bissonette 1996). Only 13 (30.2%) states have conducted scientific studies of mitigative strategies for reducing DVC and 9 of these considered only 1 technique (Romin and Bissonette 1996).

Although relatively few states use highway fencing and deer overpasses or underpasses, 91% of the 11 states using fencing and 63% of the 7 states using overpasses or underpasses reported these as effective strategies, although a majority of the responses were based on opinion (Romin and Bissonette 1996).

Hedlund et al. (2004) reviewed the most commonly-used mitigative strategies for reducing DVC, grouped into 3 categories: affecting motorist behavior, affecting deer behavior, and affecting deer populations.

Managing motorists.--Public education about DVC and speed limit reductions have not been sufficiently evaluated but are likely ineffective based on similar results from other campaigns (e.g., impaired driving and other stand-alone education programs; Hedlund et al. 2004). However, a public education campaign can be effective if it is actively enforced (i.e., the seat belt law) or provides information on time and site-specific situations, such as the beginning and location of a mule deer migration (Hedlund et al. 2004).

Passive signs such as deer-crossing signs are a commonly used technique, however they are used so frequently motorists probably ignore them (Putman 1997). Sullivan et al. (2004) tested temporary warning signs during mule deer migration. Travel speeds during this time dropped and DVC decreased by 50%, although the effect diminished during the second year of the study (Sullivan et al. 2004).

Active signs used to alert motorists when deer are near the road were evaluated in a study in Wyoming (Gordon et al. 2001). Vehicle speeds slowed when the sign was lighted but there is no corresponding DVC data (Gordon et al. 2001). More testing with active signs is needed and detection technology needs to be improved for these signs to be deemed effective (Hedlund et al. 2004).

Bashore et al. (1985) found an increase in DVC at the woodland-field interface where deer congregate and motorist visibility decreases. Additionally, the most important topographical feature influencing DVC in Illinois was the distance between the road and forest cover (Finder et al. 1999). Increased highway lighting (Reed and Woodard 1981, Hedlund et al. 2004) has also been ineffective at reducing DVCs. Clearing roadside vegetation increases driver visibility and decreases deer use of the area by reducing habitat quality near the road (Hedlund et al. 2004). Roadside clearing has been recommended in several studies (Putman 1997, Nielson et al. 2003, Hedlund et al. 2004), but stakeholder acceptance is needed as the public may feel habitat modification may affect deer activity or the environment (Nielson et al. 2003).

Managing deer behavior.--Ultrasonic warning whistles (Romin and Dalton 1992) are ineffective at reducing DVC, whereas swareflex and other reflectors have provided mixed results (Schafer and Penland 1985, Waring et al. 1991, Reeve and Anderson 1993, Putman 1997, Hedlund et al. 2004). Intercept feeding has reduced DVC almost 50% in some areas, but may only be useful on a temporary basis as deer may become dependent on supplemental food (Wood and Woolfe 1988).

When properly constructed and maintained, deer-proof fences deter deer from roadside vegetation (Falk et al. 1978, Ludwig and Bremicker 1983, Feldhamer et al. 1986). Fences must be sufficiently high and long to deter deer from jumping over or moving around a fence to feed (Bellis and Graves 1978, Ward 1982, Hedlund et al. 2004). Deer movements need to be taken into consideration and seasonal migrations or daily movements should not be inhibited if possible. Escape routes need to be established if deer enter the right-of-way (Hedlund et al. 2004). One-way gates along highway fences have been effective in moving mule deer off the highway right-of-way (Reed et al. 1974*a*), whereas deer guards have been ineffective for mule deer and elk (Reed et al. 1974*b*). Fencing is the cheapest and most effective measure for preventing DVC along short sections of road but becomes costly for large-scale management (Bashore et al. 1985).

Highway underpasses and overpasses have been effective (Ward 1982), although deer are sometimes reluctant to use them (Reed et al. 1975, Ward 1982, Reed 1981*b*) and they are costly to construct (Lehnert and Bissonette 1997). Gordon and Anderson (2004) evaluated the effectiveness of 6 highway underpasses for mule deer in Wyoming. Only 1 of the underpasses was used regularly, near a migration route, and deer use of underpasses was correlated to "openness" (entrance area divided by underpass length; Gordon and Anderson 2004). Gordon and Anderson (2004) recommended underpasses with an openness ratio of 0.8, being at >7 m wide and 2.6 m tall. Additionally, underpasses should be installed with fencing so deer can be funneled into them (Hedlund et al. 2004).

