Criteria for Evaluating Impacts of Development on Wildlife Habitats

Randy S. Kautz, *Office of Environmental Services, Florida Game and Fresh Water Fish Commission, Tallahassee, FL 32301*

Abstract: Determinations of the habitat values of a particular site can be viewed from the perspective of either individual wildlife species or the wildlife community as a whole. When reviewing development proposals, the habitat values of a site to particular species are most important when species listed as endangered or threatened are present since these species are among our highest conservation priorities. In the absence of listed species, the habitat values of a site to the entire wildlife community assume importance. Major factors which influence the richness of species and abundance of wildlife on a site include habitat type, successional stage, habitat island size, presence or absence of connecting corridors, edge effects, interspersion of various habitat types, and presence of snags or fallen logs. Impacts of development proposals can be assessed by comparing these attributes of a site before and after development.

Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies 38:121-136

Every day, wildlife habitat is being altered or destroyed throughout the southeastern United States to accommodate myriad human needs. Virtually every wetland and upland habitat type is affected by development in 1 way or another. In some states, such as Florida, the opportunity exists for wildlife biologists to provide input to a variety of development activities including dredge and fill projects in wetlands, large residential and commercial developments, mined land reclamation, and power plant and transmission line siting. An inherent difficulty in making recommendations to mitigate the impacts of development on wildlife is a thorough understanding by the biologist of the habitat requirements of a large number of species, the habitat values of different plant community types, and the effects of various perturbations. Florida, for example, is inhabited by more than 150 species of reptiles and amphibians, 425 species of birds, and approximately 80 species of mammals (Tanner and Smith 1981) that are distributed over 17 major habitat types excluding open water estuarine and marine habitats (Davis 1967).

Developing an understanding of all interactions among over 650 species of wildlife and 17 major habitat types is a virtually impossible task, particularly when considering the effects of newly-created habitat types such as intensively-managed forests, urban areas, agricultural lands, and improved pastures. Such a task becomes more intractable when the problem is extended over the entire Southeast. Since permit applications for development projects are received at a rapid rate and since the need exists to provide quick and meaningful recommendations to permitting agencies, a small number of habitat evaluation criteria is needed to reduce the complexity of wildlife-habitat interactions. This paper, therefore, is an effort to synthesize an existing body of literature to facilitate wildlife habitat evaluation and impact assessment. The focus will be on wildlife communities instead of individual species; however, individual species may be important in certain cases.

The author thanks B. Hartman, L. Harris, F. James, D. Simberloff, S. Godley, M. Allen, N. Eichholz, A. Egbert, D. McElveen, and F. Stanberry for their reviews of the manuscript and is also indebted to the Florida Game and Fresh Water Fish Commission for allowing the time to pursue the subject.

Methods

A literature review was conducted to identify a small number of habitat variables that, to a large extent, determine the value of various habitat types to wildlife communities. While the literature review included studies from throughout the United States, an effort was made to focus on studies from the Southeast. In addition, several Florida studies were used to develop information on species use of major Florida habitats to illustrate the value of some of the major criteria.

Results

None of the literature reviewed contained a simple list of criteria for use in habitat assessments that can be used in all places at all times. However, the Habitat Evaluation Procedures of the U.S. Fish and Wildlife Service, the habitat models of the U.S. Army Corps of Engineers (1980), and the literature review conducted for this paper suggest that the following habitat features are significant criteria for use in development impact assessments: (1) habitat type, (2) successional stage, (3) habitat island size, (4) corridors, (5) edge effects, (6) interspersion of habitat types, and (7) snags.

In addition, endangered and threatened species are identified as a major resource concern that must be considered in impact assessments.

Habitat Type

Habitat is generally defined as the place where an organism lives (Odum 1971). Shelford (1963) and Orr (1971) have pointed out that individual species may be broadly adapted to a large number of habitats occurring over a large area, or they may be restricted to 1 or only a few specific habitat types. Thus, specific habitat types are utilized by a wildlife community composed of a mixture of generalist and specialist species. One measure of the habitat value of a particular plant community (i.e., habitat type) is the total number of vertebrate species that utilizes the plant community in question. Although these data apparently do not exist for the major plant communities of the Southeast, I have compiled such information for 14 Florida habitat types, many of which occur throughout the Southeast (Table 1).

Natural pine flatwoods support the largest total number of species of wildlife in Florida whereas intensively-managed pine plantations support the least. Amphibian species are most abundant in cypress-tupelo (*Taxodium* sp.-*Nyssa* sp.) swamps and are least abundant in xeric and altered (i.e., improved pasture, pine plantation) habitats. Species of reptiles are most abundant in xeric habitats, and they are least abundant in improved pasture and mixed swamp habitats. Bird species richness is greatest in mixed swamp habitats and least in bayheads, another wetland type. Species richness of mammals is greatest in scrub sand pine (*Pinus clausa*) and longleaf pine-turkey oak (*P. paulstris-Quercus laevis*) sandhills, 2 xeric habitats, and is least in cypress-tupelo swamps.

