Effect of hunting and trapping on wildlife damage

Michael R. Conover

Abstract Hunting and trapping regulations are established so that these practices have little or no impact on those wildlife populations that do not cause damage. However, when wildlife populations cause significant damage, one reported benefit of allowing them to be trapped for furs or hunted is that these practices reduce wildlife damage below levels that would otherwise occur. Yet this reported benefit has not been examined critically. In this paper, I review the scientific literature to evaluate the hypothesis that hunting or fur trapping reduces wildlife damage. Hunting and trapping may reduce wildlife damage by 1) reducing wildlife populations below the environmental carrying capacity, 2) removing animals from the population before they would otherwise die, or 3) changing behavior of wildlife. It also can increase landowner tolerance of wildlife damage. Use of hunters and trappers is the most cost-effective method available to society to reduce wildlife populations, especially over large areas. Sometimes, efforts to use hunters and trappers to reduce wildlife populations are ineffective because there are too few hunters and trappers or too much land is off-limits to them. However, hunting and trapping can reduce populations below the environmental carrying capacity and reduce damage to crops from species which are trapped or hunted intensively, such as white-tailed deer (*Odocoileus virginianus*). For other game and furbearer species, hunting and trapping also may alleviate wildlife damage, but do so primarily by changing animal behavior. Finally, hunting and trapping may increase wildlife value and increase the willingness of landowners to tolerate damage from wildlife.

Key words animal behavior, deer, hunting, *Odocoileus,* outdoor recreation, trapping, wildlife damage management, wildlife management philosophy

Hunting and trapping have been enjoyed by Americans since humans first set foot on this continent. Recently, however, efforts to restrict or prohibit hunting and trapping in the United States (U.S.) have increased because in the modern, urban U.S., most citizens do neither and some nonparticipants object to these activities.

Hunting and trapping are used by state and federal wildlife agencies to manage wildlife populations according to objectives set by society. These objectives may be to maintain a wildlife population at a level that 1) creates the largest wildlife population, 2) is stable and immune to periodic population crashes, 3) produces the maximum sustained yield, 4) maximizes environmental benefits for other valuable species, 5) reduces spread of infectious disease or parasites within the population, or 6) reduces wildlife damage to acceptable levels. All of these objectives, except for the first, often require that a population be maintained at a lower level than would otherwise be the case. Hunting is used mostly to manage ungulates and other large mammals, some diurnal small mammals, and birds. Trapping is used to manage nocturnal, secretive, or semi-aquatic mammals such as marten (*Martes americana*), mink (*Mustela vison*), beaver (*Castor canadensis*), and muskrat (*Ondatra zibethicus*). A few species are hunted and trapped, such as coyotes (*Canis latrans*), red foxes (*Vulpes vulpes*), and raccoons (*Procyon lotor*).

Author's address: Jack H. Berryman Institute, Department of Fisheries and Wildlife, Utah State University, Logan, UT 84322-5210, USA; e-mail: conover@cc.usu.edu.

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Proponents of hunting and trapping often argue that a subsidiary benefit of these activities is their reduction of wildlife damage. Indeed, 90% of state wildlife agencies report that they adjust hunting seasons and bag limits to reduce wildlife damage (Conover and Decker 1991). However, the efficacy of hunting and trapping to reduce wildlife damage has not been substantiated (VerCauteren and Hygnstrom 1998). In this paper, I examine the hypothesis that hunting and trapping reduce wildlife damage.

Methods

It has already been demonstrated that selective removal of depredating animals by shooting or trapping can reduce wildlife damage (Wagner 1988, Dolbeer et al. 1993). Hence, I have limited this paper to the question of whether hunting or fur trapping does indeed reduce wildlife damage. Henceforth, I will refer to these activities as hunting and trapping.

Several potential mechanisms exist by which hunting or trapping might reduce wildlife damage. These activities may remove depredating animals or reduce wildlife populations below the levels that would otherwise occur or alter the behavior of animals. They also may increase landowner tolerance of wildlife damage. I surveyed the published scientific literature and the United States Fish and Wildlife Service's Reference Service for papers that either support or reject the likelihood of any of these mechanisms actually occurring.

An ideal way to assess effects of hunting and trapping on wildlife damage would be to make a simultaneous comparison of areas where a wildlife population is hunted or trapped to other areas where it is not. Another approach is to compare the same wildlife population during 2 periods of time: when it is hunted or trapped and when it is not. Unfortunately, studies that used either of these approaches are uncommon, owing to the difficulty of conducting large-scale studies at many sites. Consequently, I evaluate these comparative studies along with other studies to determine whether the scientific literature supports the hypothesis that hunting or trapping reduces wildlife damage.

Results and discussion

Are the species that are hunted or trapped the same as those that cause damage?

Most wildlife damage to agricultural crops is caused by the following species, which are listed in order of how frequently U.S. agricultural producers cited them as a cause of wildlife damage (Conover 1994): deer, woodchucks (*Marmota monax*), raccoons, coyotes, mice and voles (Rodentia), rabbits (Leporidae), beaver, foxes, skunks (Mustelidae), waterfowl (Anatidae), prairie dogs (*Cynomys* spp.), bears (Ursidae), and elk (*Cervus elaphus*). All of these species are harvested by either hunters or trappers, except for mice and voles.

Is hunting or trapping a cost-efficient method to remove animals from a population?

Hunting and trapping are the most cost-effective methods available to society to manage some wildlife populations. If government had to pay employees to manage wildlife populations, the cost often would be prohibitive. In contrast, many people enjoy hunting and trapping and do not have to be paid to do it. Rather, hunters and trappers pay for the privilege of hunting or trapping by purchasing licenses from the state and federal governments. Over large areas, such as an entire state, hunters and trappers are the only feasible method to impact wildlife populations. Even in small areas such as in municipalities and state parks, use of hunters or trappers is usually the most cost-effective method to remove animals. In Ohio, the Columbus and Franklin County Park District reported that it cost $133 to relocate a deer, $207 to kill it using a sharpshooter, and $45 to kill it using hunters (Peck and Stahl 1997). Palmer et al. (1980) reported that it took 1.8 hours of supervisory time for each deer removed from an enclosed federal installation in Ohio during a public hunt. In contrast, it took 2.8 hours to remove a deer using a box trap, 3.3 using biologists as shooters, 4.1 using dartguns, and 6.9 using rocket nets.

Do hunting and trapping reduce wildlife damage by keeping populations below the environmental carrying capacity, removing depredating individuals, or both?

