TERRA-3: Human Influences on Forest Wildlife Habitat

Kenneth L. Graham

US Fish and Wildlife Service

What are the likely effects of expanding human populations, urbanization and infrastructure development on wildlife and their habitats?

1 Key Findings

1.1 Impacts of Exotic Plants and Animals

- Exotic plants and animals have had a documented impact on forest wildlife and habitats. Exotic species threaten the survival of some sensitive wildlife species.

- Some forest wildlife species have benefited from exotic species, but indiscriminant use of exotic species for wildlife management purposes in the past has led to serious problems.

- Of the exotic species introduced into this country, only 4 to 19 percent have caused great harm. Another 6 to 53 percent have neutral or as yet undetermined effects.

- Approximately 42 percent of species that are listed in the United States as threatened or endangered under the Endangered Species Act are at risk because of competition with or predation by exotic species.

- More effective programs for preventing the introduction, establishment and spread of exotic species are needed. Protection and recovery of native species and ecosystems should be included as a goal in programs for control and management of exotic species.

1.2 Land-Use Changes in Forested Habitats

- Urban and agricultural land uses have interrupted the continuity of Southern forests, and created forest islands. Wildlife species differ in their response to the resulting fragmentation.

- Some wildlife species, particularly habitat specialists, have been harmed by loss and degradation of forest habitat, and population isolation caused by urbanization and agriculture.

- Other forest wildlife species have benefited from the creation of edge habitat and have adjusted to the new habitats created by man. Habitat generalists tend to adjust more easily to changes brought about by urbanization.

- Urbanization excludes some sensitive forest wildlife species but increases the presence of others. Urban habitats vary in their ability to support a diversity of forest wildlife. Advance planning and careful management can enhance the habitat value of urban and suburban conservation areas.

- For the most part, wildlife species that are tolerant of urbanization are not the rare or declining species that are of management concern.

- For species with area sensitivities, those that require forest interior, those that require specialized habitats, and those intolerant of human disturbance, special management considerations will be needed as urbanization increases in areas of the South.

- Prior to European settlement, early successional and disturbance dependent birds were found in naturally occurring and native American-maintained forest openings. Many of these disturbance-maintained ecosystems have been lost from the landscape during the last 300 years.

- The value of agricultural areas in providing habitat for early successional wildlife species (such as bobwhite) depends largely on how they are managed. "Clean farming," loss of pastures, creation of fescue-dominated pastures, and the use of heavy, fast-moving machinery have reduced the value of the habitat formerly found in pastures and agricultural fencerows.

- Agricultural crops provide foraging habitat for some forest wildlife, such as deer, black bears, raccoons, and many bird species.

- Woody fencerows enhance the habitat value of agricultural areas for some wildlife, and facilitate the movement of other forest wildlife species. However, woody fencerows in grassland habitats can reduce the habitat value to grassland-dependent birds due to increasing predator presence.

- Abandoned agricultural fields in the South have provided important old-field habitat for some early successional and disturbance-dependent wildlife species. This abandonment trend is diminishing in many areas of the Southeast, but forecast abandonment of agricultural lands in the Western portion of the region may provide at least a temporary benefit for early successional species.

- Successful conservation of some forest bird species will likely require forest management areas with thousands of acres of contiguous forest habitat. Similarly, many early successional and disturbance-dependent bird species are also area-sensitive, requiring hundreds of acres for successful conservation of some grassland bird species and dozens of acres for some scrub-shrub birds.

- The area-sensitivities documented for many forest bird species must be considered in a landscape context. Forest patch size is of greater concern in fragmented landscapes, such as the Ridge and Valley province of the Appalachians and the Mississippi Alluvial Plain, than in predominantly forested landscapes, such as heavily forested areas of the Southern Blue Ridge and Cumberland Plateau and the Ozark-Ouachita Highlands.

1.3 Linear Land Uses (Roads, Power Lines, and Trails)

- The effects of linear land uses (roads and utility rights-of-way) on forest birds should be considered in a landscape context. A continuum of effects has been documented, depending on the percent of the landscape forested, the road type and width, the maintenance needs, and other site-specific factors.

- Linear corridors such as roads and power lines can exclude sensitive forest wildlife from the adjoining habitat for distances ranging up to 330 feet or more. Effects on sensitive forest birds are of more concern in fragmented landscapes.

- In largely forested landscapes, roadsides and powerline corridors can provide important habitat for some grassland and early successional bird species with less concern required for the negative effects often attributed to fragmentation.

- Linear corridors act as barriers to the movement of some wildlife species, fragmenting populations. Examples include road effects on woodland mice, interstate highway effects on black bears, and power line effects on some neotropical migrants. Negative impacts documented for neotropical migrants as a result of fragmentation (such as reduced reproductive success in small forest patches), are of greater concern in heavily fragmented landscapes, however.

- Linear corridors act as travel lanes for other wildlife, such as grassland or scrub-shrub birds in largely forested landscapes, connecting isolated areas of habitat.

- Roadsides and power line corridors facilitate the spread of exotic plants and animals. Many exotics have been slower to gain a foothold in predominately forested landscapes.

- Road mortality has been well documented for many wildlife species, but the extent of the problem varies with a number of parameters including traffic speed and volume, road type, extent of cleared right-of-way, wildlife species present, and season. Road-related mortality is a serious problem for some rare species, such as the endangered Florida panther and the endangered Key deer.

- Sensitive forest plant species can be negatively impacted by human use of forest trails. “Collectable” wildlife may become rare along trails.

2 Introduction

2.1 Effects of Exotic Species on Forest Wildlife and Wildlife Habitat

Exotic non-native plants and animals were introduced into this country either intentionally or accidentally. In addition, many native species have been accidentally or intentionally introduced to other regions of the country, sometimes with negative consequences. The later group will not discussed in this Chapter. Since European colonization, thousands of plants and animals have been intentionally introduced into the United States. Many of these introductions have been beneficial to humans. Nonindigenous crops and livestock are the foundation of U.S. agriculture (U. S. Congress, Office of Technology Assessment 1993). Other exotic species are mainstays of horticulture and the pet and aquarium industries, or are used successfully for soil erosion control and biological control. Of the introduced species only a relatively few cause great harm. The U.S. Congress Office of Technology Assesment estimates 4 to 19 percent of exotic species fall into this category. Another 6 to 53 percent are estimated to have neutral or unknown effects. Many of our most invasive exotic species have been introduced into an environment in which they did not evolve and in some cases they have few or no natural enemies. Once established, they reproduce and spread unimpeded by (and often at the expense of) native plants and animals.

2.2 Human Land-Use Changes and Forest Wildlife

Following European settlement, historic trends in southern forest wildlife have closely followed habitat changes associated with land conversion and timber resource removal, coupled with uncontrolled exploitation of many species. For a more detailed history of southern forest wildlife see Chapter TERRA-1. Alterations in land use have changed the amounts of forest habitat available to forest wildlife species. They have fragmented forest stands and changed forest “edge” and forest “interior” habitats. Changes in the abundance, species richness, and species composition of forest wildlife have been documented in response to land use changes. This section describes the responses of forest wildlife to human land use changes.

See Chapter SOCIO-1 for a more detailed discussion of historic land-use changes and the History Background Paper. The initial conversion of forests and forest openings to farmland brought many changes in the numbers and kinds of wildlife (Bolen and Robinson 1995). Land conversions were not always negative for wildlife, however. Timber cutting for homesteads, cooperage, tanbark, heating, and land conversion (for fields and livestock) was initially beneficial to many wildlife species (Clark and Pelton 1999). Small farms carved from forests offered more edge habitat and supplemental food sources for many wildlife species. As forest timbering and land-use conversions increased, however, a combination of habitat loss and unrestricted wildlife exploitation decimated populations of black bears, white-tailed deer, and turkeys (Adams 1994, Clark and Pelton 1999).

Later, a trend toward abandonment of the small farms carved into woodlands began as the soils were depleted (Chapter SOCIO-1). As previously tilled lands reverted to shrubs and other vegetation, white-tailed deer, eastern cottontails, northern bobwhite, and some early successional bird species were highly favored (Clark and Pelton 1999, Hunter and others 2001). The conversion of agricultural land to some type of forest cover is expected to continue in some areas of the South as landowner returns from agriculture decline relative to those from forestry (Chapter SOCIO-1). Recent changes in farming practices have reduced the value of farms as habitat for some wildlife species.

Currently, strong economic growth has led to increased urbanization in parts of the South, (Chapter SOCIO-1). Urbanization fragments the natural landscape, destroys habitat required for many species, modifies habitat for others, and creates new habitat for some species (Adams 1994). This land-use shift will continue to influence the region's forests along with forest wildlife and habitat (Chapter SOCIO-1). Recent patterns of urban growth in the South have moved more people into the historically rural areas in low-density residential developments. In some areas of the South, forest cover remains relatively high, but the landscape is highly fragmented. Land-use changes that result in increased forest fragmentation could have negative impacts on a number of forest wildlife species, including many mature forest and early successional bird species.

2.3 Linear Land Uses (Roads, Power Lines, and Trails)

Along with urbanization, linear human land uses such as roads and power lines are increasingly prevalent in the South. The mortality of wildlife due to vehicle collisions and forest habitat loss are the most obvious impacts of roads on forest wildlife, but an increasing body of information suggests that the effects on wildlife populations are much more complex. About 3.85 million miles of public roads now exist in the United States (Forman 2000). Based on an assumption that some of the ecological effects of roads extend outward for more than 330 feet, Forman estimates that about one-fifth of the U.S. land area is directly affected ecologically by the system of public roads. Several compilations and review papers on the ecological effects of roads are available (National Resources Defense Council 2000, Trombulak and Frissell 2000, Forman and Deblinger 2000, and Findlay and Bourdages 2000).

Similarly, power line corridors function in a variety of ways to affect forest wildlife populations. Knight and Kawashima (1993) estimated that there were more than 0.31 million miles of powerlines in the United States, covering an estimated 5.2 million acres of land.

Trails also are linear features that bisect forest habitats and can affect sensitive forest plants and wildlife. Outdoor recreation activities are growing in popularity throughout the United States (Miller and others 1998), and recreational opportunities in the South are increasingly concentrated on the relatively small percentage of forested public land (Chapter SOCIO-6). More information about outdoor recreation in southern forests can be found in Chapter SOCIO-6.

3 Methods

To describe the documented effects of introduced exotic species, human land use changes, and infrastructure development on forest wildlife, information was incorporated from available scientific literature and the worldwide web.

4 Data Sources

Sources of information used for compiling this Chapter are cited in the text and details about these references can be found in the Literature Cited section.

5 Results

5.1 Effects of Exotic Species on Forest Wildlife and Wildlife Habitat

5.1.1 Exotic plant pathogens and forest wildlife

More than 20 species of exotic plant pathogens have been introduced into forests in the United States (Pimentel and others 1999) and exotic forest pests have greatly altered the species composition of forests in the East (Campbell 1997). Some tree species important as habitat for forest wildlife have been virtually eliminated throughout their ranges or greatly reduced in number. The loss of nuts and berries formerly produced by vanishing or severely reduced tree species has had a poorly documented but surely substantial impact on forest wildlife species (Campbell 1997). See Chapter HLTH-2 for a complete discussion of forest timber pathogens and diseases. Although the impacts of exotic plant pathogens to timber resources are well documented, the impacts on forest wildlife resources are not well described.

At the beginning of the 1900s, the American chestnut was one of the most important wildlife plants of the Eastern United States (Martin and others 1951). With this tree practically exterminated by the exotic chestnut blight, mast-dependent forest wildlife such as white-tailed deer and black bears had to settle for inconsistent acorn and hickory nut crops as their primary food (Clark and Pelton 1999). The blight almost certainly reduced the carrying capacity of southern highland habitats for mast-dependent wildlife. The blight is thought to have caused at least five indigenous insect species to become extinct or extremely rare (U.S. Congress, Office of Technology Assessment 1993). In areas where resprouting chestnuts remain in the understory, birds and mammals continue to transport virulent and hypovirulent-like strains of chestnut blight fungus (Scharf and DePalma 1981). Chinquapins in southern forests (including the Allegheny and Ozark chinquapins) vary in their susceptibility to chestnut blight. The chinquapins may not match the former value of the American chestnut in their habitat contribution to wildlife in southern forests (Martin and others 1951), but the nuts they produce are valuable to wildlife (USDA Forest Service 1999). Chestnut blight has affected chinquapins in southern forests and is expected to continue reducing the prevalence of susceptable tree species. However, no extermination of any southern wildlife species has been documented in conjunction with chinquapin losses.