The total number of species utilizing a habitat type is an important statistic, but it does not tell the whole story because many species may be found in more than 1 habitat type. To ascertain the value of the 14 Florida habitat

Table 1. Number of vertebrate species utilizing 14 north Florida habitat types. Data derived from Nelson (1952), Florida Game and Fresh Water Fish Commission (1976), McElveen (1977), Southwest, Florida Water Management District (1980, 1981), and Cutright (1981).

	N of species				
Habitat type	Amphibians	Reptiles	Birds	Mammals	Total
Pine flatwoods	19	26	81	23	149
Sandhills	10	33	76	25	144
Mixed swamp	10	14	98	18	140
Xeric hammock	11	31	78	19	139
Cypress-tupelo swamp	27	27	73	5	132
Sand pine scrub	9	25	67	28	129
Marsh and prairie	15	24	72	14	125
Mesic hammock	12	19	59	21	111
Pine-hardwood forest	14	21	62	14	111
Hydric hammock	15	21	54	16	106
Improved pasture	8	14	68	14	104
Bayhead	20	27	43	10	100
Coastal hammock	6	26	49	10	91
Pine plantation	4	18	52	12	86
Mean	13	23	67	16	119

Table 2. Number of species utilizing no more than 3 of 14 Florida habitat types.

Habitat type	*N* of species				
	Amphibians	Reptiles	Birds	Mammals	Total
Pine flatwoods	2	2	7	1	12
Sandhills	0	6	8	3	17
Mixed swamp	3	2	24	6	35
Xeric hammock	0	3	8	0	11
Cypress-tupelo swamp	7	5	9	0	21
Sand pine scrub	0	2	5	4	11
Marsh and prairie	3	8	20	2	33
Mesic hammock	0	1	2	2	5
Pine-hardwood forest	0	1	3	1	5
Hydric hammock	2	2	2	2	8
Improved pasture	0	0	15	2	17
Bayhead	3	3	0	0	6
Coastal hammock	0	4	4	1	9
Pine plantation	0	0	1	1	2
Mean	1	3	8	2	14

types to species having more specialized habitat requirements, the number of species occurring in no more than 3 habitat types were tabulated (Table 2). Mixed swamp and marsh and prairie habitats, 2 wetland types, support the greatest number of specialist species whereas pine plantations support the least. The specialist species of the 2 wetland types are predominantly birds. Mixed swamp also supports the greatest number of specialized mammals, and marshes and prairies support the greatest number of specialized reptiles. Cypress-tupelo swamps have the greatest number of specialized amphibians. Overall, wetlands comprise the most important habitat types for specialized species of wildlife.

Successional Stage

With respect to wildlife, particularly birds, forest succession may be considered in 4 stages: bare ground and grass, sapling and shrub, pole timber, and mature forest. Bird species diversity and density are generally lowest in the bare ground and grass stage immediately following clearcutting; they increase to maximum values in the pole timber stage, and decline slightly in mature forests (Kendeigh 1946, Johnston and Odum 1956, Karr 1968, Karr 1971, Shugart and James 1973, Noble and Hamilton 1975, Dickson 1978). The increases in bird species diversity and bird density have been correlated with increasing structural diversity and increasing plant species diversity as succession progresses (MacArthur and MacArthur 1961, Karr and Roth 1971, Willson 1974, Balda 1975, Meyers and Johnson 1978). The decline in diversity and density in mature forests relates to the reduction in ground cover and understory plants that are eliminated by canopy closure.

A notable exception to the trend of increasing bird species diversity occurs in southeastern forests intensively managed for pines (*Pinus* spp.). Namely, following canopy closure at 8 to 12 years after planting, diversity and density decline drastically due to the loss of midstory and ground layers and the reduction in plant species diversity. Diversity and density remain low throughout the remainder of the commercial rotation unless the plantation is thinned and sunlight stimulates the production of herbs and shrubs beneath the canopy or unless a longer rotation for sawtimber is used. Commercial forests are rarely, if ever, allowed to stand until ecological maturity has been reached.

The increase in bird species diversity with increasing ecological age occurs at the expense of some species. For example, species such as eastern bluebird (*Sialia sialis*), indigo bunting (*Passerina cyanea*), eastern meadowlark (*Sturnella magna*), and common flicker (*Colaptes auratus*), are common to early successional stages but are eliminated by the time the young forest stage has been reached. On the other hand, species such as barred owl (*Strix varia*), yellow-billed cuckoo (*Coccyzus americanus*), Acadian flycatcher (*Empidonax virescens*), and red-eyed vireo (*Vireo olivaceus*) occur only in mature forests and are not present in early successional stages.

Small mammals generally reach highest diversity and abundance in the first 3 to 5 years following clearcut but decline drastically as succession progresses (Kirkland 1974, Umber and Harris 1975, Atkeson and Johnson 1977, Mengak and Tipton 1978). Larger herbivores, such as white-tailed deer (*Odocoileus virginianus*), and predators, such as bobcat (*Lynx rufus*), will utilize clearcuts up until the time of canopy closure at which time plant and prey species diversity declines. As succession progresses to the stage of the mature forest, species such as gray squirrel (*Sciurus carolinensis*), flying squirrel (*Glaucomys volans*), gray fox (*Urocyon cinereoargenteus*), and black bear (*Ursus americanus*) become more common if sufficient area is present to meet their needs.