All wildlife populations are limited by some factor or factors; none increases forever. Many wildlife populations are limited by food; as animal populations increase, food supplies decrease until some combination of increased juvenile mortality, increased adult mortality, disease, delayed sexual maturity, and decreased birth rates causes the animal population to decline. The equilibrium point between an animal population and the capacity of the land to sustain it is called the carrying capacity (*K*), potential carrying capacity, biological carrying capacity, or environmental carrying capacity (Macnab 1985). Henceforth, I will refer to this as the environmental carrying capacity (ECC). Often, starvation and disease have a debilitating effect on animals rather than causing immediate death. Under these conditions, wildlife populations can remain above ECC for some time before an event (such as a hard winter) causes a mass die-off (Warren 1991). By the time such a die-off occurs, adverse ecological effects may have already happened. Consequently, delays occur in the recovery of food supplies (the plants that sustain the herbivore population or the prey that sustain the predator population), postponing the recovery of the animal population. This irruptive type of population cycle (irruptions followed by crashes) often is the norm in food-limited populations (Caughley 1970, Macnab 1985, McCullough 1997).

Society usually limits hunting and trapping to those species, such as muskrats, that are not obviously affected by these activities (Errington 1946, 1956; Errington et al. 1963). That is, most hunting and trapping mortality is considered to be compensatory, meaning that the deaths caused by hunting or trapping replace other forms of mortality that would have occurred otherwise. Hence, the population during the next breeding season would be the same with or without hunting or trapping. If, however, hunting or trapping mortality was so extensive that it caused a reduction in the subsequent breeding population, then this mortality would be considered to be additive to the natural causes of mortality. For most wildlife species, hunting and trapping are compensatory to a threshold point and additive thereafter, which means that below a certain level, hunting or trapping mortality has little effect on populations but as it increases past that level, it will reduce populations (Burnham and Anderson 1984, Nichols et al. 1984, Clark 1987). For instance, javelina (*Tayassu tajacu*) can withstand annual harvest rates of 15-30%, but when harvest rates increased to 65%, javelina populations declined (Day and Smith, unpublished report; Ellisor and Harwell 1979; Green et al. 1985). This implies that society can manipulate wildlife populations by adjusting hunting or trapping mortality rates. These rates can be increased or decreased by changing length or timing of the hunting or trapping season, number of people allowed to hunt, number of animals each person is allowed to harvest (bag limits), and by limiting harvest to only one sex or age class of animals.

For most game species, there is no need to reduce populations below the ECC unless the desire is to avoid population cycles. However, allowing some wildlife populations, especially those that are causing damage, to fluctuate around the ECC may not be in the best interest of society, the habitat, or the wildlife species. In these cases, the optimal wildlife population may be one that is maintained at a level below that of the ECC, because the benefits and liabilities caused by a wildlife population change with the size or density of the wildlife population (Figure 1). Generally, when a wildlife population is low, the benefits provided to society (positive values) outweigh the problems (negative values) because the animal is too rare to cause any major problem. As a wildlife population increases and the animal becomes abundant, negative values may increase faster than positive ones. If the population continues to increase, negative values may actually outweigh positive ones. At that point, society would consider the animal to be a "pest." Therefore, the goal of wildlife management is to keep a wildlife population at a level where its net value for society is maximized (Figure 1).

If a wildlife population is going to be maintained at any level below that of the ECC, society often must intervene by either increasing the species' mortality rate or decreasing its natality rate. While governmental wildlife agencies' intent may be to reduce wildlife damage by maintaining some

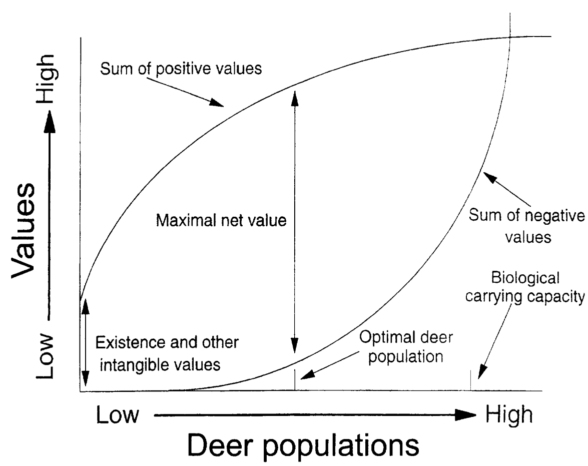

Figure 1. Hypothetical relationship between an increasing deer population and its positive values (benefits it provides) and negative values (wildlife damage) for society.

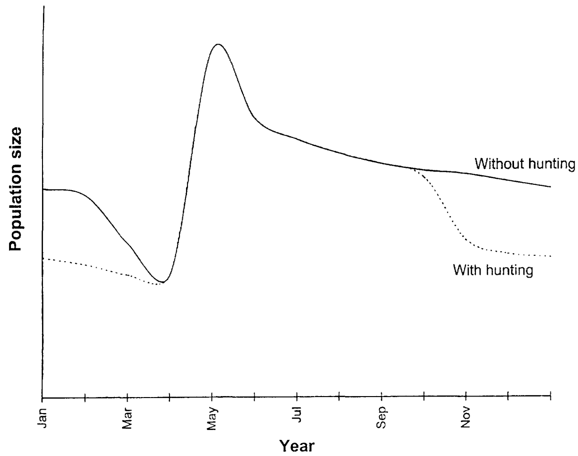

Figure 2. Annual changes in a hypothetical wildlife population when it is subject to hunting and when it is not (assuming that all hunting mortality is compensatory).

wildlife populations below the ECC by using hunters or trappers, their efforts may not remove enough animals to reduce a wildlife population to the level desired by society, either owing to insufficient numbers of hunters or trappers or because too much land is off-limits to hunters and trappers (Heusmann 1999).

Even when hunting or trapping is not intensive enough to suppress wildlife populations, these activities can still reduce wildlife damage by removing individuals from the population, some of which may have caused damage had they been allowed to live longer. Many wildlife species in North America reproduce only in spring or summer, and populations are largest then. Mortality rates are normally high when the young are still small and relatively helpless, but gradually decrease in fall as the young mature and become more self-sufficient. Winters are usually a difficult time for wildlife, and mortality rates increase as winter food supplies are depleted (Figure 2). Therefore, winters often cause population bottlenecks because there often is only enough food or cover for a certain number of individuals to survive. If the hunting or trapping season occurs in fall, one consequence is that it removes animals from the population earlier than would normally be the case. For instance, if hunting is conducted in September and the winter bottleneck normally occurs in February, hunting will reduce the population for several months even if hunting mortality is completely compensatory (Figure 2). The effect on wildlife damage of removing animals by hunting will depend on the time of year wildlife damage occurs. If the damage occurs during or after the hunting or trapping season, the amount of damage should be reduced by the same proportion as the wildlife population was reduced by hunting or trapping, assuming that depredating and non-depredating animals are equally vulnerable to hunting or trapping. If, however, the damage occurs before the hunting season begins or after the winter bottleneck, then hunting will not reduce wildlife damage, unless the hunting or trapping intensity is sufficient to reduce the population below the ECC.