Lehnert and Bissonette (1997) assessed highway crosswalk structures, designed so deer cross at specific, well-marked areas where motorists could anticipate them along highways. In areas with these structures, deer mortality decreased 42.3% and 36.8% along 2-lane and 4-lane highways, respectively (Lehnert and Bissonette 1997). Deer-vehicle collisions in these areas were attributed to lack of motorist response to crosswalk warning signs, a tendency for foraging deer to wander outside the crosswalk boundaries, and the ineffectiveness of gates in allowing deer to leave the right-of-way (Lehnert and Bissonette 1997). Design of crosswalk structures could be improved through the implementation of flashing lights eliciting motorists' attention when deer enter the crosswalk, moving the fence close to the road so vegetation is more accessible to deer, and replacing gates with earthen ramps that allow deer to jump the fence in certain areas (Lehnert and Bissonette 1997).

Managing deer population.--Because increasing deer numbers have been correlated to increasing DVC, deer herd reduction has been considered an appropriate option for reducing DVC (Hedlund et al. 2004). Herd management has been rated the most effective DVC control strategy among state transportation departments and second among state wildlife agencies (second to fencing; Sullivan and Messmer 2003). Doerr et al. (2001) and Nielson et al. (1997) noted a decline in DVC after a deer population reduction program. However, herd reduction is controversial and oftentimes unacceptable to the public. However, communities are more likely to accept population reduction strategies if human safety is an issue (Messmer et al. 1997), therefore public education about DVC may elicit approval of population management (Nielson et al. 2003). Research is needed on the minimum area and amount of herd reduction needed for a desired effect on DVC (Hedlund et al. 2004).

Fencing, constructed with underpasses, overpasses, or crosswalks, are the most widely accepted and effective deterrent of DVC, although further research is needed on more management techniques (Romin and Bissonette 1996, Hedlund et al. 2004). Because the more successful mitigative strategies rely on certain aspects of deer behavior or movement patterns, Romin and Bissonette (1996) suggested future research should concentrate on deer habitat use and behavior in areas of high deer mortality before applying a certain management strategy as applicability of techniques may be species or site specific. For example, crosswalk, underpass, and overpass structures are likely more effective for mule deer than white-tailed deer, because mule deer have fixed migratory routes and DVC are confined to relatively few locations, whereas white-tailed deer collisions occur throughout substantial road lengths (Hedlund et al. 2004). Because elk also follow defined migratory routes, these structures may be beneficial in preventing elk-vehicle collisions. Herd reduction is unquestionably effective at a local level but public issues are raised (Hedlund et al. 2004). Combinations of certain techniques may also reduce DVC (Romin and Bissonette 1996).

There is no rapid, inexpensive method to reduce DVC. Deer vehicle collision mitigation should be included as part of an overall management strategy that balances the needs between humans and wildlife (Hedlund et al. 2004). The increase in suburban areas and trend in creating green space and wildlife habitat in these areas provides an excellent opportunity for careful management to maintain a desired natural resource while mitigating conflicts.

Managing Ungulates in the Exurban to Urban Gradient

Nonlethal control

Nonlethal management techniques include habitat modification, repellents, hazing, fencing, and translocation, among others. Nonlethal techniques are generally better accepted by the public (Witham and Jones 1992), however their high cost coupled with limited effectiveness prevents using them exclusive to lethal methods (DeNicola et al. 2000). Before implementing nonlethal control, cost-benefit analyses should be completed to ensure that benefits from application exceed costs of implementation (DeNicola et at. 2000). Some nonlethal options provide temporary relief from conflict (i.e., hazing and repellents) whereas others may permanently reduce conflicts (i.e., well-maintained fencing; DeNicola et al. 2000). The effectiveness of nonlethal techniques depends on species, deer densities, alternative food sources, and weather (DeNicola et al. 2000). Nonlethal control methods are designed to complement, rather than replace deer management, and are therefore most effective when incorporated with a comprehensive deer management program (DeNicola et al. 2000).

Capture.--Numerous techniques have been used to capture deer that are applicable in urban areas including drive and drop nets; box, clover, and corral traps; common-nets; net guns; and chemical immobilization (Peterson et al. 2003). In some situations, capture is facilitated with the use of bait-sites (Kilpatrick and Stober 2002). When capturing animals in the urban-rural gradient, there is higher visibility of the operation by the public and researchers need to be aware of the varied attitudes society has about capturing wildlife: from support to aggressive opposition. Because urban residents often prefer nonlethal techniques with minimal mortality (Jones and Witham 1990, Warrren 1995, Stout et al. 1997, Peterson et al. 2003), Locke et al.

(2004) developed a portable drive-net that was very effective for capturing urban deer (i.e., no mortality).