Dutch elm disease devastated American elms as it spread across most of the country. In areas where Dutch elm disease removed the elm trees from the forest canopy, bird population surveys documented high local extirpation and colonization rates by bird species during the early 1950s (Whitcomb and others 1981). In Great Britain, reductions in bird abundance and diversity were documented in wooded farmlands accompanying elm death from Dutch elm disease and subsequent felling of dead trees (Osborne 1982, 1983, and 1985). The combination of Dutch elm disease and logging reduced the availability of suitable nesting cavities for cavity-nesting waterfowl species (Johnsen and others 1994).

Other exotic plant pathogens continue to affect wildlife habitat in southern forests by reducing the abundance of valuable forest tree species. These include dogwood anthracnose and butternut canker. Flowering dogwoods are valuable to many wildlife species for their fruit production (Martin and others 1951 and USDA Forest Service 1999). Butternuts are consumed by many species of forest wildlife.

5.1.2 Exotic plant invaders and forest wildlife:

Some troublesome weed pests (such as Johnson grass, multiflora rose, and kudzu) were intentionally introduced as crops, for wildlife enhancement or for erosion control but later became pests (Pimentel and others 1999). The majority of weeds, however, were accidentally introduced with crop seeds, from ship-ballast soil, or from various imported plant materials, such as ornamental plants. Some exotic invasive plants such as Chinese privet are shade tolerant and once established are capable of invading relatively dense forests. Many other invasives such as kudzu, mimosa tree, or princess tree are less adept at colonizing deeply shaded, mature forests except along edges, in natural or manmade forest canopy openings, or in disturbed or fragmented forests. Exotic plants have been spread by overgrazing, land-use changes, application of fertilizers and the use of agricultural chemicals (Westbrooks, 1998). Other human activities result in disturbed environments and encourage invasive plants. These activities include farming, creation of highway and utility rights-of-way, clearing land for homes and recreation areas such as golf courses, and constructing ponds, reservoirs, and lakes.

Millions of acres of forest land in the Southeast are occupied by exotic invasive plants. For many species, the acreage infested and spread rates are unknown. Kudzu and Japanese honeysuckle occupy more than 7 million acres each and their spread rates are increasing (Miller 1997). Clearcuts in the South can become infested with exotic vines such as Japanese honeysuckle and mile-a-minute, which can prevent the growth of seedlings and retard timber yields (Campbell 1997 and Nuzzo 1997). English ivy and Japanese honeysuckle can overgrow and eventually kill trees and under story plants and have fundamentally altered the character and structure of some forests (U.S. Congress, Office of Technology Assessment 1993). The herbaceous or shrub layers of large (but undetermined) areas of forest are being transformed into virtual monocultures by exotic vines, herbs, and shrubs (Campbell 1997). In some cases, these plant invasions have been shown to reduce forage or cover for wildlife. Table 1 lists some exotic plant species that are particularly noxious in forests in the Southern United States.

In recent years the impact of invasive exotics on biodiversity has become a major concern. Biological invasions by exotic species may displace native animals and plants, disrupt nutrient and fire cycles, and change the patterns of plant succession (Westbrooks 1998). Invasive exotic plants encroach into parks, preserves, wildlife refuges, and urban areas. Since many of these areas are significant for maintaining indigenous animals and plants (U. S. Congress, Office of Technology Assessment 1993), the responsible land management agencies are forced to expend increasing resources to control the most troublesome invaders. Approximately 61 percent of our

National Parks have at least a moderate level of exotic plant infestation: severely impacted parks include the Great Smoky Mountains. An estimated 400 of 1,500 vascular plant species in the Great Smoky Mountains National Park are exotic and 10 of these are currently displacing and threatening other species in the park (Pimentel and others 1999). Invasive exotic species are considered to be the second most important threat to biodiversity, after habitat loss and degradation. Approximately 42 percent, or about 400, of the 958 species that are listed in the United States as threatened or endangered under the Endangered Species Act are at risk because of competition with or predation by exotic species (Wilcove and others 1998). In south Florida, exotic plant species such as Australian pine, Brazilian pepper, and leatherleaf fern are invading disturbed areas and outcompeting native vegetation, reducing Key deer foods and habitat (U. S. Fish and Wildlife Service 1999). In spite of the severity of exotic plant invasion in southern forests, the impacts to forest wildlife in the South have only been sparsely documented. More information about the effects of exotic invasive plants on forest ecosystems can be found in Chapter TERRA-2.

5.1.2.1 Use of exotic plant species by insect herbivores

Many exotic invasive plant species lack insect herbivores adapted to live and feed on them. This factor likely contributes to their rapid spread. The number of plant-feeding insects associated with various trees is a reflection of the cumulative abundance of that tree throughout geological history (Southwood 1961). Recently introduced exotic tree species generally support relatively few insect species compared to abundant native tree species. The Chinese tallow tree is an invasive exotic that has spread rapidly across the Southern United States. Insects likely control the spread of this tree in its native China, and the lack of insect predation has aided its spread in the United States. Only one species, the leaf-footed bug, has been reported causing fruit damage to this exotic tree (Johnson and Allain 1998).

5.1.2.2 Use of exotic plant species by forest wildlife

Despite the tendency of some exotic plant invaders to form dense monocultures that exclude native flora and fauna, many species of southern wildlife use exotic plant species for forage and cover. Indeed, some invasive plant species in southern forests were introduced because they were considered beneficial for wildlife habitat (Miller 1997).

The value of Japanese honeysuckle both as cover and a food source for songbirds, gamebirds, hummingbirds, small mammals, and deer has been documented (Martin and others 1951, Hugo 1989, Miller 1997). Other exotic honeysuckles such as Amur honeysuckle also have been documented as food and cover for birds and small mammals (Whelan and Dilger 1992, Williams and others 1992, Martin and others 1951).

Multiflora rose is an invasive exotic shrub that was widely promoted by conservation agencies in the 1930s for cover, wildlife food, and as living fences (Miller 1997). It provides excellent habitat for gamebirds and songbirds (Martin and others 1951, Morgan and Gates 1982) and for cottontail rabbits (Morgan and Gates 1983).

Japanese and Chinese privets are invasive exotic shrubs that can replace native understory species and prevent forest regeneration in riparian forests and bottomland hardwood-pine forests (Miller 1997). Privets are used for food and habitat by birds and their seeds are widely dispersed by birds (Martin and others 1951, Miller 1997). Chinese privet also has been documented in northwestern Georgia as an important component of fall and winter diets of the white-tailed deer (Stromayer and others 1998).

Exotic shrubs in the buckthorn family provide excellent nesting and feeding habitat for many species of songbirds (Whelan and Dilgar 1992). The exotic shrub bicolor lespedeza provides food for songbirds, gamebirds, and hooved browsers, including white-tailed deer (Martin and others 1951, and Miller 1997).

The Chinese tallow tree in coastal South Carolina is used heavily by more than 14 bird species (Renne and others 2000). The Russian olive provides feeding habitat for songbirds, gamebirds, and hooved browsers (Martin and others 1951). Chinaberry is eaten to a limited extent by songbirds (Martin and others 1951).

Although these exotic invasive plant species provide habitat and food for southern wildlife species, no scientific investigations were found that compared the relative habitat value of these exotic invaders to the native flora that they displaced. In addition, no scientific investigations were found that documented the effects of exotic plant species invasions on a broad spectrum of southern forest wildlife species, including sensitive habitat specialists. The past introduction of exotic plants for wildlife management has unintentionally led to severe invasive exotic species problems. Many of the intended habitat benefits of these invasive species can be found in carefully selected native species. See the National Park website at http://nps.gov/plants/alien/fact.htm for some suggested native plant alternatives. Introduction of exotic plant species for wildlife enhancement should be approached with caution to avoid future invasive species problems.

5.1.3 The effects of exotic animals on forest wildlife

5.1.3.1 Exotic insect pests and forest wildlife

More than 2,000 arthropod species and 11 earthworm species have been introduced into the Continental United States, including approximately 500 exotic insect and mite species (Pimentel and others 1999). About 360 exotic insect species have become established in American forests and approximately 30 percent of these species have become serious pests. Although the negative effects of invertebrate pest species such as the gypsy moth and the balsam woolly adelgid to southern forests have been well documented (see Chapter HLTH-2), much less information is available about their effects on wildlife. See Chapter HLTH-2 for a description of the effects of insects and other forest pests on southern forests.

Balsam woolly adelgid

The balsam woolly adelgid is an aphid that inflicts severe damage in balsam-fir forests (Pimentel and others 1999). The balsam wooly adelgid has killed up to 95 percent of the Fraser firs in the Southern Appalachians.

Resultant habitat losses have impacted forest wildlife. A few species, such as the larvae of the moth *Semiothisa fraserata* may depend exclusively on the Fraser fir for food (Stein and Flack 1996). Other species such as the Weller's salamander are endemic to the spruce-Fraser fir habitat of the Southern Appalachians. Changes in the avifaunal composition of Fraser fir forests were documented in the Southern Appalachians following destruction of the Fraser fir canopy by the balsam wooly adelgid (Alsop and Laughlin 1991, Rabenold and others 1998).

Frazier fir bark provides substrate for eight rare species of mosses and liverworts (Stein and Flack 1996). The endangered spruce-fir moss spider lives in moss mats that are only found in the spruce-Fraser fir forests of Southern Appalachia (U.S. Fish and Wildlife Service 1998). Loss of the tree canopy (due to the balsam woolly adelgid) has resulted in increased light and temperature and decreased moisture on the forest floor, causing the moss mats on which the spider depends to dry up and become unsuitable.

The endangered Virginia northern flying squirrel and the endangered Carolina northern flying squirrel are found in conifer-hardwood ecotones or forest mosaics of spruce-fir associated with various hardwoods in high elevations of the Southern Appalachians (U.S. Fish and Wildlife Service 1990a). Although decimated by past logging of spruce forests, these two subspecies are currently threatened by several factors including habitat damage to conifer-hardwood ecotones by the balsam wooly adelgid and gypsy moth.

Gypsy moth

The gypsy moth was accidentally released in Medford, Massachusetts, in 1869. The spread rate of gypsy moths from 1966 through 1990 was approximately 13 miles per year (Liebhold and others 1995). Gypsy moths feed on numerous trees, shrubs, and vines, but prefer oaks (USDA Forest Service 1999).

Infestation by gypsy moths can impact forest wildlife habitat in several ways. Severe infestations can reduce the production of acorns and mast produced by susceptible tree species, reducing mast available for wildlife. However, resultant dead trees can serve as dens for some wildlife (Brooks and Hall 2000). Defoliation of the overstory can displace closed-canopy bird species while increasing the abundance of open-canopy species (Michigan State University, website for education program 1997). In some heavily overstocked forests lacking natural disturbances (such as fire), defoliation can benefit forest birds dependent upon smaller openings in mature hardwood or mixed forests. Beneficiaries include some declining or priority species such as Canada warblers and white-throated sparrows (Hunter and others 2001).

Following gypsy moth infestations, sensitive shade-dependent understory plants can become stressed by the increased sunlight reaching the forest floor (USDA Forest Service 1999). Defoliation of the overstory increases the growth of shrubs, grasses, and herbs providing some wildlife with additional cover and forage (Brooks and Hall 2000).

Red imported fire ants

The red imported fire ant infests more than 250 million acres in the United States (Allen and others 1994). Fire ants could spread across almost a quarter of the nation before range limits are reached. Southern States already infested by the species suffer damages totaling more than $1 billion per year (Pimentel and others 1999).

Red imported fire ants are most abundant in open habitats with disturbed soil, where sunlight can reach the soil surface (Stiles and Jones 1998). They are rare in shaded or undisturbed habitats such as intact forests. Fire ants can invade southern forests along the margins of linear disturbances such as roads or powerlines. In areas where the red imported fire ant is abundant, native ants are displaced by competition. Although omnivorous, the species feeds voraciously on living and dead insects. Native arthropod diversity and abundance often are reduced in heavily infested areas (Tedders and others 1990, Stiles and Jones 1998, Allen and others 1994).

Red imported fire ants have had detrimental impacts on many wildlife species (Allen and others 1994). Reptiles and amphibians tend to be vulnerable to displacement by fire ants when they compete for shared prey (invertebrates) or have an egg stage vulnerable to predation during times of high fire ant activity. Fire ants have been documented to destroy nests and cause hatchling mortality of the threatened gopher tortoise (U. S. Fish and Wildlife Service 1990b, Allen and others 1994).