So far, few references have been found on the effects of successional stage on herpetofauna. However, P. Moler (pers. commun.) believes that the early successional stages that result from clearcutting do not adversely affect amphibians in southeastern flatwoods as long as wetland forests are not cut or drained. Landers and Speake (1980) have shown that gopher tortoise (*Gopherus polyphemus*) densities are greatest in natural longleaf pine-turkey oak sandhills that burn every 2 to 4 years; however, young pine plantations, especially those frequently burned, may provide better gopher tortoise habitat than degraded natural areas (Umber and Harris 1975, Landers and Speake 1980, Auffenberg and Franz 1982). Gopher tortoise burrows, in turn, provide habitat for various herpetofaunal species including gopher frog (*Rana areolata*) and eastern indigo snake (*Drymarchon corais couperi*).

Habitat Island Size

Habitat island size directly influences the species diversity and abundance of breeding and migratory birds in a variety of habitats (Moore and Hooper 1975, Forman et al. 1976, Galli et al. 1976, Martin 1980, Harris 1980, Martin

1981, Harris 1984, Harris and Wallace, this volume). In general, the number of bird species responds logarithmically to forest area, and a sevenfold increase in area produces a doubling of the number of species of birds encountered. Abundance, on the other hand, responds linearly to area.

In Florida, the number of species observed in forested areas increases rapidly from 0 to approximately 8 species as patch area increases from 0 to 2 ha (Harris and Wallace, this volume). The species of birds occurring in small forest patches of this size are predominantly edge-adapted species such as cardinal (*Cardinalis cardinalis*), mockingbird (*Mimus polyglottos*), blue jay (*Cyanocitta cristata*), rufous-sided towhee (*Pipilo erythrophthalmus*), and yellowthroat (*Geothlypis trichas*). Birds adapted to forest interiors are added to the community as patch size increases beyond the minimum home range or territory size requirements of individual species. Exceptions to this may occur, however. For example, in some of the studies reviewed, forest interior birds were absent from some areas which were large enough to support them. Conversely, species adapted to forest interiors were sometimes observed in forest patches smaller than their accepted average territory-size requirement. The reasons for these observations are not entirely clear; however, Martin (1980) suggested that site-specific habitat conditions, competition between species in the same guild, and chance colonizations may all play a role in determining which species actually occur at a given site.

Information from the literature on breeding bird habitat specificity and territory size can be used to estimate the value of various forest patch sizes as breeding bird habitat. Small forest patches may be considered as predominantly edge habitats that support only edge-adapted species. To determine the size of forest patches that are primarily edge, it is first necessary to estimate the average width of edge habitats. Wales (1972) determined that mixed oak forest edges in the New Jersey Piedmont extend approximately 10 m inside a forest on the north side and approximately 20 m on the south side. Gates and Mosher (1981) found that edge widths in Michigan woods varied between 8.9 m and 64.4 m, and the average of 3 different measures of edge width was 26.4 m. In a study of edge effects in north-central Florida, McElveen (1977) arbitrarily defined edge as a strip 20 m wide extending 10 m into each adjoining community. Strelke and Dickson (1980) found that highest bird densities occurred in an arbitrarily defined edge 25 m wide in pine-hardwood forests in east Texas.

If the average width of edge in southeastern forests can be assumed to be 20 m, and if one assumes that a forest patch no more than 40 m wide would be all edge habitat, then a forest in the shape of a circle with a radius of 20 m would be edge habitat only. A patch of this size and shape would include an area of 0.13 ha and should support only edge-adapted species. Linear forests of indefinite length but less than 40 m wide (e.g., fencerows) should be considered as edge habitat that would support no forest-interior species; however, Martin (1981) observed forest-interior birds in linear shelterbelts as small as

0.3 ha but of unspecified width. Presumably, as linear forest patches increase in either length or width, the increase in area allows such a corridor to gradually assume some forest characteristics and thereby provide more habitat for species adapted to forest interiors.

Martin (1980) pointed out that size-dependent species occupy an area only if the area meets the minimum territory size requirement of the species. Thus, in order for a forest patch to support at least 1 breeding pair of a forest-interior species, the patch size must meet the minimum territory size requirement of the species with the smallest territory size. With an average territory size of 0.61 ha (Williamson 1971), the red-eyed vireo has one of the smallest territory size requirements of common southeastern forest-interior birds for which data are readily available (Shugart et al. 1978). Assuming again that the edge of a 0.61 ha forest island is 20 m wide, a circular forest patch of this size would comprise 70% edge habitat, and edge-adapted species would dominate. The empirical data of Harris and Wallace (this volume) for north-central Florida indicate that a 0.61-ha patch of mesic hardwoods would support 4.3 species of breeding birds, only 1 of which would be a forest interior species. The actual number of species occurring in any given forest will vary from this because the patch size necessary to support breeding pairs of any species will vary with the quality of breeding habitat as well as size (Galli et al. 1976, Rusterholz and Howe 1979).