Translocation.--Translocation of overabundant ungulates in urban areas has been considered when protectionist opposition overrides culling operations (O'Bryan and McCullough 1985). However, translocation is not more humane than culling. The survival of translocated animals has been reported as low as 15% for animals translocated from Angel Island, California (O'Bryan and McCullough 1985). Others have also reported low survival of translocated deer following release (L.J. Temple and W. Evans, unpublished report, New Mexico Department of Fish and Game, Santa Fe, 1981; Jones and Witham 1990; Jones et al. 1997; Cromwell et al. 1999). Translocation is also expensive compared to lethal removal (Stradtmann et al. 1995). It costs (1984 prices) $73.95/deer when shot over bait in Wisconsin, compared to $112.79 - $569.77/deer for live removal (Ishmael and Rongstad 1984). The direct cost of moving deer from Angel Island, California was $431/deer at the time of relocation or $2,876/deer surviving for 1 year after release (O'Bryan and McCullough 1985). Translocation is not a satisfactory technique to reduce deer in overpopulated urban areas.

Immunocontraception.--Immunocontraception has potential to reduce deer densities. Between 30 and 70% of ecological carrying capacity but is more effective if the number of females being treated is ≤200 deer (Rudolph et al. 2000). In the few field studies that have been conducted, deer contraception reduced fertility from 72-86% (Kirkpatrick et al. 1996, McShea et al. 1997). The biological techniques are available for deer contraception to be effective in reducing urban deer herds when necessary. However, society and politics have hampered the scientific advancement of immunocontraception (Kirkpatrick and Turner 1997) as a solution to urban deer populations. Additional information is needed on the influences immunocontraception has on the dynamics of the population (Hobbs et al. 2000).

Other nonlethal techniques.--There are numerous devices on the market that claim to deter unwanted ungulates, however most have not been tested. The efficacy of a specific motion-activated light and sound emitting frightening device was tested on urban male deer and elk. Both species ignored the devices and were thus, ineffective (VerCauteren et al. 2005).

Bridge grating for deer-exclusion was found to be most efficient with 10.1x12.7 cm rectangular openings and diagonal cross members. These were 99.5% efficient for Florida Key deer exclusion and also safe for pedestrians and cyclists (Peterson et al. 2003). Domestic dogs have also been used to keep deer from specified areas. Trained Australian shepherds, blue heelers, and border collies may be effective in chasing deer in urban areas but more research is required (Beringer et al. 1994).

Repellents.--Deer are generalists and will feed on a variety of natural and ornamental plants when available. However, they do have preferences for some plant species, so planting unpalatable species may be effective in reducing deer damage to landscaping around homes and gardens (Conover and Kania 1988; DeNicola et al. 2000).

Conover and Kania (1988) evaluated winter browse preferences of 61 ornamental species in Connecticut. Deer damage was generally greater on evergreens than deciduous species, although resinous species such as pines (*Pinus* spp.), spruces (*Picea* spp.), and firs (*Abies* spp., *Pseudotsuga* spp.) were only browsed lightly (Conover and Kania 1988). Winter creeper (*Euonymus fortunei*), yews (*Taxus* spp.), and holly (*Ilex* spp.) were among the most heavily-browsed ornamentals (38.3 - 97.3% browsed; Conover and Kania 1988). However, there were large differences in deer browsing among species of the same genus, including the abovementioned, so caution should be used in grouping palatable and unpalatable plants by genus (Conover and Kania 1988).

Despite site- and condition-specific variation in deer preference, the use of some of this information may be valuable for homeowners and landscapers, enabling them to avoid planting highly-palatable species in areas with high probability of deer-conflict (Conover and Kania 1988). For instance, yews are among the most widely used ornamental plants in the northeastern U.S. and one of the most highly preferred foraging species for white-tailed deer in Connecticut (Conover and Kania 1988). The effectiveness of planting unpalatable species will vary based on deer densities, alternate food sources, and weather (Conover and Kania 1988, DeNicola et al. 2000).

Repellents reduce the palatability and attractiveness of treated plants to levels lower than those of other available forage and are more effective on less-palatable plant species (DeNicola et al. 2000). The results from repellent studies are not consistent in the scientific literature and odor-based repellents often out-perform their taste-based counterparts (Andelt et al. 1991, DeNicola et al. 2000). Egg odor is an effective repellent in some situations for white-tailed deer and mule deer (Andelt et al. 1991, DeNicola et al. 2000).

Thiriam, Big Game Repellent (BGR), and Hinder reduce white-tailed deer damage on woody browse (Swihart and Conover 1990, Conover 1984, Conover 1987, Sayre and Richmond 1992, El Hani and Conover 1997), however they have been ineffective in reducing crop damage or damage to large plots (Conover 1984, Hygnstrom and Craven 1988). Soap bars eliminated or reduced white-tailed deer browsing on apple trees and Japanese yews, whereas Ro-pel was ineffective at reducing white-tailed deer damage to Japanese yews (Swihart and Conover 1990). Fecal odors of predators and coyote urine have reduced browsing on woody plants by black-tailed and white-tailed deer (Melchiors and Leslie 1985, Swihart et al. 1991). Andelt et al. (1991) tested the effectiveness of various repellents and scents on mule deer. Chicken eggs, BGR, and coyote urine performed better than thiram, Hinder, soap, and Ro-pel (Andelt et al. 1999).