Fire ants compete with native scavengers that feed on dead animals and fallen fruit. They have been implicated in declines of ground-nesting birds, such as quail and turkey, because they attack newly hatched young (USDA Forest Service 1999). Nest and chick predation by the red imported fire ant has been documented for many bird species (Allen and others 1994). The red imported fire ant has been linked to declines of migratory wintering populations of the loggerhead shrike (Grisham 1994). Injuries or death to white-tailed deer fawns and other new-born small mammals due to attack by the red imported fire ant have been widely reported (Allen and others 1994).

5.1.3.2 The effects of exotic wildlife on native forest wildlife

Stein and Flack (1996) estimate that at least 2,300 species of exotic animals now inhabit the United States. This total includes an estimated 20 species of exotic mammals, 97 species of exotic birds, and 53 species of exotic reptiles and amphibians. These species cost the U. S. economy about $27.5 billion every year (Pimentel and others 1999 and Scientific American 1999). Many of the larger exotic animals were deliberately imported for aesthetic, sport hunting, or livestock purposes. Deliberate imports include European starlings, European wild boars, ring-necked pheasants, and feral pigs. Other smaller exotic pests, such as rats, mice, red imported fire ants and balsam woolly adelgid arrived hidden in cargo holds, shipping containers, produce, and imported forest products. Echternacht and Harris (1993) indicated that at least 50 exotic wildlife species have become established in the Southeastern United States comprising about 8 percent of the 625 native and exotic wildlife species. Table 2 is based on their wildlife and faunal description. It contains a list of exotic wildlife species that are known to inhabit the Southeastern United States.

Feral pigs

Feral pigs that descended from domestic farm animals and European wild boars that were introduced for sport hunting now number about 4 million across the United States. Together, they cost the economy more than $800 million in damages per year (Pimentel and others 1999). Florida has about 0.5 million and Texas has 1 to 1.5 million.

The effects of wild pigs vary greatly from place to place, depending on the density of pigs and the sensitivity of the ecosystems involved (Singer 1981). Their rooting habit has damaged sensitive forest habitats across the South, including rare wetlands and springs in the Ozark-Ouachita Highlands (USDA Forest Service 1999). Wild pigs compete with wild turkeys and white-tailed deer for acorns and other foods. They tear up rotten logs that provide habitat for many amphibians and reptiles. In addition, hogs destroy the nests of turkeys, ruffed grouse and other ground nesting birds (Sealander and Heidt 1990, Miller and Leopold 1992). Wild pigs also carry diseases such as brucellosis and pseudorabies that represent a risk to native wildlife (Peine and Lancia 1990, New and others 1994, Tozzini 1982). No antibodies for serious diseases were detected in a 1990 survey of wild pigs in the Great Smoky Mountain National Park, however (New and others 1994).

Wild pigs occur in 13 National Parks but are especially problematic in the Great Smoky Mountains National Park (Singer 1981). Wild boars invade high-elevation northern hardwood communities from about April through August where their rooting has reduced understory plant cover up to 87 percent. Up to 77 percent of all logs and branches are moved in heavily rooted areas. Red-backed voles and shrews are normally common in pristine stands, but are absent in rooted areas.

Feral cats

Domestic cats, including both pets and free-ranging animals, now number about 100 million in the United States (Coleman and others 1997). The occurrence of cats tends to be concentrated around areas of human habitation. Studies of free-ranging domestic cats indicate that small mammals comprise about 70 percent of their prey, and birds constitute about 20 percent. Nationwide, free-ranging rural cats probably kill more than a billion small mammals and hundreds of millions of birds each year. Free-ranging cats are a serious threat to ground-nesting birds such as turkey and quail (USDA Forest Service 1999, Miller and Leopold 1992) and also attack shrub-nesting songbirds. In Florida, free-ranging cats are contributing to the imperiled status of several Federally listed species including the Lower Keys marsh rabbit, several types of beach mice, and woodrats.

Free-ranging cats can outnumber and compete with native predators, including hawks and weasels (Coleman and others 1997). Cat predation may deplete winter populations of microtine rodents and other prey of red-tailed hawks, marsh hawks and American kestrels (George 1974). Free-ranging cats also can potentially transmit new diseases to forest wildlife, including feline leukemia to cougars (Jessup and others 1993) and feline distemper and feline immunodeficiency virus to the endangered Florida panther (Roelke and others 1993).

Feral dogs

Free-ranging and feral domestic dogs are nearly ubiquitous across the United States (Drost and Fellers 2000): many problems are reported in Florida and Texas (Pimentel and others 1999). Free-roaming dogs chase and harass indigenous wildlife (U. S Congress Office of Technology Assessment 1993, Sealander and Heidt 1990) and disturb ground-nesting birds such as quail and wild turkeys by attacking adult birds, and consuming eggs and hatchlings (USDA Forest Service 1999, Miller and Leopold 1992). In southeast Alabama, free-ranging dogs prey upon the threatened gopher tortoise and destroy gopher tortoise burrows (U. S. Fish and Wildlife Service 1990b and Causey and Cude 1978). In south Florida, dog-related deaths are the second most frequent cause of man-induced mortality for the endangered Key deer (U. S. Fish and Wildlife Service 1999)

Free-ranging dogs have the ability to interbreed with coyotes and the Federally endangered red wolf (Sealander and Heidt 1990 and USDA Forest Service 1999).

European starlings

After the introduction of European starlings in the late 1800s, population growth and range expansion were explosive. Starling populations now appear to have leveled off or are decreasing in most areas across the country (Robbins 2001). Although starlings consume noxious insects and weed seeds, they also compete with native species for food and nesting cavities. Displacement of native birds by starlings has been documented in areas of the country with limited nest sites (Weitzel 1988). Starlings are known to be a very aggressive species when competing for or usurping cavities from other birds (James and Neal 1986).

Effects on reproduction and fecundity of red-bellied woodpeckers were documented due to nest cavity competition with starlings (Ingold 1994, 1996, Ingold and Densmore 1992). The effects of starling nest cavity competition on northern flickers and red-headed woodpeckers were found to be less severe. Competitive cavity losses for red-headed and northern flickers have more serious implications, however, since these two species are currently declining. Starlings are common in urban and agricultural woods, but are seldom found in densely forested areas (Ingold and Densmore 1992). Red-bellied woodpeckers that nest in more heavily wooded environments are more successful in avoiding competition with starlings. Starlings also compete with other native birds, including the eastern bluebird and purple martin for cavity nest sites (USDA Forest Service 1999).

House sparrows

Following a series of introductions in the United States, house sparrows became well established across the continent by 1910. Currently, populations appear to be stable or decreasing in most areas of the country (Robbins 2001). House sparrows are found mainly in urban and agricultural areas (James and Neal 1986) and are seldom found in predominantly forested areas.

Although they commonly nest in man-made structures, house sparrows also use deteriorating nests of other species, woodpecker cavities, and nesting boxes intended for other species. House sparrows have been documented to usurp cavities from red-bellied and red-headed woodpeckers (Ingold and Densmore 1992). In addition to native woodpeckers, house sparrows have been known to harass other native birds including robins, yellow-billed cuckoos, and black-billed cuckoos. They can displace native eastern bluebirds, wrens, purple martins, and cliff swallows from their nesting sites (Arcieri 1992, Pimentel and others 1999). The deaths of adult and nestling bluebirds were documented in South Carolina resulting from aggressive competition with house sparrows (Gowaty 1984).

5.2 Effects of Urbanization on Forest Wildlife

5.2.1 Effects of urbanization on forest birds.

5.2.1.1 Responses of forest bird communities to urbanization

A number of studies investigated changes to bird communities by comparing an urbanized site versus a less-urbanized (or more forested) site. Many investigators found that urbanization decreased the species diversity of the avian community and increased avian density (or bird biomass), favoring dominance by a few species. Bird species vary in sensitivity to urbanization, leading to loss of sensitive species and a shift in the species composition of urban versus forest bird communities. Habitat specialists, including many forest insectivores, neotropical migrants, and forest interior species, have been documented to be less tolerant of urbanization. Beissinger and Osborne (1982), Smith and Schaefer (1992), Franklin and Wilkinson (1996), Kluza and others (2000), Croonquist and Brooks (1993) and Dowd (1992) all documented shifts in avian species composition with increasing urbanization.

Some investigators studied the response of bird communities across several sites or along a gradient of increasing urbanization. Gradient studies revealed a less clear pattern in bird species diversity and density peaks; in some cases the pattern shifted seasonally. However, shifts in the avian species composition were generally found as urbanization increased (Blair 1996, Clergeau and others 1998, Lancaster and Rees 1979, and Rottenborn 1999).

Others investigated changes in the bird community at a single site through time as the area became urbanized or more forested. Butcher and others (1981), Askins and Philbrick (1987), Aldrich and Coffin (1980), Long and Long (1992) and Horn (1985) documented the loss of sensitive forest bird species after urbanization or their return after reforestation.

Table 3 lists selected forest bird species in the Southeastern United States and their tolerances to urban and suburban development.

5.2.1.2 Urban fragmentation and edge effects

Forest size and level of fragmentation and the effects on breeding birds

Increasing urbanization fragments forest habitat into smaller and more isolated tracts. Research on breeding forest birds has shown that some species have minimum area requirements. Many studies documented declines in the numbers of forest breeding migratory birds in small isolated forest patches (Danielson and others 1997). Breeding populations of migrant bird species have not declined, however, in large contiguous forest tracts in the Eastern United States (Wilcove 1988). Fragmentation is considered to be a primary contributing factor to observed neotropical migrant declines.

Whitcomb and others (1981) found that many neotropical migrant species became increasingly rare as the size of the forest decreased. In addition, "area sensitivities" varied depending on the degree of isolation from larger forest tracts. They concluded that forest tracts needed to contain hundreds or perhaps thousands of acres to conserve populations of some forest bird species. Robbins and others (1989) suggested that when managing forests for wildlife, top priority should go toward providing for the needs of area-sensitive or rare bird species. When conservation of large contiguous forest tracts is not possible, they suggested that several moderately sized contiguous forests could be helpful in maintaining rare forest breeding birds.

Reduced reproductive success of forest nesting birds in small or fragmented forests may be due to increased nest predation or nest parasitism by brown-headed cowbirds. Nest parasitism is associated with brown-headed cowbirds, which lay their eggs in the nests of other species. These "hosts" then raise cowbirds at the expense of their own offspring. Nest predation can be caused by a combination of many avian, mammalian, and reptile species. Rates of nest predation have been found to be higher in small forest tracts than in large forest tracts, and small urban forest tracts experience higher rates of predation than comparably sized forest tracts in isolated rural areas (Wilcove 1985). Migratory songbird populations suffer the most serious effects from increased predation in small forest tracts. Keyser and others (1998), Donovan and others (1995), Robinson (1992), and Robinson and others (1995) all documented reduced reproductive success of neotropical migrants and other forest nesting bird species in fragmented forests due to higher rates of nest predation and/or nest parasitism.

Recently, investigators stress the importance of overall forest cover or landscape levels of fragmentation surrounding a local area when evaluating the presence or nesting success of area-sensitive or forest-interior birds. As indicated by Villard (1998), preference for forest-interior habitat or avoidance of small fragments tends to focus attention on the local scale, whereas processes underlying these phenomena may take place over landscape or even continental scales. Therefore, forest-interior preference and area sensitivity should be considered in a landscape context. In one study, forest cover in approximately 40-square-mile study plots was found to be the most important factor affecting the distribution of forest birds (Trzcinski and others 1999). Comparatively, the independent measures of forest fragmentation produced effects that were inconsistent and far less important than overall forest cover. In addition, the reduction in nesting success of forest birds due to nest predation and parasitism was much greater in heavily fragmented landscapes with low forest cover than in heavily forested landscapes (Robinson and others 1995, Hartley and Hunter 1998). Similarly, no differences were detected in the breeding success of worm-eating warblers in small and large forest tracts when high amounts of forest canopy cover were present in the surrounding landscape (Gale and others 1997).

In addition, landscape-level factors may partially affect the distribution of mammalian nest predators and, potentially, songbird nest-predation rates. A combination of local features such as proximity to some types of edge, as well as broader landscape-level features, such as land-use patterns, was determined to influence the abundance of these mammals (Dijak and Thompson 2000). At a broader scale, raccoons were more abundant in agricultural landscapes with high densities of streams than in forested landscapes with low densities of streams. Opossums were more abundant in heterogeneous landscapes with widely spaced patches of forest and high densities of riparian habitat.