Obviously, the impact of hunting and trapping on wildlife populations and damage will vary by species. Hence, in the following section I examine the impact of hunting and trapping on some wildlife species for which there are data.

White-tailed deer. Deer cause considerable damage and other problems for society. These include >$500 million in damage to agricultural crops and at least this much in loss of timber productivity (Conover 1997*a*). Over 1.5 million deer-car collisions occur in the U.S. annually, which result in over $1 billion in damage to automobiles, 16,000 human injuries, more than 200 human fatalities, and 1.4 million deer fatalities (Conover et al. 1995). In 1998, there also were over 16,800 human cases of Lyme disease in the U.S. (Centers for Disease Control and Prevention 1999), and deer are an important reservoir for the disease (Conover 1997*a*).

Prior to the arrival of European colonists in North America, white-tailed deer densities were estimated at 3-4/km^2 and were limited by food and predation by mammalian predators such as wolves (*Canis lupus*) and cougars (*Puma concolor*), and by the hunting activities of Native Americans (McCabe and McCabe 1984). Today, predation is no longer sufficient to limit ungulate populations in most of North America (Mech 1984). However, in areas where a sufficient proportion of the land is open to hunting, this form of mortality can be sufficient to maintain ungulate populations below the ECC (Behrend et al. 1970).

Hunting's effect on reducing deer damage and controlling deer populations can best be observed by comparing areas where hunting occurs to similar areas where deer are protected. Inside Gettysburg National Military Park in Pennsylvania, white-tailed deer densities were estimated to be 28/km^2, versus densities of 8/km^2 in the surrounding county where hunting occurred (Vecellio et al. 1994, Frost et al. 1997). Based on forested areas, deer densities/forested km^2 were 136 inside the park versus

24 outside (Frost et al. 1997). Inside the park, the large deer herd was impacting forest regeneration, causing an unacceptable level of deer-automobile collisions, and preventing some crops (e.g., corn) from being grown without substantial damage (Frost et al. 1997). In contrast, corn losses to deer averaged less than 0.5% in the U.S. (Wywialowski 1996). The goal of this park was to restore the landscape to the environmental conditions that prevailed during the 1863 battle. Both agricultural crops and forests were part of this historic landscape, and overbrowsing by deer was in conflict with this goal (Underwood and Porter 1991). Hence, a controlled deer hunt, started during 1995, removed 503 deer the first year and 355 the second. Following this, corn and other historic crops were grown successfully in the park for the first time in 8 years and fee waivers for crop damage were eliminated (Frost et al. 1997).

The mission of the 1,400-ha National Zoo's Conservation and Research Center in Virginia was to maintain many of the world's wildlife species in captivity. To keep costs down, hay was produced in the Center to provide food for the captive animals. Hay production declined from 68 metric tons in 1978 to 8 metric tons in 1980 and 0 in 1981, when spotlight counts showed >400 deer using 250 ha of alfalfa and other fields. More seriously, the large deer herd (33-97 deer/km^2 depending on season and census method) was threatening the health of the captive animals. In 1979, lungworm (*Dictyocaulus viviparus*) was found in bactrian camels (*Camelus bactrianus*); in 1981, 6 reindeer (*Rangifer tarandus*) died from an infestation of meningeal worms (*Parelaphostrongylus tenuis*). Because of these problems, the Center was opened to hunting in 1981, when 126 deer were taken and the problems subsided (McShea et al. 1993).

The largest state park in Indiana, Brown County State Park (6,350 ha), was by 1972 experiencing a deer population 4-8 times larger than on the adjacent national forest where hunting occurred; deer browsing was impacting the regeneration of the endangered yellowwood tree (*Cladrastis lutea*, Mitchell et al. 1997). In response, a controlled deer hunt was conducted inside the state park, during which 466 hunters removed 392 deer (Mitchell et al. 1997). This hunt, however, drew so much publicity and controversy that the hunt for the next year was canceled. This decision, in turn, produced enough concern about the effect of high deer populations that the state passed a law mandating repeated hunts on any state property where any wildlife species threatened human health or the health of the ecosystem (Mitchell et al. 1997).

In many other parks, overgrazing by large deer herds has had a detrimental effect on native plants and animals (Warren 1991). In Great Smoky Mountain National Park in Tennessee, plant diversity declined due to high deer numbers (Bratton 1979). In Saratoga National Historic Park in New York, deer populations were high (50-60 deer/km^2) and deer browsing prevented recruitment of tree seedlings. On Cumberland Island National Seashore in Georgia, deer browsing suppressed the regeneration of live oak (*Quercus virginiana*), the tree that dominated the native forest (Warren 1991). Such problems rarely occurred in areas where there was extensive deer hunting.

Raccoons. Raccoons cause more nuisance problems for metropolitan residents than any other wildlife species (Williams and McKegg 1987, Conover 1997*b*). Their damage to corn and vegetable crops is a problem to farmers (Conover 1998), and their predation on eggs impacts the reproductive success of turtles and birds. Raccoons also serve as vectors for rabies (Jenkins and Winkler 1983, Krebs et al. 1995) and the ascarid parasite, *Baylisascaris procyonis* (Georgi 1983).

Hunting and trapping are the most important mortality factors for adult raccoons throughout most of North America (Fritzell and Greenwood 1984, Sanderson 1987, Clark et al. 1989). During the 1994-1995 trapping season, 1,899,000 raccoons were harvested (Table 1). Despite this high

Table 1. Annual fur harvest in the United States during 1994–1995 trapping season of those species which are responsible for large amounts of wildlife damage (data from Linscombe 1996).

Species	No. harvested annually	Value in U.S. dollars
Muskrat	2,186,000	$4,540,000
Raccoon	1,889,000	15,092,000
Beaver	455,000	6,604,000
Nutria (*Myocastor coypus*)	191,000	393,000
Red fox	186,000	2,512,000
Coyote	179,000	2,081,000
Virginia opossum (*Didelphis virginiana*)	172,000	201,000
Gray fox (*Urocyon cinereoargenteus*)	83,000	789,000
Skunk	73,000	316,000
Cougar	2,000	247,000

harvest rate, raccoon populations have increased 15-20 times from the low levels that occurred in the 1930s (Sanderson 1987). This has resulted from high reproductive rates, an ability to thrive in human-dominated landscapes, and an expansion of their range northward into Canada and westward in the U.S. This expansion has occurred despite the popularity of raccoon hunting and trapping. One reason is that most raccoon hunting and trapping seasons are purposely set so that the mortality is compensatory. Another factor has been the low fur prices in recent years.