In some situations, repellents can be effective. Zinc dimethyldithiocarbamate cyclohexylamine complex and tetramethylthiuram disulfide were effective in reducing herbivory by white-tailed and mule deer in the Black Hills of South Dakota (Dietz and Tigner 1968). However, this is an exception. Because of the variables that influence the efficiency of repellents, the technique is always relative and susceptible to failure. Additional research in this arena is warranted (Witmer et al. 1995, Beauchamp 1997, Mason 1997).

Effectiveness of repellents depends upon the weather, hunger levels of deer, availability of alternate forage, palatability of the species being protected, deer density, and the concentration of repellents on treated vegetation (Conover and Kania 1988, Andelt et al. 1991, Mason 1997). Repellents will not deter browsing if deer are moderately hungry (Andelt et al. 1991). Due to their high cost, restrictions for use, and variable effectiveness, repellents are better suited for orchards, nurseries, gardens, ornamentals, and other high-value plants than lower-value plants in large-scale crops or pastures (DeNicola et al. 2000). Fencing or other alternatives should be used to protect vegetation in many cases (Caslick and Decker 1979, Palmer et al. 1985, Hygnstrom and Craven 1988, Andelt et al. 1991).

Lethal control

Lethal control of wildlife populations is not popular with the public but may be more acceptable when addressing health issues. Ungulates in urban areas can influence the epidemiology of many zoonotiz diseases. When ungulates and humans are in urban environments, the risk of emerging zoonotic pathogens increases (Hollis et al. 2000). When culling can be combined as a technique to test for diseases such as chronic wasting disease, urban populations could be reduced and tested together. More than 50% of the mule deer in Estes Park, Colorado can be sampled annually and diseased deer removed (Woolfe et al. 2004).

Bow-hunting.--In Rock Cut State Park, Illinois, biologists were not effective in reducing white-tailed deer by bow-hunting within 3 years (Ver Steeg et al. 1995). Short, intense hunt periods, separated by periods of no hunting, may increase the vulnerability of deer in urban areas to bow-hunting pressure (Kilpatrick and Lima 1999). Bow-hunting may be more efficient on residential properties adjacent to undeveloped land (Kilpatrick and Spohr 2000).

Sport hunting.--When urban populations of deer increase, they often create problems and the public is more tolerant of control measures. These hunts have to be well planned but they are popular, and quotas have been obtained resulting in deer population reductions (Hansen and Beringer 1997). Hunting programs in urban and exurban areas will become more popular as increases in human development are making it more difficult to successfully manage deer with traditional methods (Harden et al. 2005). Localized management of deer herds that have undesired effects on vegetation and other parts of the ecosystem have worked in New York (McNulty et al. 1997), Gerogia (Butfiloski et al. 1997), Connecticut (Kilpatrick et al. 2001), and other parts of the country by using sport hunters.

Sharpshooting.--Using police officers (Stradtmann et al. 1995) and others trained to kill deer efficiently with minimal disturbance to the public is another approach that has worked to reduce urban ungulates (Drummond 1995). Contracting with sharpshooters was cost effective, required fewer man hours to meet objectives (DeNicola et al. 1997), was quick and humane, and provided meat to local soup kitchens. Disadvantages included safety concerns, the need for highly skilled marksmen, and objectives of animal welfare advocates (Drummond 1995). Many of the disadvantages were minimized by using police officers as marksmen because of their ties to the community, training received, ability for the public to contact them and receive answers about the program, and their activities were covered by the city should a liability issue arise (Stradtmann et al. 1995).

Other lethal tools.--In urban areas where firearms are not allowed because of public opinion or local firearm ordinances, other tools are required when ungulate populations have to be reduced. Schwartz et al. (1997) immobilized deer with succinylcholine chloride delivered in biobullets followed by immediate euthanasia with a penetrating bolt gun. Euthanasia occurred an average of ≤20 seconds after collapse.

Public involvement

There are an array of techniques and management styles addressing problem ungulates along the rural-urban gradient. However, nearly everyone agrees that the single most important principle needed to successfully address the problem is community involvement with the planning process. By including municipal, county, state, federal, and other responsible entities into the process, objectives can be broadly established and solutions planned (McAninch and Parker 1991, Kilpatrick and Walter 1997, Lund 1997, Rutberg 1997). Curtis et al. (1993) summarized some of the lessons learned from the public involvement process.

1. Reaching a consensus may not be possible but that does not mean the program failed.
2. Emphasize problem-solving techniques so mechanisms for strongly-held minority opinions are built into the process.