5.2.1.2.1 Connective corridors and offsetting the deleterious effects of fragmentation

The presence of connective corridors may help to reduce the isolation of wildlife populations in fragmented forests (MacClintock and others 1977, Machtans and others 1996, and Wegner and Merriam 1979). Corridors may provide a connection that allows wildlife to move from one patch to another across an intervening, inhospitable landscape. This phenomenon has been especially well documented for disturbance-dependent grassland and scrub-shrub bird species such as Bachman's sparrow in largely forested areas (Dunning and others 1995). It is not obvious that animals possessing the mobility of birds need corridors to cross-fragmented landscapes, but it appears that the open space between forest islands is a barrier to movement of some songbirds (Whitcomb and others 1981). Gaps of 250 feet or more produced isolation characteristics for some songbirds in small forest fragments created by power lines and roads (Robbins and others 1989). Such gaps may not represent as serious a problem in largely forested landscapes, however (Gale and others 1997). Some investigators question the conservation value of corridors or question whether sufficient experimental evidence exists to draw conclusions on their benefits (Simberloff and others 1992, Inglis and Underwood 1992). Several potential negative effects and disadvantages of corridors should be considered prior to their use in overcoming fragmentation (Simberloff and others 1992). Disagreement over the value of corridors to overcome the effects of fragmentation for various species is likely to continue for some time. The use of corridors and the effect of fragmentation on movement patterns seems to be highly species-specific (Debinski and Holt 2000).

5.2.1.2.2 Forest edges and their effect on breeding birds

Fragmented forests have a greater proportion of edge habitats. Edges have generally been regarded by wildlife managers to have a positive effect on wildlife because the number of species increases near habitat edges (Yahner 1988). This positive effect likely remains true in predominantly forested landscapes. In fragmented landscapes, however, maximizing species diversity is not always a desirable objective in light of the number of rare species that depend of large areas of habitat. Rates of nest predation and brood parasitism are greater at edges for some forest nesting birds (Gates and Gysel 1978), especially as overall forest cover becomes increasingly fragmented (Donovan and others 1997). Paton (1994) reviewed a number of studies that dealt with bird nesting success as a function of distance from an edge. Most studies found that nesting success decreased near edges as a result of increasing nest predation and parasitism rates. The strongest effects appeared to occur within about 125 feet of the edge. Indigo bunting nests along abrupt forest edges, such as agricultural edges, wildlife openings or campgrounds, had nearly twice the nest predation rate as those found along more gradual edges, such as those created by treefalls, streamsides, and gaps created by selective logging (Suarez and others 1997).

While the results of many investigations indicate that nesting success for forest birds is reduced by the proximity of edges, recent information indicates that such effects depend on the nature of the surrounding landscape. Hartley and Hunter (1998) reviewed various nest predation studies and concluded that nest predation rates decreased as the amount of overall forest cover increased. Edge effects were more apparent in largely deforested landscapes. Donovan and others (1997) found that nest predation rates were significantly higher near edges, but these increased rates were apparent only in highly and moderately fragmented landscapes, and not in unfragmented landscapes. The ovenbird may be an exception, however. Even in an extensively forested landscape, slightly reduced rates of breeding success were documented for ovenbirds near forest edges (King and others 1996). Still, ovenbird reproductive success remains high overall, and other sensitive neotropical migrants fare better in highly forested landscapes (Gale and others 1997). Ovenbirds reproduce well in mid-successional forests, and since such conditions are plentiful throughout eastern forests, the ovenbird is not considered a conservation priority species. See Chapter TERRA-1 for more information about the effects of forest fragmentation on forest wildlife.

Not all investigators agree that higher nest predation rates occur in smaller forests or along forest edges (Haskell 1995, Yahner 1996, Yahner and Mahan 1996, Matessi and Bogliani 1999, Friesen and others 1999). In addition, the magnitude and patterns of nest parasitism by brown-headed cowbirds is not consistent among studies (Robinson and others 1995, Robinson 1992, Gates and Gysel 1978, Evans and Gates 1997, Hahn and Hatfield 1995, Coker and Capen 1995, Donovan and others 1997).

5.2.1.3 Effects of urban environments on bird abundance and nesting success

In urban areas, forest breeding birds may have lower abundances and lower nesting success. A 10-acre woodlot without any nearby houses had greater species richness and higher abundances of neotropical migrant species than did a 60-acre urbanized woodlot, indicating that the diversity and abundance of neotropical migrant birds decreased with increased urban development (Friesen and others 1995). Golden cheeked warblers declined near urban development, apparently due to the increased presence of blue jays and greater nest predation (Engles and Sexton 1994). Declines of neotropical migrants were documented over a 50-year period in the North Carolina Highlands Plateau, likely due in part to the close proximity of residential development and urban fragmentation (Holt 2000). Nest predation rates were found to be greater for woodlands in the vicinity of human settlement (Matessi and Bogliani 1999). Mammalian nest predators were found to be more abundant in floodplain forests that adjoined residential and agricultural lands (Cubbedge and Nilon 1993).

5.2.1.4 Urban areas as habitat for birds

Urban woodlands are unsuitable habitat for many forest bird species, including many neotropical migrant birds, birds that require large habitat areas for breeding, birds that breed only in forest interior habitats, many scrub-shrub and grassland species, and those sensitive to urban disturbance. Urban and suburban preserves tend to be small and isolated from other forests. However, urban woodlands still provide habitat for some wildlife species and seasonally support migrating birds. Not all urban habitats are the same.

Woody vegetation volume is important in determining breeding bird diversity in urban settings (Goldstein and others 1986). Urban woodlots of 20 acres or more can support dense and diverse populations of breeding birds, provided that they have adequate shrub understory, mature and dead standing trees, and vegetative edge types of sufficient width and proper quality (Linehan and others 1967). Large urban parks, with well-preserved natural forest habitat, support bird populations more characteristic of native forests (Gavareski 1976). Urban parks, cemeteries, schoolyards and other open spaces are prime sites for wildlife management (Bolen and Robinson 1995). For example, Washington, D.C., has only house sparrows, pigeons (rock doves), and starlings in the downtown area, but nearby in the spring gardens surrounding the White House, 19 species are present.

In urban environments, the objective of wildlife management should be to maintain biological diversity by retaining sufficient habitat for the maximum number of wildlife species (Milligan and others 1995). Urban wildlife habitat designs must consider the size, composition, connectivity, dynamics of the habitat patches, and human perceptions of the habitat areas. At the same time, however, urban wildlife habitats must be at a scale compatible with the surrounding urban uses. Constraints are necessary to promote human health and safety, and to meet habitat requirements of the different wildlife species.

Urban habitats pose additional risks to resident avifauna. An estimated 98 million birds are killed each year in the United States from window collisions with high-rise buildings (Bolen and Robinson 1995). In addition, an estimated 2 to 4 million birds are killed each year in the Eastern United States due to collisions with communication towers (Weisensel 2000). The relative contributions of these mortality sources to the declines of any conservation priority bird species were not described in these references.

5.2.2 The effects of urbanization on birds of prey and scavengers

Birds of prey such as hawks, eagles, and owls can be vulnerable to the effects of urbanization because they are at the tops of food chains and their home ranges are larger than those of most other birds (Adams 1994). Hawk species differ in their requirements for nesting habitat and tolerance for forest openings and human disturbance. Cooper's hawks abandon nest sites when housing construction and residential disturbance encroach on established nest sites (Bosakowski and others 1993). There is evidence, however, of adaptability of various hawk species to urban settings. Broad-winged hawks are more tolerant of forest openings when selecting nest sites than red-shouldered, red-tailed, or Cooper's hawks (Titus and Mosher 1981). Red-shouldered hawks in New York and New Jersey have higher nest productivity with increasing distance from human habitation (Speiser and Bosakowski 1995).

Bald eagles generally select well forested areas near water bodies, and avoid areas of human development and areas of high boat and pedestrian traffic (Buehler and others 1991a and 1991b, Chandler and others 1995). On the lower Melton Hill Reservoir and the adjoining Clinch River in eastern Tennessee, residential and industrial development was found to be the primary factor limiting habitat suitability for eagle nesting (Buehler 1995).

When not searching for food, black and turkey vultures tend to prefer forested habitats free of buildings for roosting and nest sites (Coleman and Fraser 1989). Nests are frequently located away from human disturbance in rock crevices, in roadless, forested, and undeveloped areas. Nesting success for vultures was found to increase farther from buildings, due to lower disturbance and less depredation by dogs.

Although some raptors are sensitive to urban disturbance, there may be differences among individuals, species, and regions of the country. Raptors that are tolerant of urban environments include Mississippi kites, sharp-shinned hawks, Cooper's hawks, red-shouldered hawks, and red-tailed hawks (Adams 1994). Urban woodlands, even those composed primarily of exotic vegetation, lawns, and urban development, are acceptable to some red-shouldered hawks (Bloom and others 1993). One pair of red-shouldered hawks successfully fledged young within 65 feet of people engaged in jogging, picnics, and baseball games. American kestrels also have adapted to urban environments where suitable nesting cavities are available (Adams 1994).

The screech owl thrives in some suburban environments, especially those with large wooded lots (Gehlbach 1986). Burrowing owls, barn owls, and, occasionally, great horned owls have also been found in metropolitan environments (Adams 1994). Burrowing owls benefit from light levels of urban development and reach their highest densities in areas 55 to 65 percent developed. Other population-limiting factors are encountered beyond that development level, however.

5.2.3 The effects of urbanization on mammals

In general, urban environments support fewer species of mammals than surrounding rural areas (Adams 1994). The species that occur in urbanized environments tend to be habitat generalists rather than specialists. Urbanized areas can support high populations of exotic species, such as the house mouse and Norway rat. In less urbanized areas where large green spaces remain, more species are likely to be encountered. Downtown Boston cemeteries support 20 species of resident mammals (Bolen and Robinson 1995).

Small and medium-sized mammals, especially granivores, are the most abundant mammals found in urban and suburban environments (Adams 1994). In one study, mammals in urban greenspaces were primarily habitat generalists that utilize a mosaic of habitat types (VanDruff and Rowse 1986). Deer mice, meadow voles, tree squirrels, ground squirrels, chipmunks, and woodchucks are common residents of urban areas (Adams 1994). Some small mammals, however, are habitat specialists that do not easily adjust to changes brought about by urbanization. Fragmentation of habitat in the Great Dismal Swamp of Virginia and North Carolina by residential subdivisions and industrial parks may be contributing to the decline of five indigenous subspecies of mammals (Rose 1991). The Allegheny woodrat is restricted to only a few habitats and is listed as threatened in Pennsylvania because of statewide declines (Balcom and Yahner 1996). Increases in residential and agricultural development were observed near sites of extirpation. The few sites still occupied by the woodrat generally had less-fragmented surroundings (agricultural lands) than sites of extirpation.

Large herbivores do not easily find suitable habitat in highly urbanized settings (Adams 1994). Their large body sizes and correspondingly large home ranges exclude them from many urban environments. Nevertheless, many cities in North America have very high densities of white-tailed deer. Problems with damage to urban vegetation in sensitive areas such as flower gardens and parks coupled with high instances of deer-vehicle accidents have prompted some cities to initiate population control activities (Bolen and Robinson 1995).

Small insectivorous mammals such as shrews, moles, and bats are commonly encountered in most residential areas. Suburban residential areas often make excellent habitat for medium sized omnivores such as raccoons (Hoffmann and Gottschang 1997), opossums, armadillos, and skunks (Adams 1994).

Red foxes are the more tolerant of urban areas than gray foxes. They occasionally den in large wooded areas within some larger cities. Urban foxes are common in many British cities, even in the districts most densely populated by humans (MacDonald and Newdick 1982). In a Boston cemetery, resident red foxes hunt a burgeoning gray squirrel population (Bolen and Robinson 1995). Gray foxes are more wary of urbanized areas, but can be found in rural residential areas (Harrison 1997). The threshold for avoidance of residential areas by gray foxes is between 130 and 325 residences per square mile. Coyotes are becoming more common in urban and suburban settings (Adams 1994). Coyotes occur in suburban Seattle and Los Angeles, in residential areas north of New York City, and in Lincoln, Nebraska. In Lincoln, one coyote spent more than 70 percent of his time in a 35-acre residential subdivision (Bolen and Robinson 1995).

Large predators such as wolves, cougars, and bears are not part of urban mammal communities (Adams 1994). They have been eradicated from most rural areas as well. Black bear distribution in coastal North Carolina is negatively correlated with human density and positively correlated with percent of total forested land (Jones and others 1998).

5.2.4 The effects of urbanization on reptiles and amphibians

Some amphibians and reptiles have characteristics that make them vulnerable to the effects of urbanization (Adams 1994). They are less mobile than birds or mammals, and dispersal rates are slower. With habitat fragmentation, many amphibians and reptiles exist in localized distributions rather than one continuous population. Urbanization tends to exclude specialized reptiles and amphibians, while species with broad ecological tolerances and more general habitat needs tend to be more successful. Many reptiles and amphibians are eliminated when wetlands and aquatic habitats are lost due to drainage, channelization, or filling. Removal of ground cover and underbrush eliminates habitat for many salamanders and snakes (Adams 1994).