Cougars. Cougars are a threat to livestock and people, and, depending on relative densities of cougars and deer, they may significantly affect deer populations. However, studies indicate that although individual cougars kill 13 to 20 deer annually, cougar predation did not limit deer populations when the cougar:deer ratio was 1:135 to 1:201 (Mech 1984 citing Hornocker 1970). On the other hand, studies in California indicate that predation by cougars decreased bighorn sheep (*Ovis canadensis*) populations and was the driving force behind habitat shifts of the bighorn sheep (Wehausen 1996). Wehausen (1996) hypothesized that if bighorn sheep were driven into suboptimal habitat by cougar predation, some sheep populations would be extirpated.

Efforts to suppress cougar populations in the nineteenth and early twentieth centuries were largely successful, but cougar populations recovered once they were afforded some protection from human predation in the western U.S. During the 1994-1995 season, 2,000 cougars were harvested in the U.S. (Table 1). Where hunting occurred, it was typically the largest cause of cougar mortality (Ross and Jalkotzy 1992). Within their study area, Ross and Jalkotzy (1992) found that up to 21% of the "harvestable" population (11% of total population) were taken. In Wyoming, hunting mortality of cougars ranged from approximately 4% to 31% between 1976 and 1984 (Logan et al. 1986). Lindzey et al. (1988) found a 28% annual mortality rate in an unharvested cougar population in Utah; when a simulated 27% sport harvest rate was imposed on this population, the mortality was largely compensatory but still caused some reduction in some local cougar populations (Lindzey et al. 1992).

In California, sport hunting of cougars ended in 1972. In 1990, a ballot initiative (Proposition 117) changed the status of cougars to a "specially protected mammal," prohibited cougar hunting, and restricted the killing of cougars under depredation permits (Mansfield and Charlton 1998). Since the end of sport hunting, cougar populations in the state have increased (Torres et al. 1996). There also has been an increase in number of cougar attacks on humans. From 1885 to 1985, there were 2 confirmed cougar attacks on people in California, resulting in 3 human fatalities. From 1985 to 1997, there were 8 attacks on people, including 2 human fatalities (Mansfield and Charlton 1998). There also has been a recent increase in the number of pets and livestock killed by cougars in California (Torres et al. 1996).

Coyotes. Each year, coyotes attack people (Baker and Timm 1998). They also kill over 65,000 cattle and calves valued at over $24 million (United States Department of Agriculture 1992) and over 300,000 sheep and lambs worth over $13 million (United States Department of Agriculture 1991). Coyote predation also can significantly impact survival of pronghorn (*Antilocapra americana*) fawns (Smith et al. 1986) and deer fawns (Stout 1982, Hamlin et al. 1984, Mech 1984, Knowlton and Stoddart 1992). In Maine during winter, coyotes kill adult deer, most of which are healthy. Hence, these losses are considered to be additive (Lavigne 1992). For all of these reasons, state and federal governments have maintained programs to reduce coyote predation on livestock, and many states do not regulate the harvest of coyotes through quotas or seasons. However, despite these efforts, coyote densities have not diminished except in small, localized areas (Wagner 1988). Instead, coyote populations have increased during the twentieth century and their range has expanded into the eastern U.S.

Urban Canada geese. Breeding populations of Canada geese (*Branta canadensis*) have increased in North American cities to the point where the birds are considered nuisances (Conover and Chasko 1985, Chasko and Conover 1988). In response to numerous complaints about geese, Massachusetts conducted special hunting seasons on urban geese beginning in 1988. The special seasons were timed to occur either before (early September) or after (January and February) flocks of migratory Canada geese had moved through the state. During the 1997 special hunting seasons, 25% of Massachusetts' urban goose population was harvested. Yet this harvest rate was insufficient to reduce the size of the goose population because of its high reproductive rate (Heusmann 1999).

Beavers. History has clearly demonstrated that trapping can be used to suppress beaver populations. In fact, uncontrolled trapping during the 1700s and 1800s led to decline of beaver populations across the U.S. and extirpation of this species from large areas of its former range (Trefethen 1975). Today, in much of North America, beaver populations have largely recovered and the problem of too many beavers is more common than the problem of too few. When their populations are too high, beavers can destroy their habitat by overbrowsing willow (*Salix* spp.) and aspen (*Populus tremuloides*, Todd 1981). This, is turn, may lead to an increase in economic damage as beavers switch their foraging to valuable timber species and disperse into agricultural or developed areas where they are unwanted (Todd 1981). Hence, beaver management often involves trying to harvest the surplus animals. However, in the last few decades, low pelt prices have not provided enough incentive for trappers to accomplish this goal. Consequently, beaver populations and their damage have been increasing in many parts of the country (Bhat et al. 1993).

Muskrats. High muskrat populations can overgraze aquatic vegetation, causing a substantial loss of emergent vegetation and damage to their habitats (Errington et al. 1963). This loss of aquatic vegetation also can adversely impact waterfowl that prefer a mix of open water and emergent vegetation (Neal 1977, Todd 1981). During the 1994-1995 trapping season, over 2,000,000 muskrats were harvested in the U.S. (Table 1). Due to their high reproductive and mortality rates (annual mortality rates near 90% in unharvested populations due to diseases, starvation, or adverse weather; Schmitke 1971) and high immigration rates (Simpson and Boutin 1989), most trapping mortality on muskrats replaces normal forms of mortality so that few muskrat populations are controlled by fur trappers (Neal 1977, Todd 1981, Clark 1987). However, high harvest rates have suppressed some local muskrat populations (Boutin and Birkenholz 1987).

Thus hunting or trapping may keep some wildlife populations (e.g., deer, cougars) below the ECC, but not others (e.g., raccoons, coyotes, urban Canada geese, beavers, muskrats). Another way hunting and trapping may be reducing wildlife damage is by causing animals to alter their behavior.

Do hunting and trapping reduce wildlife damage by changing animal behavior?

All forms of nonconsumptive outdoor recreation (e.g., hiking, camping, boating, photography) can adversely impact wildlife (Boyle and Samson 1985, White et al. 1999). Most of these recreational activities also cause the animals to lose their fear of humans. For instance, some bears have lost their fear of humans through frequent, innocuous contacts with people; this has produced serious and sometimes fatal consequences for bears and humans (McCullough 1982, Albert and Bowyer 1991). Coyote attacks on humans have become more common in recent decades as urban coyotes have lost their fear of humans (Baker and Timm 1998). Cougar attacks on people also have increased (Mansfield and Charlton 1998). As McCullough (1982: 31-32) so eloquently stated, "It seems that earlier grotesque beliefs that predators were evil and had to be eradicated to make the world safe for lambs were overthrown at the cost of creating a new and beguiling myth of the benevolence of nature. With each new report of a human injured or killed by a bear we question where management went wrong. We failed to recall that most problems with bears in parks stem not from human malevolence but from too much benevolence. As with most conflicts between powerful adversaries, it is dangerous to appear weak."