3. Procedures for receiving comments from people in the community should be part of the process.
4. Attempt to involve all interests in the process, particularly those with the ability to block implementation of the recommendations.
5. Use the media aggressively to publicize management planning efforts.
6. Provide ample time and resources for the project to work.
7. Know the timelines and provide timely responses to participants.

Many of the management plans consider reduction by harvesting a certain number of individuals. However, because of the social biology and ecology of white-tailed deer, they form family groups of females. These females are highly philopatric to ancestral group ranges and by removing all deer from 400 – 2,000 ha may create voids in distribution of deer that may not be recolonized for up to 5 years (Porter et al. 1991). The approach described by Porter et al. (1991) is called the rose petal hypothesis. The older females occupy home ranges at the center of the group and younger individuals occupy overlapping home ranges that extend the periphery. These overlapping home ranges take form roughly analogous to the petals of a rose (Mathews 1989).

If the rose petal hypothesis is applicable, managers may want to consider pulsing controls that are more efficient than steady-state regimes. A pulsing control solution culls the population intermittently and can be effective when animals can be rounded up or baited. Success will also depend on the desired harvest, continual use of the pulsing control, and the acceptability of lethal control. Models for pulsing controls are presented by Rondeau and Conrad (2003).

Desert bighorn sheep

Human activities have altered and eliminated mountain sheep habitat in many areas of the southwestern U.S. Mountain sheep populations in North America have declined from >500,000 to approximately 70,000 and are among the rarest ungulates (Valdez and Krausman 1999). As humans continue to use mountain sheep habitat for development, recreation, resource extraction, or other uses, additional declines can be expected. Populations that have expired in Arizona, California, and New Mexico have been associated with human development (Krausman et al. 2001) but cause and effect studies were not carried out. However, human activities adjacent to sheep populations have been suggested as significant causes for declines (Giofriddo and Krausman 1986; Etchberger et al. 1989; Krausman 1993, 1997; Harris et al. 1995; Krausman et al. 1996; Rubin et al. 1998; Turner et al. 2004). Unfortunately, very little research has attempted to alter the problem. This may simply represent a situation that when urbanization advances rapidly it overwhelms natural ecosystems. When development occurs adjacent to and in mountain sheep habitat, sheep often decline and ultimately can become extinct. Society is faced with a difficult choice: restrict suburban expansion and control human activities within sheep habitat or accept the reality that bighorn sheep and expanding developments are not compatible (Krausman et al. 2001).

General Management of Urban and Exurban Landscapes for Wildlife

Successful management of wildlife along the rural-urban gradient is about managing people as much as managing wildlife.

> "People generally want wildlife in urban and suburban areas, even if they are ambivalent about some of the potential conflicts. Having a rich assemblage of native plants and animals around us is an indication that nature

> still prospers in the places where we dwell. It is a sign that *our* habitat still retains some of its ecological integrity. In the long run, the greatest benefit of having a well-connected system of habitats in our cities and suburbs …: children as well as adults will have abundant opportunities for contact with wild nature…And with a positive attitude toward nature they will likely be good citizens willing to support strong conservation measures for their broader environment" (Noss 2004:7).

Urban areas will "elicit their greatest loyalty, commitment, and stability when they function as places where people can consistently encounter satisfying connections with natural as well as economic and cultural wealth" (Kellert 2004:14). Furthermore, understanding artificial ecosystems (e.g., urban areas) may yield important insights to the management of natural ecosystems (Savard et al. 2000).

Because people have such a large investment in managing wildlife in urban areas, their attitudes have to be considered when formulating management plans. Human dimensions, which includes public attitudes, is also one of the triads of any successful wildlife management plan (Krausman 2000).

Studies have examined attitudes of residents (i.e., newcomers, longtime non-farm residents, and farm household residents) in exurban areas in Ohio (Smith and Sharp 2005), attitudes of residents in Arizona about elk management in the wildland-urban interface (Lee and Miller 2003, Heydlauff et al. 2005), and a series of studies of public attitudes across the U.S. (Decker and Gaavin 1987, Baker et al. 2004, Mannan et al. 2004, Silmer et al. 2004) discovering people's attitudes about urban wildlife. Surveys of attitudes are often general but can be specific. Wolch and Lssiter (2004) examined attitudes of African American women in Los Angeles, California toward wildlife, and Sasidharon and Thapu (2004) examined the ethnicity and variations in concerns for wildlife. In general, society supports urban wildlife even if they do not support the same species assemblages as natural areas (Noss 2004). Urban areas are important in the conservation and management of biodiversity. As such, urban planners increasingly integrate corridor systems to link habitat patches in urban areas with surrounding suburban, rural, and exurban lands (Adams 2005). When planners work with ecologists, connectivity can be maintained between anthropogenic development and wild landscapes to minimize the influence of urbanization on wildlife, sustain native populations, and further conserve biodiversity (Shaw et al. 2009).