In order for a forest patch to support a full complement of forest-interior bird species, the patch must be large enough to support at least 1 breeding pair of the forest-interior species with the largest territory size requirement. With an average territory size of 30.4 ha (Fitch 1958), the barred owl is one of the southeastern forest-interior species with a very large territory-size requirement (Shugart et al. 1978). A circular, 30.4-ha forest patch would contain 88% forest-interior habitat and 12% edge habitat. Based on the work of Harris and Wallace (this volume), a 30.4 ha mesic forest in Florida should support approximately 28 species of birds of which 17 species would use the forest for breeding habitat.

While a 30.4-ha forest should in theory support a breeding pair of barred owls, it might well be asked whether or not that is a sufficiently large population to prevent the local extinction of barred owls in the forest patch over the long run. Lynch and Whitcomb (1978) report that many forests experience repeated colonizations and extinctions by various species of birds. While they observed that turnover rates were lowest in a large, continuous expanse of forest and highest in their smallest study plot, their data base was too small to determine an effect of patch size on turnover rate. Presumably, species' use of a particular site will vary with year-to-year changes in habitat quality. Species which maintain large territories may be absent from a site during unusually harsh years. Thus, a 30.4-ha forest may be able to support 1 breeding pair of barred owls only during "good" years; during "bad" years, barred owls would probably be absent as a breeding species from a patch this small.

The question of minimum population size may also be viewed from a genetic perspective. Franklin (1980) argued that an effective population size of 50 individuals is necessary to prevent inbreeding depression and that an effective population of 500 individuals is necessary to maintain sufficient genetic diversity to allow for evolution in the event of major environmental changes. Berry (1971), however, has suggested that an effective population size of 50 individuals is sufficient to both prevent inbreeding depression and allow for evolution. D. Simberloff (pers. commun.) reports that recent unpublished data suggest that a minimum effective population size of 50 to 150 is more nearly correct from a genetic standpoint. Therefore, if a forest patch is so isolated from other forests that colonization by otherwise expected species does not occur, a forest patch should be at least large enough to support an effective population of at least 50 individuals, and maybe 150 individuals, of each species to prevent local extinction. In the case of the barred owl, if a range of from 25 to 75 breeding pairs can be assumed to be an effective population, a forest patch of from 760 ha to 2,280 ha in area would be necessary to prevent the local extirpation of barred owls if the forest patch is so isolated that colonization by barred owls from other populations is unlikely. Forests this large should also include even larger populations of many other forest-interior bird species.

Forest patch size is a wildlife resource concern that is assuming increasing importance to managers. Lynch and Whitcomb (1978) have shown that the diversity and abundance of breeding birds in the deciduous forests of the eastern United States have declined dramatically over the last 3 decades; and highly migratory, insectivorous, forest-interior species have suffered the most. Lynch and Whitcomb feel this observation has been the result of the fragmentation of the eastern forest, increasing isolation of remaining forest patches, and decreasing forest patch size. They reach this conclusion because remaining large forest patches appear to have supported stable breeding bird populations over the period of record whereas smaller forest patches have not. These species may also be affected by the destruction and fragmentation of tropical forests in the areas where they spend the winter.

From a conservation or land use planning standpoint, the question has been raised as to whether several small forests are as valuable to wildlife as 1 large forest of equal area. Forman et al. (1976) have stated that 1 large forest island contains a larger number of species of birds than an equal area of 2 or more small islands. They conclude that land use planning should emphasize protecting large forests rather than an equal area of small forests to maximize bird species diversity. In the New Jersey Piedmont, Forman et al. (1976) indicate that the most valuable forests are larger than 40 ha in size and that more than 3 large forests are needed within a 500 sq km area of the Piedmont in order to maintain the maximum regional diversity of upland forest birds.

The concepts of habitat island size probably also apply to herpetofauna

(P. Moler, pers. commun.). The eastern indigo snake, which has one of the largest home range sizes of southeastern herpetofauna excluding crocodilians and sea turtles, has a home range of from 40 ha to 80 ha (Moler, unpubl. data). Since most reptiles and amphibians have home ranges considerably smaller than this, preservation strategies aimed at protecting birds or mammals with larger home ranges than indigo snakes should also protect herpetofauna in general.

Similarly, forest management strategies that protect breeding birds will also protect small, forest-adapted mammals such as the gray squirrel which has a home range size of from 0.5 ha to 16 ha (Allen 1982). However, conservation of habitat large enough to support large wide-ranging mammals poses a problem. For example, gray foxes have a monthly home range size of 299 ha in Missouri (Haroldson and Fritzell 1984) and black bears have home range sizes of from 3,900 ha to 16,300 ha in northern Florida (D. Maehr, pers. commun.). Both of these species prefer large areas dominated by forests; but, since these species have overlapping, non-exclusive home ranges, it is difficult to determine population densities. This makes it almost impossible to determine the minimum area required to preserve an effective population of from 50 to 150 individuals. Suffice it to say that large areas of relatively wild forest lands are necessary to maintain the populations of wide-ranging species.