Hunting and trapping are 2 of the few human activities that reinforce an animal's fear of humans. When hunted, black-tailed prairie dogs (*Cynomys ludovicianus*) spent more time in alert posture, less time foraging, and could no longer be closely approached by humans (Vosburgh and Irby 1998). In response to hunting and trapping, animals tend to avoid areas where they might come into contact with humans. For instance, wolves and black bears avoid areas with roads, especially those frequented by humans, because such roads increase their vulnerability to hunting and trapping (Mech et al. 1988, Brody and Pelton 1989, Thurber et al. 1994).

When exposed to intensive hunting pressure, deer may become more wary, shift their home ranges, become more nocturnal, and spend more time in dense cover or other areas where they are safe from hunters (Sparrowe and Springer 1970, Pilcher and Wampler 1982, Swenson 1982, Kufeld et al. 1988, Root et al. 1988). Such behavioral changes reduce vulnerability of deer to hunting (Swenson 1982) and should reduce extent of deer damage to crops because the deer would be less willing to venture out into an open field or areas frequented by humans. In Gettysburg National Military Park, deer often ventured far (>200 m) into agricultural fields to forage before deer hunting was permitted (Vecellio et al. 1994); in areas where deer are hunted, deer damage usually occurs only along a field's edge, close to cover (Conover 1989). Deer normally shift their home ranges closer to fields when they contain palatable plants (VerCauteren and Hygnstrom 1998), and for this reason number of deer foraging in an agricultural field can increase rapidly. In contrast, deer shift their home ranges away from open areas and into dense cover during the hunting season (VerCauteren and Hygnstrom 1998). Hence, hunting can prevent the local buildup of deer around vulnerable fields. Adkins and Irby (1994, 1997) presented data from Montana indicating that most complaints (63%) about wildlife damage came from areas where hunter access was limited. They suggested that these unhunted lands acted as local refuges and that cooperation of all landowners in areas with game-damage problems was necessary to alleviate the problem. Bhat et al. (1993) made the same argument about the need for area-wide trapping of beavers. Likewise, most crop damage caused by waterfowl occurs when the birds are concentrated. This is particularly a problem in fields around federal wildlife refuges where waterfowl may congregate in the hundreds of thousands in the fall. In such cases, hunting can be used to disperse depredating birds (Rusch et al. 1985).

A common nonlethal technique to reduce wildlife damage is using human effigies (scarecrows) and devices that produce a loud bang (e.g., firecrackers, propane exploders, cracker shells). The main limitation with nonlethal techniques is that animals quickly learn that they are harmless and habituate to them (Conover 1981). One way to delay this habituation to human effigies and cannons is to occasionally shoot an animal so that others' fear of humans and loud noises is reinforced. Hence, hunting should increase effectiveness of these nonlethal techniques.

Most of the effect of hunting on animal behavior results because animals learn to associate humans with the loud noise produced by a firearm and its lethal consequences. In contrast, humans are rarely present when a trap is sprung, making it more difficult for an animal to associate the negative consequences of trapping with humans. Thus, trapping may not cause the same fear of humans that hunting does, but it may enhance an animal's general wariness and decrease its willingness to venture far from what it considers secure cover.

Do hunting and trapping increase landowner tolerance of wildlife damage?

Wildlife damage is a serious problem for landowners, especially those who manage their property for timber or agricultural production (Conover et al. 1995, Conover 1998). Most farmers suffer a loss of productivity due to wildlife damage. In a national survey, 80% of farmers responding reported suffering wildlife damage in the prior year and 54% reported that their yearly losses from wildlife exceeded $500 (Conover 1998). These losses occurred despite a mean annual expenditure of over 40 hours and $1,000 by each farmer trying to solve or prevent wildlife damage. Usually, the only lethal option available to farmers who experience wildlife damage is to increase hunting or trapping on their farms. They also may be issued special depredation permits that allow them or their agents to shoot or trap wildlife outside the normal season. Most farmers believe that these techniques are helpful (McIvor and Conover 1994). Hence, the elimination of hunting and trapping would deprive them of one of the few techniques they can legally use and in which they have confidence.

Much (45%) of the land base in the United States is controlled by agricultural producers, who make the decisions about how wildlife habitat will be managed on their property (Conover 1994). Hence, their perceptions about wildlife are important. Unfortunately, 53% of U.S. agricultural producers reported that the wildlife damage they were experiencing exceeded their tolerance. This, in turn, can reduce a farmer's willingness to manage property for wildlife. In a national survey, 38% of farmers reported that they would oppose the creation of a wildlife sanctuary near their farm due to their concerns about wildlife damage and 24% stated that wildlife damage had reduced their willingness to provide habitat for wildlife on their property (Conover 1998).

Landowners' perceptions of wildlife are based on the relative importance of benefits and liabilities they derive from it. Though wildlife can be detrimental to a farm's profitability, it also provides benefits to landowners, and thus most agricultural producers are much more tolerant of wildlife damage. Many of these benefits are derived from the opportunity to hunt or trap on their property. Most agricultural producers (77%) in the U.S. either hunt or allow their friends and relatives to hunt on their property (Conover 1998). Farmers who hunt deer were more likely than nonhunting farmers to improve wildlife habitat, to favor an increase in the local deer population, and to be tolerant of deer damage (Tanner and Dimmick 1983).

Hunting and trapping also provide economic incentives to landowners and thereby increase landowners' tolerance of wildlife damage. Five percent of agricultural producers in the U.S. make money by charging hunters a fee (Conover 1998), and in some areas this source of income can exceed the amount derived from livestock or timber production (Burger and Teer 1981). This provides a strong incentive for rural landowners to maintain habitat for wildlife (Williams and Lathbury 1996). In Wisconsin, 24,000 ha of private land are licensed by the state as fur farms and managed to provide habitat for furbearers (Payne 1980). Agricultural producers who profit economically from the presence of wildlife are likely to be more tolerant of wildlife damage. Actually, many rural landowners benefit economically from wildlife, but that benefit is indirect. In some parts of the U.S., rural land with abundant game sells for more than land without it (Bolle and Taber 1962, Applegate 1981). Real estate agents realize this, and many real estate advertisements mention that game species occur on the property.

Although hunting and trapping help to reduce wildlife damage, they are not sufficient to end it. Some wildlife damage is caused by species that are not hunted or trapped, such as red-winged blackbirds (*Agelaius phoeniceus*) and starlings (*Sturnus vulgaris*). Some wildlife damage occurs in areas where hunting or trapping is prohibited or cannot be conducted safely, such as within cities, or in areas where there are insufficient hunters and trappers to keep wildlife populations in check. Some wildlife damage is caused by individual "problem animals." An example would be a cougar that has developed a taste for mutton, an alligator (*Alligator mississippiensis*) at a popular swimming hole, or a moose (*Alces alces*) in downtown Anchorage. The specific problems caused by these individual animals may not cease until they are killed or relocated. Often, removal of specific problem animals is best achieved by the use of trained wildlife professionals, such as those employed by the United States Department of Agriculture's Wildlife Services, rather than hunters or fur trappers.