Amphibians are especially susceptible to local extirpations and constraints on recolonization due to the short distances traveled, site fidelity, and physiological constraints (Blaustein and others 1994). The effects of forest habitat loss during urbanization may be especially severe for forest-dwelling salamanders. Schlauch (1976) found that woodland salamanders such as the blue-spotted, spotted, marbled, and eastern tiger salamander were reduced in distribution in urbanized areas of Long Island. Loss of ponds, lowered water tables, urban pollution, reduced amounts of woodlands, and collections for pets were contributing factors. In addition, the northern two-lined salamander disappeared from most areas on Long Island due to destruction of suitable springs. This species needs cool and flowing spring water to breed. In western North Carolina, the abundance and diversity of salamanders were drastically reduced following clearcutting of the forest (Petranka and others 1993, Ash 1997). There is substantial debate about the recovery and long-term stability of salamander communities in managed forests (Petranka 1999, Ash 1999), but deforestation associated with urban development would be permanent, with little likelihood of recovery for many salamander species.

Recolonization of suitable areas can also be problematic for some reptiles, especially those that are habitat specialists. The Florida scrub lizard is a rare endemic and its largest remaining population is in Florida sand pine scrub on the Ocala National Forest (Tiebout and Anderson 1997). The lizard has limited vagility and can only occupy young seral stages of a regenerating forest (less than 7 to 9 years of age). Scrub lizards probably do not disperse through forests older than about 12 years of age. Fire suppression and the lack of forest successional dynamics have contributed to the rarity of this lizard.

The threatened gopher tortoise also is sensitive to urbanization. Egg and hatchling mortality can be quite high in urban areas (See Section 5.1.3.1. and Section 5.1.3.2.). This problem is compounded by low reproductive rates (Adams 1994). The gopher tortoise has been extirpated from urban areas in Mobile County, Alabama (Nelson and others 1992). Populations are more stable however, in areas with less severe habitat disturbance. Habitat modifications and land-use changes associated with urbanization and agricultural development have eliminated the timber rattlesnake from much of its historic range in east Texas (Rudolf and Burgdorf 1997).

Although urbanization excludes some sensitive forest reptiles and amphibians, urban environments may provide habitat for some species. The heavily urbanized western end of Long Island still supported 28 of the 37 species documented to historically exist on Long Island (Schlauch 1976). The less developed, eastern end supported 35 of the 37 species. Herpetofauna found to be urban tolerant by Schlauch (1976) included the red back salamander, Fowler's toad, the brown snake, the garter snake, and the eastern box turtle. Due to pet collection, box turtles disappeared quickly from areas near any ground-level nature trails, however.

5.2.5 Other general effects of urbanization on forest wildlife

Many habitats, such as the longleaf pine ecosystem or pine-oak woodlands of the Southern Appalachians, are dependent upon fire for maintenance. Fire suppression has affected the quality of wildlife habitats in some southern forests. In many forest areas, management now includes prescribed burning. However, the increasing presence of roads and residential areas has interfered with the use of prescribed fire. For more information on the effects of fire suppression and prescribed burning, see Chapter TERRA-4, and the Fire Background Paper.

For more information about the effects of air pollution on forest health, see Chapter HLTH-3. For more information about the effects of increasing demand for timber products on southern forests, see Chapter TIMBR-1.

5.3 Effects of Agricultural Land Use on Forest Wildlife

5.3.1 Effects of agricultural/urban interfaces on forest wildlife

For more information about the effects of fragmentation on forest birds, mammals, and herpetofauna, see the previous discussions in Section 5.2.1, Section 5.2.2, Section 5.2.3, Section 5.2.4 and Chapter TERRA-1.

5.3.1.1 Forest wildlife densities and movement along the forest/agricultural edge

Forest wildlife species differ in their responses to forest/agricultural edges. Some wildlife species are limited to forest interior habitats, and avoid edges. Other wildlife species are adapted to edges and forest openings, or may be attracted to special habitats created at forest/agricultural interfaces. Small mammal species exhibited differing responses at forest/field edges (or forest wildlife openings) (Manson and others 1999, Menzel and others 1999, Wegner and Merriam 1979). Increased numbers of mammalian nest predators were found along forest-field edges (Gates and Gysel 1978), higher densities of mammalian predators were found in floodplain forests adjoining residential and agricultural land (Cubbedge and Nilon 1993), and raccoons were found to be more abundant in forest edges adjacent to agricultural fields and streams (Dijak and Thompson 2000). In contrast, Heske (1995) found no differences in the abundance of furbearing and small mammals along forest/farm edges versus forest interiors in southern Illinois.

5.3.1.2 Nest predation of forest nesting birds adjacent to agricultural areas

For information about the effects of small forest fragments and forest edges on the success of forest-nesting birds see Section 5.2.1.2

Some avian species in forests near agricultural areas have reduced nest success rates. Rates of nest predation for songbirds were ubiquitously high in a study site bordering agricultural fields. Mammalian predators (especially raccoons) were abundant throughout the study site and present on all transects surveyed (Heske and others 1999). Similarly, higher predation rates for ground nests were documented in forests fragmented by agricultural land, due to more abundant avian predators (Huhta and others 1996). Increased numbers of nest predators (crows and blue jays) were noted during bird surveys in the Great Smoky Mountain National Park (Wilcove 1988). Apparently, agricultural and other land conversions outside the park boundaries caused an increase of these nest predators, even in this large, relatively contiguous forest area.

5.3.2 Agricultural areas as habitat for forest wildlife

5.3.2.1 Early successional species

Many bird species dependent on open habitats such as grasslands, prairies, savannas, glades and barrens are now in serious decline in the Eastern United States (Hunter and others 2001). Today, many of these early successional and disturbance-dependent species are found associated with active and abandoned farmland, pastures, and other man-made forest clearings. Prior to European settlement, these species were found in naturally occurring and Native-American-maintained forest openings. Many of these disturbance-maintained ecosystems have been lost from the landscape during the last 300 years. Some species dependent on them found suitable nesting habitat in man-made fields following loss of the natural openings. Populations of disturbance-dependent birds and other wildlife vary along with trends in agriculture. Conversions of pastures to more intensively cultivated row crops or intensively mowed, fescue dominated pastures, the maturing of abandoned farm fields in some areas of the South, and the trend to larger fields of cash crops with accompanying loss of fence-row habitat have all affected early successional species. Information from the 1997 National Resource Inventory indicates that the 13 Southern States lost about 2.2 million acres of pasture between 1992 and 1997, a net loss of about 3.4 percent (USDA Natural Resources Conservation, 2000). “These species are in trouble not only because of the intensification of farming and declining numbers of pastures, hay meadows, and abandoned fields, but also suppression of natural disturbances – fires, beaver activity, and floods - that generate natural grasslands and shrublands” (Askins 2001).

The introduction of exotic, cool-season pasture grasses was probably in response to overgrazing of native warm-season species and deteriorating range conditions (Twedt and others in press). Use of “improved” cultivars such as tall fescue, red fescue, Bermuda grass, weeping love grass, and many others began in the mid-1930s. Exotic grasses such as tall fescue can be grazed quite close to the ground and can be hayed during the mid-nesting season of many grassland bird species. Depending on their management, intensively grazed or frequently mowed fescue pastures offer little or no cover for wildlife and can be poor habitat for northern bobwhite (Barnes and others 1995) and other grassland species.

Eastern cottontail populations were found to remain highest in areas with relatively high amounts of pasture, stable woodlands, hayfields, and fields planted in small grains, such as wheat, oats and barley (Mankin and Warner 1999). The presence of pasture seemed to be the most important factor, however. In contrast, increases in row crops such as corn and soybeans were accompanied by declines in cottontails. Pasture environments apparently maintained cottontail abundance because they are closest to their preferred vegetation structure (old fields and early successional shrub lands). Similarly, landscape features such as percentage of woodland on farms, percentage of farmland in nonrow crops, percentage of land in soil-protecting crops, and percentage of land in conservation tillage were used to calculate habitat indices (Ribic and others 1998). These indices are important in determining areas likely to support high populations of northern bobwhites and cottontails. Indices indicating farming disturbance such as percentage of land under grazing and percentage of land on which fertilizers, pesticides, and herbicides were applied were associated with lower populations.

5.3.2.2 Importance of vegetated fencerows, hedgerows, and wooded corridors

The presence of woody fencerows in agricultural areas provides important habitat for many wildlife species (Bolen and Robinson 1995). In areas where agriculture constitutes a majority of the land use, fencerows with a continuous row of trees and shrubs can provide habitat for up to 36 species of birds per 6.2-mile segment, whereas fencerows without woody vegetation support 9 or fewer species over the same distance. Forest edges bordered by multiflora rose hedgerows had higher bird species diversity than open forest edges, but habitat generalists and forest-edge species provided most of the increased bird diversity (Morgan and Gates 1982). Forest edges with hedgerows had more cover in the first 6 feet above ground level than open forest edges, and retained more of this cover during the winter. In addition, cottontails were also more frequent in forest edges where hedgerows were present compared to open forest edges (Morgan and Gates 1983). Similarly, farmstead shelterbelts were documented to be valuable habitat for small mammals in agricultural areas (Yahner 1983).

Vegetated fencerows may be important for the movement of some wildlife species, allowing them to reach isolated forest patches across a matrix of open agricultural fields. Chipmunks and white-footed mice tend to move between wooded habitats down vegetated fencerows rather than crossing open fields (Wegner and Merriam 1979). Similarly, many forest-nesting bird species move from one wooded habitat to the next along vegetated fencerows rather than flying directly across open fields. Even when woodland birds such as eastern pewee, red-eyed vireo, and wood warblers foraged in open fields, they first moved from the woods down fencerows, then from fencerows into the open fields. MacClintock and others (1977) documented that a narrow, disturbed “corridor” of grazed woods and early second-growth forest could reduce the isolation of a forest patch, allowing it to maintain a high diversity of forest-nesting birds.

Fencerows in agricultural areas may have negative effects on some species, however. Nest survival for loggerhead shrikes in fencerows was documented to be lower than for those nesting in the adjoining pastures due to higher nest predation (Yosef 1994). Most of the potential nest predators observed during the study either flew or walked along fencelines, and appeared to avoid crossing open pastures. Similarly, area-sensitive grassland bird species avoided nesting in grassy pastures within the first 165 feet of wooded fencerows (O'Leary and Nyberg 2000). Sensitive grassland nesters included two conservation priority species -- Henslow's sparrow and bobolink.

5.3.2.3 Foraging habitat for forest wildlife

Agricultural areas including grain fields, pastures, fruit orchards, gardens, and vineyards are important forage areas for many wildlife species (Martin and others 1951). Not all forage use of agricultural land by wildlife results in damage to crops. Foraging by insectivorous birds and mammals and consumption of weed seeds by wildlife is beneficial to agriculture. Wildlife often consume waste grain left behind by mechanical harvesting machines or consume fruit that has fallen on the ground. In other cases, however, loss and damage to crops by wildlife have been clearly documented. Martin and others (1951) documented the value of several agricultural commodities for wildlife. Corn is consumed by over 100 species of wildlife, including 17 species of upland gamebirds, 59 species of songbirds, 10 species of fir and game mammals, 6 species of small mammals, and 3 species of hoofed browsers. Wheat is consumed by more than 94 species of wildlife, and oats are consumed by at least 91 different species. Rice and apples are other important agricultural commodities eaten by foraging wildlife in the South.

Fallow fields were the most common habitat selected by bobwhite, even though crop fields, wildlife management plots planted annually in small grain and woods managed by prescribed burning were available nearby (Yates and others 1995). Apparently, insects were the most important food resource for feeding bobwhite hatchlings. Insect sampling revealed that fallow fields had more insects than other available habitats.

Black bears in the Southeast feed more in agricultural areas than in other parts of the United States, but their use of these areas may increase their vulnerability to hunting, lowering the overall rates of survival especially for males (Hellgren and Vaughn 1994). In coastal North Carolina, corn crop damage by black bears amounted to about 0.6 percent of the total area surveyed (Maddrey and Pelton 1995). Most of the damage was within 165 feet of the forest edge. In questionnaires completed by coastal North Carolina farmers, deer were the major cause of crop depredation (Maddrey and Pelton 1995). Crop damage by black bears, birds, and raccoons was reported less frequently.