Because of the broad jurisdiction over lands, only a few planning efforts have been conducted at sufficiently broad spatial scales to integrate conservation across the urban-rural gradient (Shaw et al. 2004). For example, Montana's Growth Policy Resource Book (2008) makes few statements related to wildlife. A recent effort to integrate urban and exurban planning on a large scale is Pima County's Arizona Sonoran Desert Conservation Plan (SDCP) described by Shaw et al. (2009) and Steidl et al. (2009).

The SDCP is centered around Tucson, Pima County, Arizona which is one of the fastest growing communities in the U.S. for decades. By 2000-2001, approximately1 ha of natural desert was being lost to urban development every 5 hours (Benedict et al. 2005). The population of Pima County is about 1,000,000 and most people are concentrated in the Tucson metropolitan area, leaving most of the county's 14,400 km^2 as wildlands, Indian reservations, rural, or exurban lands. To preserve and maintain the diverse and abundant flora and fauna, the SDCP (http://222.pima.gov/sdcp/) was "guided by these goals: (1) define urban form to prevent urban sprawl and protect natural and cultural resources; (2) provide a natural-resource-based framework for making regional land-use decisions; (3) protect habitat for and promote recovery of species listed under ESA; (4) obtain a Section 10 permit under ESA for a multi-species HCP;

and (5) develop a system of conservation lands to ensure persistence of the full spectrum of indigenous plants and animals by maintaining or restoring the ecosystems on which they rely, preventing the need for future listings" (Steidl et al. 2009:xx). The SDCP began in 1997 with the creation of the Science and Technology Advisory Team to advise the county. "More than a decade later, the planning process continues" with ≥6 significant accomplishments (Shaw et al. 2009:xx).

1. Unanimous adoption by the County Board of Supervisors of the SDCP and its conservation guidelines as an integral part of the County's comprehensive land-use plan.
2. Passage of a bond initiative in 2004 providing the $174 million for open space including at least $112 million for acquisition of lands and easements to protect land with high biological importance.
3. Purchase of land and easements that by 2008 had already protected over 30,000 ha of high priority conservation lands.
4. Involvement of hundreds of citizens in educational workshops and public hearings.
5. Involvement of more than 150 scientists as sources of information and as reviewers for the plan.
6. Development of a comprehensive, county-wide geographic database that enables sophisticated environmental modeling.

Important lessons have been learned from this long-term project (Shaw et al. 2009).

1. Conservation planning can provide a reliable basis for well-balanced land-use planning.

2. The goal of a conservation plan should be conservation, not compliance with the bureaucratic requisites of environmental legislation.
3. Science is a process and knowledge in land-use planning science performs rigor, consistency, and replicability; a mechanism to set goals; is transparent and accountable; should be evaluated and validated by experts; and use the peer-review process.
4. Separate scientific and political processes.
5. Urban and exurban landscapes are critical elements in land-use planning for conservation.
6. There are numerous secondary benefits of conservation in urban areas.

The scientific process of the SDCP is discussed in more detail by Steidl et al. (2009:xx).

The SDCP is designed to minimize urbanization, suburbanization, and exurban development within Pima County and to maintain and preserve habitats and landscapes where possible through land purchases and trades. The policy is sound; biologists may not know all the life history characteristics of the wildlife they are responsible for but are certain that viable populations will not exist without habitat. This concept is not new. In 1975, Hayden (1975) examined the effects of anthropogenic development on wildlife and their habitat around Lolo, Montana and made 9 management suggestions (most related to habitat).

1. Wildlife habitat can be managed and conserved best when in large undeveloped blocks that are not fragmented with subdivisions.
2. Natural barriers should be maintained between natural and altered landscapes.
3. Connectivity between subdivisions and wildland should be maintained.
4. Wildlife should not be harassed by domestic animals.

5. County commissioners need to be part of the planning process so fragile and critical habitats are not negatively influenced.
6. Education is warranted to develop a land ethic for those purchasing lands along the rural-urban gradient.
7. The use of conservation easements should be encouraged.
8. Additional tax breaks should be encouraged so large private landscapes are not fragmented further.
9. Zoning restrictions should be enacted and enforced to prevent development of riparian forests, sloughs, marshes, and other important valley bottom areas.