Summary

It often is in the best interest of society to maintain a wildlife population below the level of the environmental carrying capacity. Reasons may include the desire to 1) reduce the frequency of massive die-offs in that species, 2) produce the maximum sustained yield of animals for harvest, 3) maximize environmental benefits for other species, 4) reduce spread of disease or parasites, and 5) reduce wildlife damage to acceptable levels. In most cases, using hunters and trappers is the only cost-effective and efficacious method available to reduce wildlife populations over large areas. Hunting and trapping reduce wildlife damage by many different mechanisms, including 1) reducing a wildlife population below the environmental carrying capacity, 2) removing individuals from a population earlier in the year than would normally happen through natural causes of mortality, and 3) changing the behavior of animals so that they are less likely to cause damage. Hunting and trapping also can increase landowner tolerance of wildlife and wildlife damage. If hunting or trapping were to end, some wildlife populations would increase, animals would become more habituated to humans, wildlife damage would increase, landowner tolerance for wildlife would decrease, and some rural property values would fall. Because of these events, wildlife habitat would be lost because landowners would simultaneously lose a major incentive to maintain wildlife habitat and be confronted with greater levels of wildlife damage.

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Literature cited

ADKINS, R. J., AND L. R. IRBY. 1994. Private land hunting restriction and game damage complaints in Montana. Wildlife Society Bulletin 22: 520-523

ADKINS, R. J., AND L. R. IRBY. 1997. Game damage complaints, posting, and the importance of landowner cooperation: authors' response to Dunk and Kristan. Wildlife Society Bulletin 25: 195-196.

ALBERT, D. M., AND R. T. BOWYER. 1991. Factors related to grizzly bear-human interactions in Denali National Park. Wildlife Society Bulletin 19:339-349.

APPLEGATE, J. E. 1981. Landowner's behavior in dealing with wildlife values. Pages 64-72 *in* R. T. Dumke, G. V. Burger, J. R. Marsh, editors. Wildlife management of private lands. Wisconsin Chapter, The Wildlife Society, Department of Natural Resources, Madison, USA.

BAKER, R. O., AND R. M. TIMM. 1998. Management of conflicts between urban coyotes and humans in southern California. Proceedings of the Vertebrate Pest Conference 18:299-312.

BEHREND, D. F., G. F. MATTFIELD, W. C. TIERSON, AND J. E. WILEY III. 1970. Deer density control for comprehensive forest management. Journal of Forestry 68:695-700.

BHAT, M. G., R. G. HUFFAKER, AND S. M. LENHART. 1993. Controlling forest damage by dispersive beaver populations: centralized optimal management strategy. Ecological Applications 3: 518-530.

BOLLE, A. W., AND R. D. TABER. 1962. Economic aspects of wildlife abundance on private lands. Transactions of the North American Wildlife and Natural Resource Conference 62:255-267.

BOUTIN, S., AND D. E. BIRKENHOLZ. 1987. Muskrat and round-tailed muskrat. Pages 313-325 *in* M. Novak, J. A., Baker, M. E. Obbard, and B. Malloch, editors. Wild furbearer management and conservation in North America. Ontario Ministry of Natural Resources, Ottawa, Canada.

BOYLE, S. A., AND F. B. SAMSON. 1985. Effects of nonconsumptive recreation on wildlife: a review. Wildlife Society Bulletin 13: 110-116.

BRATTON, S. P. 1979. Impacts of white-tailed deer on the vegetation of Cades Cove, Great Smoky Mountain National Park. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 33:305-312.

BRODY, A. J., AND M. R. PELTON. 1989. Effects of roads on black bear movements in western North Carolina. Wildlife Society Bulletin 17:5-10.

BURGER, G. V., AND J. G. TEER. 1981. Economic and socioeconomic issues influencing wildlife management on private lands. Pages 252-278 *in* R. T. Dumke, G. V. Burger, and J. R. March, editors. Wildlife management of private lands. Wisconsin Chapter, The Wildlife Society, Madison, USA.

BURNHAM, K. P., AND D. R. ANDERSON. 1984. Tests of compensatory vs. additive hypotheses of mortality in mallards. Ecology 65: 105-112.

CAUGHLEY, G. 1970. Eruption of ungulate populations with emphasis on Himalayan thar in New Zealand. Ecology 51: 53-72.

CENTERS FOR DISEASE CONTROL AND PREVENTION. 1999. Summary of notifiable diseases, United States 1998. Morbidity and Mortality Weekly Report 47(53):1-93.

CHASKO, G. G., AND M. R. CONOVER. 1988. Urban-suburban Canada Geese: Too much of a good thing? Living Bird 7:8-13.

CLARK, W. R. 1987. Effects of harvest on annual survival of muskrats. Journal of Wildlife Management 51:265-272.

CLARK, W. R., J. J. HASBROUCK, J. M. KIENZLER, AND T. F. GLUECK. 1989. Vital statistics and harvest of an Iowa raccoon population. Journal of Wildlife Management 53:982-990.

CONOVER, M. R. 1981. Evaluation of behavioral techniques to reduce wildlife damage. Pages 332-344 *in* L. Nelson, Jr. and J. M. Peek, editors. Proceedings of the wildlife-livestock relationships symposium. University of Idaho, 20-22 April 1981, Moscow, USA.

CONOVER, M. R. 1989. Relationships between characteristics of nurseries and deer browsing. Wildlife Society Bulletin 17: 414-418.

CONOVER, M. R. 1994. Perceptions of grass-roots leaders of the agricultural community about wildlife damage on their farms and ranches. Wildlife Society Bulletin 22:94-100.

CONOVER, M. R. 1997*a*. Monetary and intangible valuation of deer in the United States. Wildlife Society Bulletin 25: 298-305.

CONOVER, M. R. 1997*b*. Wildlife management practices of metropolitan residents in the United States: practices, perceptions, costs, and values. Wildlife Society Bulletin 25:306-311.

CONOVER, M. R. 1998. Perceptions of American agricultural producers about wildlife on their farms and ranches. Wildlife Society Bulletin 26:597-604.

CONOVER, M. R. AND G. G. CHASKO. 1985. Nuisance goose problems in the Eastern United States. Wildlife Society Bulletin 13: 228-233.

CONOVER, M. R., AND D. J. DECKER. 1991. Wildlife damage to crops: perceptions of agricultural and wildlife professionals in 1957 and 1987. Wildlife Society Bulletin 19:46-52.