Corridors

MacClintock et al. (1977) have provided evidence that connecting small isolated forest patches with larger forests via forested corridors serves to increase the number and diversity of breeding birds in the smaller forest. Essentially, forested corridors serve to decrease the isolation of forest fragments and increase the probability that forest fragments will support a larger number of species. Harris (1984) has explored this concept further and has suggested the use of riparian corridors to link isolated habitat islands. Streamside management zones, which are corridors of forested vegetation left along stream bottoms in areas managed for timber, may be ideally suited to this purpose. Warren and Hurst (1980) have shown that gray squirrel densities are high in streamside management zones in Mississippi, and they cite another study that demonstrated the value of bottomland travel corridors to turkeys (*Meleagris gallopavo*). The ideal width of a corridor is unknown, but the corridor in the study of MacClintock et al. (1977) varied from 100 m to 200 m wide, and the streamside management zone in the Warren and Hurst (1980) study was 100 m wide.

Edge Effects

Edge effect may be described as the tendency for a greater number of species and greater number of individuals to be found along the edge of 2 adjoining communities than within the interiors of the 2 communities (Leopold 1933). Anderson et al. (1977), McElveen (1977), Strelke and Dickson

(1980), Chasko and Gates (1982), and Kroodsma (1984) have demonstrated that higher numbers of birds and bird species occur in forest edges 25 m wide than occur within forest interiors or within adjacent plant communities of earlier successional stage. The tendency is largely related to the development of a dense shrub layer beneath the forest canopy at the forest edge due to increased sunlight. The shrub layer attracts species more typical of nonforested communities such as cardinal, prairie warbler (*Dendroica discolor*), yellow-breasted chat (*Icteria virens*), and indigo bunting. At the same time, the forest canopy at the edge is utilized by birds typical of forest interiors, but these species do not utilize adjacent shrub and herbaceous habitats. The edge effect becomes less pronounced as the less mature plant community progresses to the shrub stage, which lessens the contrast between successional stages and leaves early successional stage species less dependent on the shrubby growth within the forest edge. Use of the less mature habitats is largely related to the patchiness of shrubs (Roth 1976).

Interspersion of Habitat Types

Harris (1980) has pointed out that many species of wildlife, particularly wide-ranging animals, require a variety of habitat types within their home ranges for existence. For example, Maehr and Brady (1984) have shown that black bears in Florida require a variety of habitat types in order to meet seasonal food requirements. Similarly, Umber and Harris (1975) estimated that white-tailed deer (*Odocoileus virginianus*) were more abundant in planted pines in the spring but moved to areas dominated by oaks (*Quercus* spp.) in the fall.

Of 284 species of vertebrates using the 14 major habitat types appearing in Table 1, 64% of them utilize 4 or more different habitat types. Although some of the species are ubiquitous opportunists that survive almost anywhere, many are dependent on several habitat types in close proximity to meet their requirements for food, cover, and reproduction.

Wetland and aquatic habitats are particularly important to the maintenance of an amphibian fauna. Fully aquatic amphibians, such as the two-toed amphiuma (*Amphuima means*), dwarf siren (*Pseudobranchus striatus*), greater siren (*Siren lacertina*), and most of the ranid frogs, require permanent water. Most salamanders and hylid frogs require seasonally flooded wetlands for reproduction but disperse throughout adjacent uplands for the remainder of the year. The presence of streams, lakes, ponds, and wetlands scattered throughout the landscape are especially valuable to the production of these small vertebrates that are essential to the food chains of larger wildlife species.

Snags

Snags may be defined as any dead, dying, or living trees suitable as nest sites for cavity nesting birds (Conner 1978). Species of birds that are primary or secondary cavity nesters include wood duck (*Aix sponsa*), American kestrel (*Falco sparverius*), screech owl (*Otus asio*), virtually all woodpeckers (*Picidae*), great-crested flycatcher (*Myiarchus crinitus*), tree swallow (*Iridoprocne bicolor*), Carolina chickadee (*Parus carolinensis*), tufted titmouse (*Parus bicolor*), brown-headed nuthatch (*Sitta pusilla*), brown creeper (*Certhia familiaris*), Carolina wren (*Troglodytes ludovicianus*), and eastern bluebird. Other species which utilize snags are squirrels, mice, wood rats, bats, raccoons (*Procyon lotor*), and opossums (*Didelphia marsupialis*). Without snags in forested and clearcut areas, a major segment of the wildlife community will be reduced or absent, a fact appreciated by Balda (1975) and Zeedyk and Evans (1975).

Conner et al. (1983) have provided estimates of the number of snags required by various woodpeckers. They suggest that excellent habitat would have as few as 5 snags per 4 ha for common flickers and pileated woodpeckers (*Dryocopus pileatus*) to as many as 40 snags per 4 ha for downy woodpeckers (*Picoides pubescens*). Poor habitat would have only 1 snag per 4 ha for common flickers and pileated woodpeckers and would have 8 snags per 4 ha for downy woodpeckers. The snag requirements of other woodpeckers fall between these extremes. Conner et al. (1983) provided no estimates of the snag requirements of other vertebrate species.