Raccoons frequently use agricultural areas for foraging. One study found that raccoons in an agricultural area foraged mainly on corn which accounted for up to 76.2 percent of their diet (Sonenshine and Winslow 1972). Coyotes were found to be well adapted to agricultural areas in Vermont (Person and Hirth 1991). They preferred hardwood forests in the winter and spring, and farmland during the summer and fall.

Great horned owls are habitat generalists that prefer open cropland and pastures for foraging (Morrell and Yahner 1994). Barn owls also prefer to forage in pastures and grass-dominated agricultural areas (Bolen and Robinson 1995).

Wintering flocks of grackels, red-winged blackbirds, starlings, and brown-headed cowbirds use fields and feedlots for foraging. One such wintering flock removed 1,300 to 7,000 tons of corn each winter from a total foraging range of about 541,000 acres (White and others 1985). In a control measure, over 1 million birds were killed with the surfactant PA-14 one winter. Recruitment of birds from surrounding areas caused the roost to return to prekill levels within about 2 weeks. Roost fidelity for such wintering flocks averages only 3.5 to 4.4 nights per individual. Thus, the daily population turnover rate for the roost is about 23 percent.

5.3.2.4 Hazards of agriculture to wildlife

Although agricultural areas are habitat for many wildlife species, they can also subject them to hazards not encountered in natural areas. Mowing equipment and nighttime mowing has increased the mortality of eastern cottontails, bobwhite, and other wildlife attracted to pastures and hayfields (Bolen and Robinson 1995).

Many wildlife species forage in agricultural fields, but crop losses have resulted in lethal and nonlethal depredation control measures (Bolen and Robinson 1995). Under some conditions, certain crops may be harmful to wildlife. Geese that consume dry soybeans may harm or obstruct their esophagi as the swelling soybeans cause hemorrhaging and necrosis, or prevent the passage of food to the stomach. Aspergillosis is a fungal infection of the respiratory tract, contracted by birds exposed to molding crops. Once contracted, the infection can be spread to other birds, causing sizable die-offs.

Wildlife living and foraging in agricultural areas are exposed to insecticides, herbicides and fertilizers (Bolen and Robinson 1995). Many insecticides are not species-specific, and can be lethal to wildlife through direct exposure or through ingestion of contaminated prey species. Some of the more toxic pesticides, including the chlorinated hydrocarbons DDT, Aldrin, and others, are now banned in the United States, but because of long residual times and heavy pesticide buildups it has taken some time for their deleterious effects to fade. Most herbicides approved for use today are not directly toxic to forest wildlife if applied correctly. Indiscriminant use can indirectly harm wildlife, however, by reducing important vegetation for cover and forage. Fertilizers in granular form can resemble seeds or grit and offer a potential hazard to birds that might ingest a large number of granules.

5.3.3 Old field successional areas

Some areas of the South are likely to experience a reduction in agricultural land uses with a subsequent return to forest habitat (see Chapter SOCIO-1 for more information). Many of these increases in forest acres will undoubtedly be in the form of pine plantations rather than natural forest types, however (see Chapter TIMBR-1). See Chapter TERRA-4 for a discussion of the influence of pine plantations on forest wildlife and habitats.

Abandoned agricultural land undergoes a series of vegetation changes that provide important habitat for a number of wildlife species. The return to old-field habitat benefits many disturbance-dependent bird species. Successful management for many of these rare and declining birds will require adequate space for area-sensitive species, connecting corridors between early successional habitat areas, and availability of areas in specific vegetation stages to offset natural plant succession (Hunter and others 2001). Breeding bird density and species composition shift as abandoned farm fields undergo natural vegetative succession to mature forests (Johnston and Odum 1956). A few species, such as the cardinal, persist through many plant successive stages, but most birds appear to have a definite range of vegetative stages. Browsing mammals such as deer also benefit as abandoned agricultural areas undergo the vegetative transition into scrub-shrub habitats (Adams 1994).

Old-field habitats can vary in vegetative structure. The presence of exotic vegetation in agricultural environments is an influence that persists long after fields are abandoned. Previous type of agricultural use can influence the vegetative structure, and hence the wildlife habitat, in a particular abandoned field. Abandoned pastures differed markedly in their vegetation compared to previously cultivated old fields (Stover and Marks 1998). Exotic herbaceous plants in an old-field environment reached their peak abundance within 65 feet of the forest edge (Meiners and Pickett 1999).

Restored bottomland hardwood forests failed to regain their wildlife habitat value relative to mature forests even 50 years after agricultural usage (Shear and others 1996). Although the regenerating forests had similar structural attributes to the uncut forests, the lack of heavy seeded, mast-producing tree species (oaks and hickories) made them generally less useful for mast-dependent forest wildlife. Conversely, bottomland hardwood reforestation efforts that rely solely on oak planting are slow to produce a substantial 3-dimensional forest that provides useful habitat for nongame species, including many neotropical migrants (Twedt and Portwood 1997). More naturally invading species became established in bottomland hardwood restoration areas sown with acorns than in areas planted with oak seedlings (Twedt and Wilson in press).

5.3.4 Other general effects of agriculture on forest wildlife

Agricultural land uses have resulted in fire suppression and interruption of presettlement forest fire patterns. Lack of fire in most forest habitats has greatly affected the quality of wildlife habitat. For more information on the effects of fire suppression and prescribed burning, see Chapter TERRA-4.

Agricultural disturbance permitted introduction of a great many exotic plant and animal species. See Section 5.1 of this Chapter for information about the impacts of exotic plant and animal species on forest wildlife.

5.4 Effects of Linear Land Uses (Roads, Power Lines, and Trails) on

Forest Wildlife

5.4.1 Habitat displacement of wildlife by roads and powerlines

Some forest wildlife are excluded from or are less numerous in areas adjacent to roads and highways. Woodland breeding birds and terrestrial birds were found to have reduced densities adjacent to highways (Reijnen and others 1995, Kuitunen and others 1998). Some species clearly avoided the road, while others appeared to favor road-forest edges. Birds responding to corridor/forest edges along a powerline corridor could be divided into edge, deep forest, and unaffected species (Kroodsma 1982).

Road and powerline corridors may vary in their effects on forest wildlife, depending on corridor width. Forest-interior, neotropical migrant birds exhibited diminished abundances along wide power line corridors (50 to 75 feet) but not along narrow forest openings (of 25 feet) along unpaved dirt roads (Rich and others 1994). Such edge effects may not be as important for birds nesting in predominantly forested landscapes. In a landscape more than 70 percent forested, worm-eating warblers in small forest patches, separated by paved two-lane roads and house lots, were found to have nesting success comparable to those nesting in large forest tracts (Gale and others 1997). However, even in heavily forested landscapes, ovenbirds showed reduced densities of breeding territories and reduced pairing success within 500 feet of forest roads (Ortega and Capen 1999). Therefore, while edges of narrow corridors may be acceptable habitat for some bird species, they may be unsuitable for others. These issues must be evaluated in terms of the conservation concerns for the species at issue in a given situation (see Chapter TERRA-4 and Section 5.2.1.2 of this Chapter for discussions concerning ovenbird response to edges vs. conservation status).

Forest roads reduced the abundance and species richness of macroinvertebrate soil fauna (Haskell 2000). This effect extended up to 330 feet into the forest. Although wider roads and those with a more open canopy produced steeper declines, even narrow roads through forests produced marked edge effects.

5.4.2 Early successional and forest edge habitat

Some wildlife are attracted to roadsides and powerline rights-of-way because of grassland, early-successional or edge habitat. The value of roadsides and utility corridors has been documented for grassland and habitat generalist species of small mammals (Adams and Geis 1983, Johnson and others 1979).

Corridor width and vegetative characteristics influence the attractiveness of the habitat for bird species. Road rights-of-way are important habitat for birds that nest in edges and ecotones (Warner 1992). The number of roadside nests and species increased with roadside width. Mowing schedules, diversity of vegetation, and vegetative structural complexity affected the habitat value of roadsides for nesting birds. Narrow powerline corridors (40 feet wide) had a reduced diversity of birds compared to wider corridors (100 feet or more) (Anderson and others 1977). Wide corridors attracted more grassland bird species. Powerline corridors with increased patchiness of shrub vegetation, showed increased fledging success of nesting birds (Chasco and Gates 1992). Fledging success decreased, however, as the habitat became more homogeneous. Many early successional and disturbance-dependent bird species can be found in roadsides and utility rights-of-way (Hunter and others 2001, Meehan and Hass 1997), but corridors lacking shrub growth may have fewer nesting and wintering birds (Meehan and Hass 1997). Corridor nesting birds were more dense in the corridor interiors than along the edge (Kroodsma 1987).

5.4.3 Linear corridors as dispersal barriers for wildlife

Small forest mammals, such as eastern chipmunks, gray squirrels, and white-footed mice,) were found reluctant to venture onto road surfaces when the distance between cleared road margins exceeded 65 feet (Oxley and others 1974). Four-lane highways acted as effective barriers against the movements of these small forest mammals. Medium-sized mammals, such as woodchucks, porcupines, raccoons, and striped skunks, crossed wider cleared road margins more often, but suffered higher road mortality than small mammals. Similarly, the movements of white-footed mice across roads, including narrow gravel roads, were found to be infrequent (Merriam and others 1989), and paved roads were found to be a significant barrier to the movements of woodland mice (Mader 1984). Even small forest roads not open to public traffic were seldom crossed.

The presence of roads appeared to substantially hinder the movements of forest amphibians (Gibbs 1998). In a different study, primary and secondary roads did not affect the presence and movement of forest frogs and toads (DeMaynadier and Hunter 2000). The movement of forest salamanders was significantly inhibited by primary forest roads, but the minor forest roads had little effect.

Black bears in the Pisgah National Forest of North Carolina almost never crossed an interstate highway; roads with low traffic volume were crossed more frequently than those with high traffic volume (Brody and Pelton 1989). Bears also appeared to adjust their home ranges to areas with lower road densities.

The nature of the corridor edge may determine how strongly that edge serves as a boundary for wildlife. Abrupt vegetative transition from forest to mowed grass on the edge of a power line corridor was found to be a barrier to forest birds, and served as a natural territorial boundary for many bird species (Chasco and Gates 1992). When the vegetative contrast of the corridor was softened by shrubby vegetation, however, there was greater overlap between mixed-habitat and forest bird species. Power line corridors with abrupt edges were also avoided by small and medium-sized mammals because of difficulties in crossing the dense grass mats (Gates 1991). Corridors with a wide shrub zone along the edge had increased use and permeability to movement.

Wildlife underpasses can be an effective way to relieve the barrier effect of roads for some wildlife species (Clevenger and Waltho 2000). Wildlife differ in their abilities to utilize underpasses. In south Florida, white-tailed deer, raccoons, bobcats, the endangered Florida panther, alligators, and black bears were all documented to use underpasses to traverse an interstate highway (Foster and Humphrey 1995). Considerations for topography, habitat quality, location, and the level of human activity in the vicinity are important in designing a successful wildlife underpass (Clevenger and Waltho 2000).

5.4.4 Linear corridors as dispersal routes for wildlife

Road rights-of-way also can facilitate the movement of wildlife. Some grassland and early-successional species such as Bachman's sparrow require grassy and shrub-dominated corridors to facilitate their movement to and from isolated patches of suitable habitat (Dunning and others 1995). Meadow voles greatly expanded their range in central Illinois after the establishment of continuous strips of dense, grassy vegetation along interstate highways (Getz and others 1978). In contrast, the prairie vole is not restricted in movement by interruptions in grassy habitats. This species remains dominant in grassy sites not connected to the interstate (such as pastures and county roadsides). Similarly, a shrubby powerline corridor and edges served as a travel lanes for red foxes and striped skunks in a fragmented landscape (Gates 1991), but mammalian nest predator abundance was found to be influenced by both local and landscape-level features (Dijak and Thompson 2000).

Black bears use roads in the Great Dismal Swamp National Wildlife Refuge as travel corridors through the dense pocosin vegetation (Hellgren and others 1991). Such road use by bears is more characteristic among "unharvested" or protected populations. Hunted bear populations generally avoid roads, especially those with unrestricted use by humans.

Wooded roadside corridors serve as travel lanes for native forest mammals, but use of corridors taper off with distance from the forest (Downes and others 1997a and 1997b). Wooded road corridors appear to be used heavily, by non-native house mice and black rats, reducing their value as a remedy for habitat fragmentation. Males of some mammal species may utilize corridor habitats in greater numbers than females, indicating that roadside forest corridors may function as intraspecific filters.