CONOVER, M. R., W. C. PITT, K. K. KESSLER, T. J. DUBOW, AND W. A. SANBORN. 1995. Review of human injuries, illnesses, and economic losses caused by wildlife in the United States. Wildlife Society Bulletin 23:407-414.

DOLBEER, R. A., J. L. BELANT, AND J. L. SILLINGS. 1993. Shooting gulls reduces strikes with aircraft at John F. Kennedy International Airport. Wildlife Society Bulletin 21:442-450.

ELLISOR, J. E., AND W. F. HARWELL. 1979. Ecology and management of javelina in south Texas. Texas Parks and Wildlife Department, Federal Aid Report Series 16, Austin, USA.

ERRINGTON, P. L. 1946. Predation and vertebrate populations. Quarterly Review of Biology 21:144-177, 221-245.

ERRINGTON, P. L. 1956. Factors limiting higher vertebrate populations. Science 124:304-307.

ERRINGTON, P. L., R. J. SIGLIN, AND R. C. CLARK. 1963. The decline of a muskrat population. Journal of Wildlife Management 27: 1-8.

FRITZELL, E. K., AND R. J. GREENWOOD. 1984. Mortality of raccoons in North Dakota. Prairie Naturalist 16:1-4.

FROST, H. C., G. L. STORM, M. J. BATCHELLER, AND M. J. LOVALLO. 1997. White-tailed deer management in Gettysburg National Military Park and Eisenhower National Historic Park. Wildlife Society Bulletin 25:462-469.

GEORGI, J. R. 1983. Parasites of wildlife transmissible to domestic animals and humans. Proceedings of the Eastern Wildlife Damage Control Conference 1:287-291.

GREEN, G. E., W. E. GRANT, AND E. DAVIS. 1985. Effects of hunting on javelina in south Texas. Wildlife Society Bulletin 13:149-153.

HAMLIN, K. L., S. J. RILEY, D. PYRAH, A. R. DOOD, AND R. J. MACKIE. 1984. Relationships among mule deer fawn survival, coyotes, and alternate prey species during summer. Journal of Range Management 48:489-499.

HEUSMANN, H W. 1999. Special hunting seasons and resident Canada goose populations. Wildlife Society Bulletin 27: 456-464.

HORNOCKER, M. G. 1970. An analysis of mountain lion predation upon mule deer and elk in the Idaho Primitive Area. Wildlife Monographs 21.

JENKINS, S. R., AND W. G. WINKLER. 1983. Mid-Atlantic states raccoon rabies outbreak. Proceedings of the Eastern Wildlife Damage Control Conference 1:293-297.

KNOWLTON, F. F., AND L. C. STODDART. 1992. Some observations from two coyote-prey studies. Pages 101-121 *in* A. H. Boer, editor. Ecology and management of the eastern coyote. Wildlife Research Unit, University of New Brunswick, Fredericton, Canada.

KREBS, J. W., M. L. WILSON, AND J. E. CHILDS. 1995. Rabies-epidemiology, prevention, and future research. Journal of Mammalogy 76:681-694.

KUFELD, R. C., D. C. BOWDEN, AND D. L. SCHRUPP. 1988. Influence of hunting on movements of female mule deer. Journal of Range Management 41:70-72.

LAVIGNE, G. R. 1992. Sex/age composition and physical condition of deer killed by coyotes during winter in Maine. Pages 141-159 *in* A. H. Boer, editor. Ecology and management of the eastern coyote. Wildlife Research Unit, University of New Brunswick, Fredericton, Canada.

LINDZEY, F. G., B. B. ACKERMAN, D. BARNHURST, AND T. P. HEMKER. 1988. Survival of mountain lions in southern Utah. Journal of Wildlife Management 52:664-667.

LINDZEY, F. G., W. D. VAN SICKLE, S. P. LAING, AND C. S. MECHAM. 1992. Cougar population response to manipulation in southern Utah. Wildlife Society Bulletin 20:224-227.

LINSCOMBE, R. G. 1996. U.S. fur harvest (1970-1995) and fur value (1974-1995) statistics by state and region. International Association of Fish and Wildlife Agencies, Washington, D.C., USA.

LOGAN, K. A., L. L. IRWIN, AND R. SKINNER. 1986. Characteristics of a hunted mountain lion population in Wyoming. Journal of Wildlife Management 50:648-654.

MACNAB, J. 1985. Carrying capacity and related slippery shibboleths. Wildlife Society Bulletin 13:403-410.

MANSFIELD, T. M., AND K. G. CHARLTON. 1998. Trends in mountain lion depredation and public safety incidents in California. Proceedings of the Vertebrate Pest Conference 18:118-125.

MCCABE, R. E., AND T. R. MCCABE. 1984. Of slings and arrows: an historical retrospection. Pages 19-72 *in* L. K. Halls, editor. White-tailed deer: ecology and management. Stackpole, Harrisburg, Pennsylvania, USA.

MCCULLOUGH, D. R. 1982. Behavior, bears, and humans. Wildlife Society Bulletin 10:27-33.

MCCULLOUGH, D. R. 1997. Irruptive behavior in ungulates. Pages 69-98 *in* W. J. McShea, H. B. Underwood, and J. H. Rappole, editors. The science of overabundance: deer ecology and population management. Smithsonian Institute, Washington, D.C., USA.

MCSHEA, W. J., C. WEMMER, AND M. STUWE. 1993. Conflict of interests: a public hunt at the National Zoo's Conservation and Research Center. Wildlife Society Bulletin 21:492-497.

MECH, L. D. 1984. Predators and predation. Pages 189-200 *in* L. K. Halls, editor. White-tailed deer: ecology and management. Stackpole, Harrisburg, Pennsylvania, USA.

MECH, L. D., S. H. FRITTS, G. L. RADDE, AND W. J. PAUL. 1988. Wolf distribution and road density in Minnesota. Wildlife Society Bulletin 16:85-87.

MITCHELL, J. M., G. J. PAGAC, AND G. R. PARKER. 1997. Informed consent: gaining support for removal of overabundant white-tailed deer on an Indiana state park. Wildlife Society Bulletin 25:447-450.

NEAL, T. J. 1977. A closed trapping season and subsequent muskrat harvests. Wildlife Society Bulletin 5:194-196.

NICHOLS, J. D., M. J. CONROY, D. R. ANDERSON, AND K. P. BURNHAM. 1984. Compensatory mortality in waterfowl populations: a review of the evidence and implications for research and management. Transactions of the North American Wildlife and Natural Resources Conference 49:535-554.

PALMER, D. T., D. A. ANDREWS, R. O. WINTERS, AND J. W. FRANCIS. 1980. Removal techniques to control an enclosed deer herd. Wildlife Society Bulletin 8:29-33.

PAYNE, N. F. 1980. Furbearer management and trapping. Wildlife Society Bulletin 8:345-348.