Each of these recommendations benefits habitats by minimizing disturbance and maintaining and enhancing wildlife habitat. To maintain large blocks of land, conservation development (i.e., cluster building with open space) has been in use for decades (Zipperer et al. 2000, Compas 2007, Milder et al. 2008) but rarely evaluated. Land trusts, landowners, and developers in the eastern U.S. used revenue from limited development to finance the protection of land and natural resources. Ten of these cluster developments (0.07-1.79 dwellings/ha) were randomly selected from 101 that had been identified. Each unit was designed to emphasize conservation. To evaluate their effectiveness as "conservation units," land alteration, edge effect, spatial configuration, and connectivity, impervious surface, riparian buffers, impacts to site construction targets, restoration, and land management were measured and contrasted with traditional subdivisions. The conservation and limited development projects protected sensitive conservation resources and resulted in significantly more conservation benefits than traditional subdivisions. More than 85% of the conservation developments were protected as interior habitat and the layout of development addressed conservation, restoration, and stewardship needs of the site-specific conservation targets (Milder et al. 2008).

In Gallatin County, Montana, planners and ecologists were also sensitive toward minimizing the impacts of exurban growth on habitat loss and fragmentation (Compus 2007). By contrasting development patterns from 1973 to 2004, Compus (2007:58) found that “(1) major subdivisions became more ‘clustered’ and less land consumptive, (2) minor sub-divisions revealed the opposite trend and are recently consuming more land, (3) distances from existing development decreased for major subdivisions, and (4) increasing numbers of parcels were near or within riparian areas. These findings indicated a differential impact of planning across scales and types of subdivision and a mixed success of planning in mitigating the environmental impacts of rural residential development.”

Maintaining connectivity is a relatively new approach in urban areas. As part of an overall biodiversity conservation strategy for Coquitlam, British Columbia, Canada, Rudd et al. (2002) performed a connectivity analysis to determine the numbers and patterns of corridors required to connect urban green spaces. For an urban area of 2,600 ha, with 54 green spaces (636.5 ha), Rudd et al. (2002) determined that 325 linkages were necessary to connect 50% of the green spaces. Within an urban zone, that would demand backyard habitat, roads, and right-of-ways be used for connectivity. More research will be required in this arena.

Large scale approaches to maintaining and preserving wildlands in Montana are underway. Acquisition and leasing important habitats by state wildlife agencies, federal natural resource agencies, and NGO (e.g., Montana Fish, Wildlife & Parks, U.S. Fish and Wildlife Service, Forest Service, Bureau of Land Management, National Park Services, the Nature Conservancy, The Trust for Public Land) provides for successful programs (Henderson and

O'Herren 1992). Some examples include Montana's acquisition and lease of thousands of hectares of winter range in the Blackfoot-Clearwater Wildlife Management area, acquisition of the Lee Metcalf National Wildlife Refuge, streamside Rattlesnake Greenway with Land and Water Conservation Funds administered by the U.S. Department of Agriculture, Forest Service, and the Montana Legacy Project. The Montana Legacy Project has plans to conserve important forest habitats in northwestern Montana owned by Plum Creek Timber Company. The goals are to preserve vital wildlife habitat and water resources, conserve traditional access for hunting, fishing, and outdoor recreation, and to keep sustainable harvesting operations in the forests and timber in local mills. This project has partnered with the Nature Conservancy and The Trust for Public Land and will purchase 126,467 ha of timberland for >$500,000,000. Details are being worked out but most lands will eventually be absorbed by the U.S. Forest Service and the State of Montana.

Large (i.e., The Montana Legacy Project) and smaller acquisitions or conservation easements are beneficial toward land preservation for outdoor recreation, education, protection of natural habitats and open space, and the preservation of historical areas and structures (Henderson and O'Herren 1992). Because of the positive benefits to wildlife and wildlife habitat, land trusts are becoming more popular and active throughout the state (i.e., The Montana Legacy Project, Montana Land Reliance, The Nature Conservancy, Five Valleys Land Trust, Gallatin Valley Land Trust, Flathead Land Trust, and Vital Grounds [Henderson and O'Herren 1992]).

Land use planners have to be part of the solution of maintaining biodiversity in the face of the rapid land use changes. Eight suggestions for land use planners and developers to consider how they impact natural landscapes were provided by Dale et al. (2000) and the Environment Law Institute (2003).

1. Examine the impacts of local decisions in a regional context.
2. Plan for long-term change and unexpected events.
3. Preserve rare landscape elements and associated species.
4. Avoid land uses that deplete natural resources over a broad area.
5. Retain large contiguous or connected areas that contain critical habitats.
6. Minimize the introduction and spread of non-native species.
7. Avoid or compensate for effects of development on ecological processes.
8. Implement land use and land management practices that are compatible with the natural potential of the area.

RESEARCH NEEDS

More than 90% of Americans work or live in metropolitan areas causing problems for wildlife and fisheries. Simply developing impervious surface cover ≥10% can be detrimental to sensitive aquatic species. When cover levels reach 25%, populations are impaired resulting in ecosystems that are non-supporting of native species (Center for Watershed Protection 2003). Because urbanization is a growing issue in North America, many of the problems aquatic species face have already occurred (e.g., containments, fish passage, impervious surface cover, altered fish assemblages, physical alteration of riparian areas). To enhance riparian habitats and species, habitat restoration of streams in urban areas will be necessary (Ehrenfield 2000). In exurban landscapes, managers, planners, and biologists should work together to meet their common goals and monitoring should become a standard practice so we can learn from the examples provided in rural and suburban landscapes.