5.4.5 Road mortalities and forest wildlife

Mortality along roads and highways has been well documented for many species of wildlife, but a number of factors influence the severity, including season, weather events, type of road, location of road, and road density. During a 14-month period along a dual lane highway, road mortalities were documented for 11 species of mammals, 12 species of birds, 5 species of reptiles, 9 species of amphibians, and insects belonging to 11 orders (and more than 249 different species) (Seibert and Conover 1991). Amphibian mortalities were higher in certain seasons and after rains. Populations of timber rattlesnakes were reduced in areas of eastern Texas having high road densities (Rudolph and Burgdorf 1997). Road-related mortality was a significant threat to raptors, especially northern saw-whet owls and eastern screech owls (Loos and Kerlinger 1993), but road kill numbers varied with season, location, road type, and species involved.

Mortality rates of small forest mammals such as Eastern chipmunks, gray squirrels, and white-footed mice were highest when cleared road margins were about 45 to 115 feet (Oxley and others 1974). Mortality rates for these small mammals dropped as cleared margins grew wider, mainly because they seldom attempted crossings of wider forest clearings. Mortality of medium sized mammals such as woodchucks, porcupines, raccoons and striped skunks increased with increased cleared width, reaching a peak when traffic density was high and young were emerging. Small mammal road mortalities on interstate highways was found to be greatest for species with highest densities in the right-of-way habitat, but the loss did not appear to be detrimental to populations of these species (Adams and Geis 1983). Road mortalities for white-tailed deer along interstate highways have been documented by Reilly and Green (1974) and Puglisi and others (1974). Road mortality of vertebrates were recorded in north Florida (Cristoffer 1991). Mortality increased with increasing speed limits and increasing density of roadside vegetative cover.

Population impacts of road-induced mortality can be significant for some wildlife species. In south Florida, road kills are the largest source of man-induced mortality for the endangered Florida panther and the endangered Key deer (U. S. Fish and Wildlife Service 1999).

5.4.6 Spread of Exotic Plants and Animals

Roads and power line corridors provide habitat and mechanisms for the spread of some exotic plants and animals. All high- and low-use roads sampled in an experimental forest contained at least one exotic plant species, some had as many as 14 (Parendes and Jones 2000). Even abandoned spur roads with no traffic over the last 20 to 40 years still had numerous exotic plants. Narrow, linear forest openings associated with roads and power lines appear conducive to establishment of the red imported fire ant (Stiles and Jones 1998). See the review in Trombulak and Frissell (2000) and the information compiled by the National Resources Defense Council (2000) for more information about the spread of exotic plants and animals along roads.

5.4.7 Other effects to wildlife from roads and power lines

Roads can provide hunters and poachers with increased access into forested areas (Natural Resources Defense Council 2000). Many large mammals are exposed to increased hunting pressure near roads, and some may have difficulties maintaining their populations near roadsides. In the Appalachian Highlands, management of black bears requires a special concern for road density (Clark and Pelton 1999). While overall black bear populations in the Southern Appalachians are considered stable to increasing at the present time, most black bear mortality is human-induced and includes hunting, poaching, and road kills. Hunting and poaching efficiencies increase along with improved vehicle access, and black bear habitat suitability is increased when the density of roads is kept low, or if logging roads are closed after the timber has been harvested (Clark and Pelton 1999). Similarly, Brody and Pelton (1989) concluded that the primary effect of roads in bear habitat in western North Carolina was an increase in the vulnerability of bears to hunting.

Roads can subject wildlife to increased levels of heavy metals, salts, and organic compounds through accumulation in plants, soil, and water (see the review in Trombulak and Frissell 2000). Corridor maintenance by mowing presents a hazard for some ground-nesting birds and other wildlife species (Bolen and Robinson 1995).

For a discussion of indirect effects of roads, including promotion of further human land use changes, see the review in Trombulak and Frissell (2000).

5.4.8 Effects of trails on forest wildlife.

The effects of trails appear to be better documented for plants than other taxa. Trampling by hikers and other forest recreational users has been implicated in the decline of sensitive forest understory plants (Gross and others 1998).

Research from regions outside of the South has documented shifts in forest bird composition along trails (Miller and others 1998, Hickman 1990, and Van der Zande and others 1984). Such effects may depend on the intensity and timing of the recreational disturbance, however (Van der Zand and others 1984).

In other more general studies, research indicates that human intrusion can alter bird behavior and community structure. Disturbance by pedestrians and vehicles was found to reduce the number of bird species on wooded streets, as well as species persistence, guild density, and probability of occupation by individual bird species (Fernadez-Juricic 2000). Crows were found to be more vigilant in areas of high human disturbance than in areas of low human disturbance (Ward and Low 1997). Since vigilance and foraging are mutually exclusive behaviors, the level of human activity can affect the foraging success of sensitive bird species. Others have found, however, that low levels of human intrusion (one person for 1 or 2 hours per week) did not significantly affect the vertical distributions of any forest bird species in three vegetation strata above the ground (Gutzwiller and others 1998). The forest bird species studied were apparently able to tolerate low levels of human intrusion.

Black bears also are sensitive to human disturbance and may be affected by the presence of trails. Hibernating black bears were found to readily abandon their dens and cubs in response to investigator disturbance (Goodrich and Berger 1994).

As observed by Schlauch (1976), some "collectable" wildlife such as box turtles or salamanders disappear quickly in the vicinity of ground-level nature trails, due to pet collection.

Not all wildlife are disturbed or excluded by trails. Mammalian nest predators, including raccoons, skunks, and coyotes, were observed to be common along trails (Miller and others 1998), and seem to be abundant in edge habitats (Gates and Gysel 1978).

6 Discussion and Conclusions

6.1 Effects of Exotic Plants and Animals

Exotic forest pests, including insects and plant pathogens, have changed the structure of some forest types, and changed the density and composition of wildlife associated with them. Exotic plant species have also displaced native forest trees and understory plants in some areas, but the resultant effects to forest wildlife are not well described. Exotic plants have been introduced to enhance wildlife habitat, but their indiscriminant use in the past has led to serious invasions. Exotic animals have harmed some forest wildlife by displacing native species, preying on native wildlife or damaging sensitive forest habitats. Only a small percentage of exotic species (4 to 19 percent) have been documented to cause great harm. Another 6 to 53 percent have neutral effects or their effects are not as yet documented.

A large number of potentially invasive exotic species can impact native wildlife and their habitats in the United States. New plant species continue to be imported. Approximately 6,741 plant species are recognized as weeds elsewhere in the world. Only 2,363 occur in the contiguous United States (Westbrooks 1998). In addition, an estimated 26,000 plant species are capable of becoming invasive once they are introduced into new environments (Campbell 1997). Approaches have been recommended for better predicting the invasive potential of exotic plant species (Mack 1996). They include simultaneous field comparisons between cogeners, one naturalized and one native, and following the fate of a species deliberately sown in a natural community beyond its current range, with or without environmental manipulation. Predictions may become better if several approaches are combined simultaneously.

Many of the most invasive plant species across the nation are still offered for sale (Campbell 1997). This is especially true for invasive forest exotics. About 67 percent of invasive forest vines, including kudzu, are still available for purchase along with about 90 percent of the most invasive forest trees. Federal and State governments have no unified policy for limiting entry, reacting to emergency importation threats, or fostering integrated control methods (Miller 1997). No regional agency or organization has clearly defined responsibility or jurisdiction to organize regional integrated weed management programs. Exotic pest plant councils have been formed in an attempt to address this gap, and various Federal agencies have formed the Federal Interagency Committee for Management of Noxious and Exotic Weeds. Control of exotic plants is further complicated by the fact that much of the forestland in the Southeast is privately owned. Less than 18 percent of forested land in the Southern Appalachians is publicly owned (SERAMBO 2000).

Many experts have published recommendations for dealing with the issue of exotic plants and animals (Campbell 1997, Stein and Flack 1996, and Miller 1997). Recommendations include:

- Development of more effective ways to prevent new introductions.
- Early detection and eradication of new exotics.
- Better control and management of established invaders.

- Protection and recovery of native species and ecosystems.
- Better public education and support for controlling exotics.
- Better integration of control efforts on the part of responsible government and non-governmental entities.
- Support for research aimed at identifying invasive species that could potentially damage our forests.
- Support for further research aimed at developing effective ways to control exotics.

6.2 Effects of Urbanization

Urbanization has resulted in the loss of forest habitat and fragmentation of forested landscapes. These habitat changes have had the greatest detrimental impacts to specialized forest wildlife species with narrow habitat requirements. Habitat generalists have been better able to adjust to changes brought about by urbanization. Based on the current trends of urbanization across the South, it is likely that forested habitats will continue to be permanently altered and the amount of available forest habitat will decrease in some areas. Increasing urbanization changes the species diversity, overall abundance, and, more importantly, shifts the species composition of forest wildlife. Some forest wildlife species are especially sensitive to fragmentation, forest edges, and human disturbance. Some species disappear from forest areas even with light levels of urban intrusion. Other species have lost the kind of early successional or quality-disturbed habitats that they require.

For species with area sensitivities, those that require forest interior, those that require specialized habitats, and those intolerant of human disturbance, special management considerations will be needed as urbanization increases in areas of the South. Some species will likely require forest conservation areas with thousands of acres of contiguous habitat to be successfully conserved. Protection may be needed to limit roads and human disturbance in these areas. Barring the feasibility of this conservation approach, finding several adjoining larger tracts or areas connected by corridors may be the next best alternative. To conserve forest wildlife species dependent on early successional habitats, forestry management strategies should be formulated to provide a constant availability of these habitats and provide connective corridors for low-vagility species.

With these considerations in mind, urban wildlife habitats will remain important for some wildlife species as suitable forest habitats decline in some urbanizing areas of the South. Urban wildlife preserves should be planned with the realization that size, habitat composition, connectivity, forest dynamics (management needs), and human perceptions of the preserve will ultimately affect the variety and composition of the species conserved there. Innovative designs in small conservation areas may be needed to avoid creating “ecological traps” for ground-nesting birds.

6.3 Effects of Agricultural Land Uses

Agricultural land uses have interrupted the continuity of southern forests, and created forest islands. Wildlife differ in their response to the resulting fragmentation. For some species of birds and small mammals, the forest/agricultural boundary acts as a barrier to movement, fragmenting and isolating populations. The presence of woody, vegetated fencerows may help to facilitate movement of some wildlife, however. Some long-distance migrant bird species and species that nest in forest interiors appear to be adversely affected by forest fragmentation particularly in heavily fragmented landscapes with low overall forest cover. The presence of nearby agricultural areas has been shown to reduce the nesting success of some forest bird species. Other taxa of wildlife also exhibit a species-specific response.

Many bird species dependent on open habitats such as grasslands, prairies, savannas, glades and barrens are now in serious decline in the Eastern United States. Agricultural areas, especially grasslands and fallow fields, provide habitat for some of these early successional birds and other wildlife such as eastern cottontails and quail. The presence of vegetated fencerows may further enhance the value of agricultural habitats for some wildlife species while decreasing the value for some grassland species.

Forest wildlife species utilize agricultural areas as foraging habitat. Foraging wildlife can be beneficial for agriculture when they consume insects, mice, or weed seeds. Consumption of crops can also be relatively harmless when it involves consumption of “waste” grain left behind by mechanical harvesters or consumption of fallen fruit. Still, damage to crops and consumption of agricultural commodities is an important issue, and has resulted in some wildlife species being subjected to lethal and non-lethal depredation control measures. The attraction of wildlife to agricultural areas has also subjected them to injury and death due to faster more powerful farm machinery, pesticides, and the dangers of other injury and disease.

Old-field successional habitats are important for some wildlife species, but may also serve as introduction points for exotic vegetation into the forest, especially along the edges of forest fragments (Brothers and Spinarn 1992). The former agricultural land use may affect the vegetative structure of the resulting old-field habitat, and restoration to full utility as habitat for forest wildlife may not occur even after a number of years.

Government programs that encourage the removal of land from intensive cultivation, the establishment of stable ground cover for soil conservation, and the deliberate creation of wildlife habitat areas in predominantly agricultural environments can greatly influence the abundance of and diversity of wildlife species (Bolen and Robinson 1995).

6.4 Effects of Linear Land Uses (Roads, Power Lines, and Trails)

The effects of roads and power line corridors on forest wildlife are species dependent. For some forest wildlife, the corridors exclude or result in avoidance of the area for distances of 330 feet or more. For grassland and early-successional forest species, roadsides and power line rights-of-way provide valuable habitat, but the value is influenced by the width of the corridor, the nature of the corridor vegetation, maintenance practices in the corridor, and the abruptness of the forest edge. For some forest wildlife species, roads and power line corridors act as barriers, fragmenting populations. Corridors can also act as intraspecific filters, allowing movement of a certain age class or gender. For other species, corridors act as travel lanes, connecting isolated areas of habitat. Unfortunately, roads and powerline corridors can also act as travel lanes for the spread of exotic plants and animals. Road mortality for many species of forest wildlife has been well documented. Speed limit, road type, width of the cleared corridor, and other factors affect the mortality levels found on a given highway segment. Roads also have other effects, including mortality due to increased access by legal and illegal hunters, increased pollution along roadsides, and accelerated land-use changes along roads.