PECK, L. J., AND J. E. STAHL. 1997. Deer management techniques employed by the Columbus and Franklin County Park District, Ohio. Wildlife Society Bulletin 25:440-442.

PILCHER, B. K., AND G. E. WAMPLER. 1982. Hunting season movements of white-tailed deer on Fort Sill Military Reservation, Oklahoma. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies 35: 142-148.

ROOT, B. G., E. K. FRITZELL, AND N. F. GIESSMAN. 1988. Effects of intensive hunting on white-tailed deer movement. Wildlife Society Bulletin 16:145-151.

ROSS, P. I., AND M. G. JALKOTZY. 1992. Characteristics of a hunted population of cougars in southwestern Alberta. Journal of Wildlife Management 56:417-426.

RUSCH, D. H., S. R. CRAVEN, R. E. TROST, JR., J. R. CARY, R. L. DREISLEIN, J. W. ELLIS, AND J. WETZEL. 1985. Evaluation of efforts to redistribute Canada geese. Transactions of the North American Wildlife and Natural Resources Conference 50:506-524.

SANDERSON, G. C. 1987. Raccoons. Pages 486-499 *in* M. Novak, J. A. Baker, M. F. Obbard, and B. Malloch, editors. Wild furbearer management and conservation in North America. Ministry of Natural Resources, Ottawa, Ontario, Canada.

SCHMITKE, R. G. 1971. Some aspects of muskrat ecology at Big Island Lake, Alberta. Thesis, Utah State University, Logan, USA.

SIMPSON, M. R., AND S. BOUTIN. 1989. Muskrat, *Ondatra zibethicus*, population responses to harvest on the Old Crow Flats, Yukon Territory. Canadian Field-Naturalist 103:420-422.

SMITH, R. H., D. J. NEFF, AND N. G. WOOLSEY. 1986. Pronghorn response to coyote control—a benefit:cost analysis. Wildlife Society Bulletin 14:226-231.

SPARROWE, R. D., AND P. F. SPRINGER. 1970. Seasonal activity patterns of white-tailed deer in eastern South Dakota. Journal of Wildlife Management 34:420-431.

STOUT, G. G. 1982. Effects of coyote reduction on white-tailed deer productivity on Fort Sill, Oklahoma. Wildlife Society Bulletin 10:329-332.

SWENSON, J. E. 1982. Effects of hunting on habitat use by mule deer on mixed-grass prairie in Montana. Wildlife Society Bulletin 10:115-120.

TANNER, G., AND R. W. DIMMICK. 1983. An assessment of farmers' attitudes toward deer and deer damage in west Tennessee. Proceedings of the Eastern Wildlife Damage Control Conference 1:195-199.

THURBER, J. M., R. O. PETERSON, T. D. DRUMMER, AND S. A. THOMASMA. 1994. Gray wolf response to refuge boundaries and roads in Alaska. Wildlife Society Bulletin 22:61-68.

TODD, A. W. 1981. Ecological arguments for fur-trapping in boreal wilderness regions. Wildlife Society Bulletin 9:116-124.

TORRES, S. G., T. M. MANSFIELD, J. E. FOLEY, T. LUPO, AND A. BRINKHAUS. 1996. Mountain lion and human activities in California: testing speculations. Wildlife Society Bulletin 24:451-460.

TREFETHEN, J. B. 1975. An American crusade for wildlife. Winchester and Boone and Crockett Club, New York, New York, USA.

UNDERWOOD, H. B., AND W. F. PORTER. 1991. Values and science: white-tailed deer management in eastern national parks. Transactions of the North American Wildlife and Natural Resources Conference 56:67-85.

United States Department of Agriculture. 1991. Sheep and goat predator loss. National Agricultural Statistics Service, Washington, D.C., USA.

United States Department of Agriculture. 1992. Cattle and calves death loss. National Agricultural Statistics Service, Washington, D.C., USA.

Vecellio, G. M., R. H. Yahner, and G. L. Storm. 1994. Crop damage by deer at Gettysburg Park. Wildlife Society Bulletin 22: 89-93.

VerCauteren, K. C., and S. E. Hygnstrom. 1998. Effects of agricultural activities and hunting on home ranges of female white-tailed deer. Journal of Wildlife Management 62: 280-285.

Vosburgh, T. C., and L. R. Irby. 1998. Effects of recreational shooting on prairie dog colonies. Journal of Wildlife Management 62:363-372.

Wagner, F. H. 1988. Predator control and the sheep industry. Regina, Claremont, California, USA.

Warren, R. J. 1991. Ecological justification for controlling deer populations in eastern national parks. Transactions of the North American Wildlife and Natural Resources Conference 56:56-66.

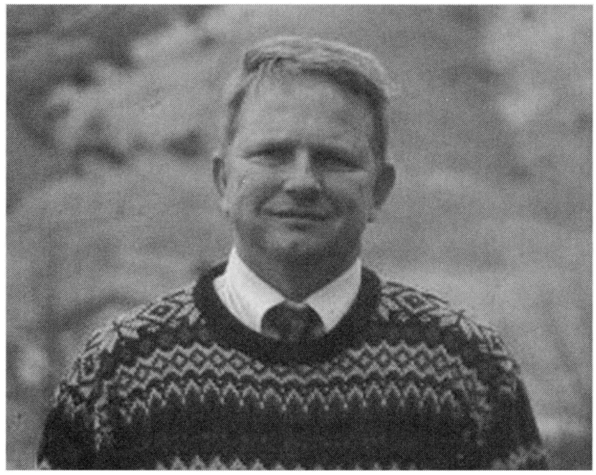
Wehausen, J. D. 1996. Effects of mountain lion predation on bighorn sheep in the Sierra Nevada and Granite Mountains of California. Wildlife Society Bulletin 24:471-479.

White, D., Jr., K. C. Kendall, and H. D. Picton. 1999. Potential energetic effects of mountain climbers on foraging grizzly bears. Wildlife Society Bulletin 27:146-151.

Williams, C. E., and M. E. Lathbury. 1996. Economic incentives for habitat conservation on private land: applications to the Inland Pacific Northwest. Wildlife Society Bulletin 24: 187-191.

Williams, O., and J. McKegg. 1987. Nuisance furbearer management programs in urban areas. Pages 156-163 *in* M. Novak, J. A. Baker, M. F. Obbard, and B. Malloch, editors. Wild furbearer management and conservation in North America. Ministry of Natural Resources, Ottawa, Ontario, Canada.

Wywialowski, A. P. 1996. Wildlife damage to field corn in 1993. Wildlife Society Bulletin 24:364-271.



Michael R. Conover is a professor in the Fisheries and Wildlife Department, Utah State University. He is also director of the Jack Berryman Institute, a national institute with a mission of resolving human–wildlife conflicts through teaching, research, and extension.

Associate editor: *Conner*