The habitat for amphibians and reptiles altered by development will also need to be restored if the maintenance of native species is an important concern. In addition, roads need to be carefully planned as they fragment habitats (Carr and Fahrig 2001) and cut off connectivity to important habitat patches (Noël et al. 2007).

Avifauna are similarly negatively influenced by anthropogenic influences. Overall, species richness and diversity declines as urbanization increases. As such, numerous suggestions have been made to enhance and maintain avifauna. Unfortunately, the issues have not been tested and research will be necessary to determine their success in enhancing populations. Common research needs identified include educating residents about their negative impacts on wildlife, developing buffer zones around sensitive areas, maintaining native plant communities that are >1 ha, maintaining and enhancing riparian corridors, determining how urbanization influences insects, and determining if non-native vegetation can be used to restore habitats for avifauna. Cluster development of homes has also been suggested as a way to enhance wildlife habitats but additional research is necessary to determine the ecological role of cluster housing.

As humans infiltrate relatively undisturbed landscapes, they bring the problems society is now trying to deal with: domestic animals, anthropogenic food that is available to wildlife, use of key areas precluding use by wildlife, poor or no management, or management without enforcement. To overcome these problems if humans and wildlife are to coexist, we need to recognize the negative impacts caused by domestic animals and pass and enforce regulations to minimize them. Numerous problems are caused by uncontrolled anthropogenic food available to wildlife and some habitats are being altered to the extent that natural forage is no longer available. Each of these are serious problems that will cause challenges to those charged with managing wildlife resources and it will be an expensive, time consuming, and sometimes controversial process if habitats for wildlife along the urban-exurban gradient are to be maintained and enhanced. Planning with enforcement has to be a key ingredient or the unplanned, random, and chaotic urban development scheme will continue to alter habitats.

There are various calls for research to address the serious problems of urbanization and wildlife. The U.S. Environmental Protection Agency advocates 4 key areas of research (Munns 2006).

1. We need mechanistically-based extrapolation research to improve the basis for predicting the response of wildlife from existing information.
2. Researchers should coordinate efforts to determine how toxicology influences population biology across heterogeneous landscapes.
3. Researchers should identify techniques that adequately measure the numerous stressors on wildlife.
4. Researchers need to identify the spatial and temporal scales appropriate for wildlife risk assessments.

There are numerous stressors to terrestrial and aquatic resources along the urban-rural gradient: invasive and exotic species, nutrient enrichment, direct human disturbance, toxic chemicals, plus others (Munns 2006). To scientifically evaluate any stressors, we need better "…development of population dynamics models to evaluate the effects of multiple stressors at varying spatial scales, methods for extrapolating across endpoints and species with reasonable confidence, stressor-response relations and methods for combining them in predictive and diagnostic assessments, and accessible data sets describing the ecology of terrestrial and aquatic species" (Munns 2006:23).

In developing research to address the complex issue of altered wildlife habitat better descriptions of the urban-rural gradient are needed and the selection of research sites should be representative. Researchers should also ensure that the metrics used to determine influences are adequate so research can identify the mechanism affected by human activity that links population processes to community level patterns (Donnelly and Marzluff 2004).

The conflicts created for wildlife along the rural-urban gradient are human caused and will only be addressed properly (if they can) by collaborative cooperation. The first step in any program is to establish the objectives. What is it society wants: what species, how many, where, and when? Urbanization clearly has negative consequences to most wildlife and only with careful planning and monitoring will the full effects be known or successful programs documented. There is abundant information along the rural-urban gradient that can be used in managing the influence of exurban development on wildlife. It all begins with open communication, clear statement of objectives, and appropriate use of biophysical-behavioral, social-structural, valuation, and institutional-regulatory forces to achieve the objectives desired.

> "Unfortunately, as the habitat for wildlife deteriorates or at best barely holds its own, various major groups of citizens vitally interested in wildlife are locked in legal and conceptual babble over whether wild animal populations should be 'preserved' or should be managed and used. A substantial part of the public still wants to hunt or trap animals for sport or for profit. But a growing public contingent wants to prevent any use of animals except for viewing and esthetic appreciation. This ideological contest diverts energy and attention from the much more serious need for habitat preservation and restoration. It is a pointless exercise to save the life of some animal while the habitat in which it must live is disappearing. Unless the habitat is maintained there will be little wildlife left to hunt or to see. There is need for more political solidarity in the wildlife front, and less factionalism (A. Starker Leopold 1978)."

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