Wildlife and plants can be affected by the presence of trails through the forest. Trampling by hikers and other outdoor recreationists have been found to cause declines in some sensitive plant species. In addition, shifts in forest bird composition have been documented along trails. Other wildlife such as bears are sensitive to human disturbance and may avoid trails. "Collectable" wildlife species may be extirpated from the vicinity of trails due to pet collection.

7 Needs for Additional Research

7.1 Effects of Exotic Plants and Animals

The effects of exotic plant invasions on forest wildlife remain poorly documented. Much of the information available is based on land-manager observations or expert opinions. There is a need for more scientific investigations to systematically document how southern forest wildlife communities on both local and regional scales are affected when forests are invaded by exotic plant species. "Early warning" research is needed to identify potentially invasive forest exotics to better guide quarantine efforts. Research is needed to develop more effective control and management tools for exotic plants and animals.

7.2 Human Land Use Changes

The effects of urbanization and agriculture are better understood for birds than other taxa of forest wildlife. More studies that take place in agricultural and urbanizing areas of southern forests would allow comparisons with avian species studied in other areas of North America. Species responses may differ across their respective ranges.

More information is needed about the effects of land-use changes on mammals, herpetofauna, and invertebrates in Southern forests to identify species likely to be adversely affected by urbanization.

More studies are needed that document which species are most likely to benefit from connective corridors used to overcome the deleterious effects of fragmentation. More research is needed to determine if corridors have adverse impacts on forest habitats and to identify circumstances under which adverse impacts should be expected.

More information is needed about the breeding success of ground-and low-nesting forest birds in small preserves. Information is needed to formulate management strategies that avoid the creation of "ecological traps" for breeding birds.

7.3 Linear Land Uses (Roads, Power Lines, and Trails)

Relatively little data on the effects of roads and power lines on forest wildlife are available for amphibians, reptiles and invertebrates. More information specific to wildlife in southern forests is needed to allow for behavioral differences from one part of a species range to another.

The effect of forest trails on wildlife is better documented for plants than other taxa. More information is needed about wildlife in southern forests.

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10 Tables and Figures

Table 1--Exotic invasive plants of southern forests. Sources: Miller 1997; USDA Forest Service 1999, and Rural Action Inc. 1999

Common Name	**Scientific Name**	**Plant Description**
Silktree or Mimosa tree	*Albizia julibrissin*	Tree
Chinaberry	*Melia azedarach*	Tree
Tallowtree or Popcorn tree	*Sapium sebiferum*	Tree
Tree of heaven or Stinktree	*Ailanthus altissima*	Tree
Empress or Princess tree	*Paulownia tomentosa*	Tree
Bicolor lespedeza	*Lespedeza bicolor*	Shrub
Burning bush	*Euonymus alatus*	Shrub
Japanese privet	*Ligustrum japonicum*	Shrub
Chinese privet	*Ligustrum sinense*	Shrub
Common privet	*Ligustrum vulgare*	Shrub
Multiflora rose	*Rosa multflora*	Shrub
Autumn olive	*Elaeagnus umbellata*	Shrub
Amur or Bush honeysuckle	*Lonicera maackii*	Shrub
Japanese barberry	*Berberis thunbergii*	Shrub
Japanese honeysuckle	*Lonicera japonica*	Vine
Japanese climbing fern	*Lygodium japonicum*	Vine
English ivy	*Hedera helix*	Vine
Kudzu	*Pueraria montana*	Vine
Mile-a-minute	*Polygonum perfoliatum*	Vine
Periwinkle	*Vinca minor*	Vine

Oriental bittersweet	*Celastrus orbiculatus*	Vine
Chinese wisteria	*Wisteria sinensis*	Vine
Winter creeper	*Euonmus fortunei*	Vine
Cogongrass	*Imperata cylindrica*	Grass
Japanese grass or stiltgrass	*Microstegium vimineum*	Grass
Johnsongrass	*Sorghum halepense*	Grass
Tall fescue	*Fescue elatior*	Grass
Common teasel	*Dipsacus sylvestris*	Herb
Crown vetch	*Coronilla varia*	Herb
Garlic mustard	*Alliaria petiolata*	Herb
Japanese knotweed	*Polygonum cuspidatum*	Herb
Musk thistle	*Carduus nutans*	Herb
Purple loosestrife	*Lythrum salicaria*	Herb
Sericea or Chinese lespedeza	*Lespedeza cuneata*	Herb
Spotted knapweed	*Centaurea maculosa*	Herb
Sweet clover	*Melilotus alba*	Herb

Return to first reference in text

Table 2--Introduced terrestrial wildlife species of the Southeastern United States. Adapted from: Echternacht and Harris 1993

Common Name	**Scientific Name**	**Animal Description**
Giant toad	*Bufo marinus*	Amphibian
Cuban treefrog	*Osteopilus septentrionalis*	Amphibian
Greenhouse frog	*Eleutherodactylus planirostris*	Amphibian
Puerto Rican coqui	*Eleutherodactylus* coqui	Amphibian
Spectacled caiman	*Caiman crocodilus*	Reptile
Tokay gecko	*Gekko gecko*	Reptile
Ashy gecko	*Sphaerodactylus elegans*	Reptile
Ocellated gecko	*Sphaerodactylus argus*	Reptile
Yellowhead gecko	*Gonatodes albogularis*	Reptile
Indo-Pacific gecko	*Hemidactylus garnotti*	Reptile
Mediterranean gecko	*Hemidactylus turcicus*	Reptile
Jamaican giant anole	*Anolis garmani*	Reptile
Brown anole	*Anolis sagrei*	Reptile
Largehead anole	*Anolis cybotes*	Reptile
Knight anole	*Anolis equestris*	Reptile
Puerto Rican crested anole	*Anolis cristatellus*	Reptile
Brown basilisk	*Basiliscus vittatus*	Reptile
Green iguana	*Iguana iguana*	Reptile
Spinytail iguana	*Ctenosaura pectinata*	Reptile
Northern curlytail lizard	*Leiocephalus carinatus*	Reptile

Red-sided curlytail lizard	*Leiocephalus shreibersi*	Reptile
Rainbow whiptail	*Cnemidophorus lemniscatus*	Reptile
Giant amevia	*Ameiva ameiva*	Reptile
Braminy blind snake	*Ramphotyphlops braminus*	Reptile
Black francolin	*Francolinus francolinus*	Bird
Ring-necked (green) pheasant	*Phasianus colchicus*	Bird
Plain chacalaca	*Ortalis vetula*	Bird
Rock dove	*Columba livia*	Bird
Ringed turtle-dove	*Streptopelia risoria*	Bird
Rose-ringed parakeet	*Psittacula krameri*	Bird
Budgerigar	*Melopsittacus undulatus*	Bird
Canary-winged parakeet	*Brotogeris versicolurus*	Bird
Monk parakeet	*Myiopsitta monachus*	Bird
Red-crowned parrot	*Amazona viridigenalis*	Bird
Yellow-headed parrot	*Amazona oratrix*	Bird
Red-whiskered bulbul	*Pycnonotus jocosus*	Bird
European starling	*Sternus vulgaris*	Bird
Hill myna	*Gracula religiosa*	Bird
Spot-breasted oriole	*Icterus pectoralis*	Bird
Java sparrow	*Padda oryzivora*	Bird
House sparrow	*Passer domesticus*	Bird
Squirrel monkey	*Saimiri sciureus*	Mammal
Rhesus macaque	*Macaca mulatta*	Mammal

Red-bellied squirrel	*Sciurus aureogaster*	Mammal
Black rat	*Rattus rattus*	Mammal
Norway rat	*Rattus norvegicus*	Mammal
House mouse	*Mus musculus*	Mammal
Nutria	*Myocaster coypus*	Mammal
Wild boar	*Sus scrofa*	Mammal
Sambar deer	*Cervus unicolor*	Mammal

Return to first reference in text

Table 3--Some southeastern forest bird species and their sensitivities to urban and suburban development[*]

Common name	**Scientific name**	**Urban/suburban association**
Mature-forest assemblage (late-successional forests)		
Pine warbler	*Dendroica pinus*	tolerant
Red-eyed vireo	*Vireo olivaceus*	intolerant
Red-bellied woodpecker	*Melanerpes carolinus*	tolerant
Wood thrush	*Hylocichla mustelina*	intolerant
Ovenbird	*Seiurus aurocapollus*	intolerant
Hooded warbler	*Wilsonia citrina*	intolerant
Acadian flycatcher	*Empidonax virescens*	intolerant
Scarlet tanager	*Piranga olivacea*	intolerant
Northern parula	*Parula americana*	intolerant
Black-and-white warbler	*Mniotilta varia*	intolerant
Hairy woodpecker	*Picoides villosus*	tolerant
Pileated woodpecker	*Dryocopus pileatus*	intolerant
Yellow-throated warbler	*Dendroica dominica*	intolerant
Prothonotary warbler	*Protonotaria citrea*	intolerant
Kentucky warbler	*Oporornis formosus*	intolerant
Louisiana waterthrush	*Seiurus motacilla*	intolerant
Shrubland assemblage (early-successional clearcuts)		
Indigo bunting	*Passerina cyanea*	intolerant
Yellow-breasted chat	*Icteria virens*	intolerant

Common yellow-throat	*Geothlypis trichas*	intolerant
White-eyed vireo	*Vireo griseus*	intolerant
Prairie warbler	*Dendroica discolor*	intolerant
Field sparrow	*Spizella pusilla*	intolerant
Gray catbird	*Dumetella carolinensis*	tolerant
Forest-edge assemblage (fragmented landscapes)		
Brown-headed cowbird	*Molothrus ater*	tolerant
Northern mockingbird	*Mimus polyglottos*	tolerant
Chipping sparrow	*Spizella passerina*	tolerant
American robin	*Turdus migratorius*	tolerant
Eastern bluebird	*Sialia sialis*	tolerant
Common grackle	*Quiscalus quiscula*	tolerant
Eastern kingbird	*Tyrannus tyrannus*	rural/agricultural
Red-headed woodpecker	*Melanerpes erythrocephalus*	somewhat tolerant
Orchard oriole	*Icterus spurius*	rural/agricultural
House finch	*Carpodacus mexicanus*	tolerant
Habitat generalist assemblage		
Cardinal	*Cardinalis cardinalis*	tolerant
Carolina wren	*Thryothorus ludovicianus*	tolerant
Tufted titmouse	*Baeolophus bicolor*	tolerant
Blue-gray Gnatcatcher	*Polioptila caerulea*	intolerant
Carolina chickadee	*Poecile carolinensis*	tolerant
Blue jay	*Cyanocitta cristata*	tolerant

Great crested flycatcher	*Myiarchus crinitus*	somewhat tolerant
Summer tanager	*Piranga rubra*	intolerant
Downy woodpecker	*Picoides pubescens*	tolerant
Yellow-billed cuckoo	*Coccyzus americanus*	intolerant
Eastern wood pewee	*Contopus virens*	intolerant
Mourning dove	*Zenaida macroura*	tolerant
Common crow	*Corvus brachyrhynchos*	tolerant
Northern bobwhite	*Colinus virginianus*	intolerant
Brown thrasher	*Toxostoma rufum*	intolerant
Northern flicker	*Colaptes auratus*	tolerant
American goldfinch	*Carduelis tristis*	tolerant
Red-shouldered hawk	*Buteo lineatus*	tolerant
Yellow-throated vireo	*Vireo flavifrons*	intolerant
Ruby-throated hummingbird	*Archilochus colubris*	tolerant
Eastern phoebe	*Sayornis phoebe*	tolerant
Eastern screech-owl	*Otus asio*	tolerant
Common nighthawk	*Chordeiles minor*	tolerant
White-breasted nuthatch	*Sitta carolinensis*	tolerant

[*] Adapted from: Canterbury and others, 2000. Based on results from: Engels and Sexton (1994), Smith and Schaefer (1992), Dowd (1992), Beissinger and Osborne (1982), Rottenborn (1999), Linehan and others (1967), Blair (1996), Goldstein and others (1986), Friesen and others (1995), Long and Long (1992), Askins and Philbrick (1987), Aldrich and Coffin (1980), Bolen and Robinson (1995), Zimmerman (1991), and Hines and Anastasi (1973).

Return to first reference in text