In addition, decaying logs on the forest floor are undoubtedly important to the production of small amphibians, reptiles, and mammals; no literature could be found documenting this. Speake et al. (1978) have pointed out that windrows of debris left as a result of site preparation in managed forests provide cover and food for indigo snakes as well as other organisms. To some extent, windrows mimic the habitat values provided by fallen logs.

Endangered and Threatened Species

Endangered and threatened species, listed by either state or Federal governments, are among the highest conservation priorities. Reasons for listing may vary from over-collection (e.g., indigo snake) and biomagnification of pesticides (e.g., peregrine falcon, *Falco peregrinus*) to habitat destruction (e.g., wood stork, *Mycteria americana*). Whatever the reason for listing, the presence of these species at a project site should play a major role in the development of mitigation plans for the proposed development. Each state also has species not yet listed that should be given special consideration to ensure that their plights do not become severe.

Discussion

When evaluating the impacts of development projects on wildlife habitats, it is first necessary to gain an understanding of the pre-existing habitat values of a project site. Since it is an almost impossible task to determine the habitat value of a site for all vertebrate species that occur in a given area, it is necessary to evaluate the quality of a number of habitat features that have known importance to a wide variety of species. The habitat criteria suggested for consideration herein include: habitat type, successional stage, habitat island size, connecting corridors, edge effects, interspersion of habitat types, and number of snags present. The quality of each of these features gives an indication of the overall habitat value of a site.

Perhaps the first conservation priority on the site of a development project concerns the protection of listed endangered and threatened species. The locations of these species on a site must be known, and probable impacts of a project should be determined based on the type of development and the life history requirements of the species involved. Recommendations can then be formulated for the protection of these species.

Once the concerns of endangered and threatened species have been resolved, impacts on the wildlife community in general should be assessed using the evaluation criteria outlined above to compare pre- and post-development conditions. Generally speaking, development should be sited in early successional stage habitats in order to impact the least number of species. Where development must occur in forested areas, care should be taken to preserve large forest islands, preferably as large as 2,000 ha if possible. Forest islands remaining within a development should be at least as large as 1 ha to 2 ha in order to attract some forest-interior species. Forest patches should also be connected to each other and ultimately to 1 large forest patch with forested corridors approximately 100 m wide to enhance wildlife use of the smaller patches. Areas that are permanently or seasonably flooded should be retained on site to maintain herpetofaunal populations. Representations of as many habitat types as possible of acceptable size should be left on site to preserve habitat diversity, and it is important that wildlife be able to travel between separated habitat types without interference. Forests remaining on a project site should be managed so as to maintain an adequate number of snags for cavity-nesting wildlife.

The actual recommendations concerning wildlife mitigation on a particular site depend upon the habitat types present, the size of the area, and the wildlife conservation objectives at the site. The wildlife objectives are likely to involve value judgments on the part of the biologist. For example, a popular recommendation for increasing deer populations in a large homogeneous expanse of forest is to cut small openings in the forest. The creation of early successional stage habitats in the forest and the increase in edge length increases the value of a site for species adapted to early successional stages and edges. However, this occurs at the expense of species adapted to forest interiors, particularly birds. Temple (unpubl. data) has shown that predation on birds is higher along forest edges and that some bird species occur only in the core area of a forest (i.e., the forest interior excluding edges). Thus, chopping up a large forest with openings may benefit deer and early successional stage species, but the populations of forest interior species will decline as a result of the loss of core area habitat and increased predation along edges. It is this author's opinion that in the southeast today, fragmented forests, early successional stage habitats, and edge habitats are in abundance and that large blocks of undisturbed forest are at a premium. This would suggest recommendations for preserving large intact forested tracts even if it means fewer deer.

Problems such as the siting of transportation/utility corridors can be treated in a similar fashion. For example, a pertinent question may be whether it is better to recommend 1 large corridor or several small ones. Several small corridors cutting through a forested area will undoubtedly benefit species such as deer, but core areas will be greatly reduced. One large corridor cutting through a forest will maintain more core area for forest interior species, but it will also reduce the amount of edge habitat available and will probably produce lower deer populations. The final recommendation from the biologist will have to be based on a clear understanding of the wildlife conservation needs or priorities of an area. Some species of wildlife will benefit and some will suffer from any decision.

In Florida, sandhill, sand pine scrub, and xeric hammock habitats are important because they are limited in areal extent, and they are home to a large number of endemic reptiles and amphibians (Carr 1940, Campbell and Christman 1982). In addition, wetland habitats, in general, are important travel corridors and are utilized by a large number of bird species, particularly in winter.

Literature Cited

Allen, A. W. 1982. Habitat suitability index models: gray squirrel. U. S. Dep. Int., Fish and Wildl. Serv., FWS/OBS-82/10.19, Ft. Collins, Colo. 11pp.

Anderson, S. H., K. Mann, and H. H. Shugart, Jr. 1977. The effect of transmission-line corridors on bird populations. Am. Midl. Nat. 97(1):216–221.

Atkeson, T. D. and A. S. Johnson. 1977. Succession of small mammals on pine plantations in the Georgia Piedmont. Am. Midl. Nat. 101(2):384–392.

Auffenberg, W. and R. Franz. 1982. The status and distribution of the gopher tortoise (*Gopherus polyphemus*). Pages 95–126 *in* R. B. Bury, ed. North American tortoise conservation and biology, U. S. Dep. Int., Fish and Wildl. Serv., Wildl. Res. Rep. 12, Washington, D.C.

Balda, R. P. 1975. Vegetation structure and breeding bird diversity. Pages 59–80 *in* D. R. Smith, tech. coord. Proceedings of the symposium on management of forest and range habitats for nongame birds. U. S. Dep. Agric., For. Ser., Gen. Tech. Rep. WO-1, Washington, D.C.

Berry, R. J. 1971. Conservation aspects of the genetical constitution of populations. Pages 177–206 *in* E. Duffy and A. S. Wall, eds. Scientific management of animal and plant communities for conservation. Blackwell Sci. Publ. Oxford, England.

Campbell, H. W. and S. P. Christman. 1982. The herpetological components of Florida sandhill and sand pine scrub associations. Pages 163–171 *in* N. J. Scott, Jr., ed. Herpetological communities. U. S. Dep. Int., Fish and Wildl. Ser., Res. Rep. 13, Washington, D.C.

Carr, A. F., Jr. 1940. A contribution to the herpetology of Florida. Univ. Fla., Biol. Sci. Ser., Vol. III, No. 1, Gainesville.

Chasko, G. G. and J. E. Gates. 1982. Avian habitat suitability along a transmission-line corridor in an oak-hickory forest region. Wildl. Monogr. 82. 41pp.

Conner, R. N. 1978. Snag management for cavity nesting birds. Pages 120–128 *in* R. M. De Graaf, tech. coord. Proceedings of the workshop management of southern forests for nongame birds. U. S. Dep. Agric., For. Ser., Gen. Tech. Rep. SE-14, Asheville, N.C.

———, J. C. Kroll, and D. L. Kulhavy. 1983. The potential of girdled and 2,4-D injected southern red oaks as woodpecker nesting and foraging sites. So. J. Appl. For. 7:125–128.

Cutright, N. J. 1981. Bird populations in five major west-central Florida vegetation types. Fla. Sci. 44(1):1–13.

Davis, J. H. 1967. General map of the natural vegetation of Florida. Agric. Exp. Sta. Inst. Food and Agric. Sci., Univ. Fla., Gainesville.

Dickson, J. G. 1978. Forest bird communities of the bottomland hardwoods. Pages 66–73 *in* R. M. DeGraaf, tech. coord. Proceedings of the workshop management of southern forests for nongame birds. U. S. Dep. Agric., For. Ser., Gen. Tech. Rep. SE-14, Asheville, N.C.

Fitch, H. S. 1958. Home ranges, territories and seasonal movements of vertebrates. Univ. Kans. Publ., Mus. Nat. Hist. 11:63–326.

Florida Game and Fresh Water Fish Commission. 1976. Cross Florida barge canal restudy report, wildlife study, Volume I. Prepared for Dep. of the Army, Jacksonville Dist., Corps of Engineers, Jacksonville, Fla.

Forman, R. T. T., A. E. Galli, and C. F. Leck. 1976. Forest size and avian diversity in New Jersey woodlots with some land use implications. Oecologia. 26:1–8.

Franklin, I. R. 1980. Evolutionary change in small populations. Pages 135–149 *in* M. E. Soule and B. A. Wilcox, eds. Conservation biology: an evolutionary-ecological perspective. Sinhauer Assoc., Inc., Sunderland, Mass.

Galli, A. E., C. F. Leck, and R. T. T. Forman. 1976. Avian distribution patterns in forest islands of different sizes in central New Jersey. Auk. 93:356–364.

Gates, J. E. and J. A. Mosher. 1981. A functional approach to estimating habitat edge width for birds. Am. Midl. Nat. 105(1):189–192.

Haroldson, K. J. and E. K. Fritzell. 1984. Home ranges, activity, and habitat use by gray foxes in an oak-hickory forest. J. Wildl. Manage. 48(1):222-227.

Harris, L. D. and R. D. Wallace. 1984. Breeding bird species in Florida forest fragments. Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies. 38:in press.

———. 1980. Forest and wildlife dynamics in the Southeast. Trans. N. Am. Wildl. and Nat. Resour. Conf. 45:307–322.

———. 1984. The fragmented forest: island biogeography theory and the preservation of biotic diversity. Univ. Chicago Press, Chicago, Ill. 211 pp.

Johnston, D. W. and E. P. Odum. 1956. Breeding bird populations in relation to plant succession on the Piedmont of Georgia. Ecology. 37:50–62.

Karr, J. R. 1968. Habitat and avian diversity on strip-mined land in east central Illinois. Condor. 70:348–357.

———. 1971. Structure of avian communities in selected Panama canal and Illinois habitats. Ecol. Monogr. 41:207–233.

——— and R. R. Roth. 1971. Vegetation structure and avian diversity in several new world areas. Am. Nat. 105:423–435.

Kendeigh, S. C. 1946. Breeding birds of the beech-maple-hemlock community. Ecology. 27:226–245.

Kirkland, G. L., Jr. 1974. Preliminary sampling of small mammals on clearcut and uncut red spruce stands in West Virginia. Proc. W. Va. Acad. Sci. 46(2):150–154.

Kroodsma, R. L. 1984. Ecological factors associated with degree of edge effect in breeding birds. J. Wildl. Manage. 48(2):418–425.

Landers, J. L. and D. W. Speake. 1980. Management needs of sandhill reptiles in southern Georgia. Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies 34:515–529.

Leopold, A. 1933. Game management. Charles Scribner's and Sons, New York. 481pp.

Lynch, J. F. and R. F. Whitcomb. 1978. Effects of the insularization of the Eastern deciduous forest on avifaunal diversity and turnover. Pages 461–489 *in* A. Marmelstein, ed. Classification, inventory, and analysis of fish and wildlife habitat: proceedings of a national symposium. U. S. Dep. Int., Fish and Wildl. Ser., FWS-78/76, Washington, D.C.

MacArthur, R. H. and J. W. MacArthur. 1961. On bird species diversity. Ecology. 42:594–598.

MacClintock, L., R. F. Whitcomb, and B. L. Whitcomb. 1977. Island biogeography and "habitat islands" of eastern forest II. Evidence for the value of corridors and the minimization of isolation in preservation of biotic diversity. Am. Birds. 31:6–16.

Maehr, D. S. and J. R. Brady. 1984. Food habits of Florida black bears. J. Wildl. Manage. 48(1):230–235.

Martin, T. E. 1980. Diversity and abundance of spring migratory birds using habitat islands on the Great Plains. Condor. 82:430–439.

———. 1981. Limitation in small habitat islands: chance or competition? Auk. 98:715–734.

McElveen, J. D. 1977. The edge effect on a forest bird community in north Florida. Proc. Southeast. Assoc. Fish and Wildl. Agencies. 31:212–215.

Mengak, M. T. and A. R. Tipton. 1978. Comparison of small mammal occurence in 1- and 4-year-old pine plantations. Va. J. Sci. 29(2):65.

Meyers, J. M. and A. S. Johnson. 1978. Bird communities associated with succession and management of loblolly-shortleaf pine forests. Pages 50–65 *in* R. M. DeGraaf, tech. coord. Proceedings of the workshop management of southern forests for nongame birds. U. S. Dep. Agric., For. Ser., Gen. Tech. Rep. SE-14, Asheville, N.C.

Moore, N. W. and M. D. Hooper. 1975. On the number of bird species in British woods. Biol. Conserv. 8:239–250.

Nelson, G. E., Jr. 1952. The birds of Welaka. Q. J. Fla. Acad. Sci. 15(1):21–39.

Noble, R. E. and R. B. Hamilton. 1975. Bird populations in even-aged loblolly pine forests of southeastern Louisiana. Proc. Annu. Conf. Southeast. Assoc. Game and Fish Comm. 29:441–450.

Odum, E. P. 1971. Fundamentals of ecology. W. B. Saunders Co., Philadelphia, Pa. 574pp.

Orr, R. T. 1971. Vertebrate biology. W. B. Saunders Co., Philadelphia, Pa. 544pp.
Roth, R. R. 1976. Spatial heterogeneity and bird species diversity. Ecology. 57(4): 773–782.
Rusterholz, K. A. and R. W. Howe. 1979. Species-area relations of birds on small islands in a Minnesota lake. Evolution. 33:468–477.
Shelford, V. E. 1963. The ecology of North America. Univ. Ill. Press, Urbana, Ill. 610pp.
Shugart, H. H. and D. A. James. 1973. Ecological succession of breeding bird populations in northwestern Arkansas. Auk. 90:62–77.
———, T. M. Smith, J. T. Kitchings, and R. L. Kroodsma. 1978. The relationship of nongame birds to Southern forest types and successional stages. Pages 5–16 *in* R. M. DeGraaf, tech. coord. Proceedings of the workshop management of southern forests for nongame birds. U. S. Dep. Agric., For. Ser., Gen. Tech. Rep. SE-14, Asheville, N.C.
Speake, D. W., J. A. McGincy, and T. R. Colvin. 1978. Ecology and management of the eastern indigo snake in Georgia: a progress report. Pages 64–69 *in* R. R. Odom and L. Landers, eds. Proceedings of the rare and endangered wildlife symposium. Ga. Dep. Nat. Resour., Game and Fish Div., Tech. Bul. WL-4, Athens, Ga.
Strelke, W. K. and J. G. Dickson. 1980. Effect of forest clearcut edge on breeding birds in East Texas. J. Wildl. Manage. 44(3):559–567.
Southwest Florida Water Management District. 1980. Upper Hillsborough flood detention area environmental assessment. Southwest Fla. Water Manage. Dist., Brooksville, Fla. 143pp.
———. 1981. Green Swamp flood detention area environmental assessment. Southwest Fla. Water Manage. Dist., Brooksville, Fla. 106pp.
Tanner, W. F. and L. D. Smith. 1981. Natural environment. Pages 14–73 *in* E. A. Fernald, ed. Atlas of Florida. Fla. State Univ. Found., Inc., Tallahassee, Fla.
Umber, R. W. and L. D. Harris. 1975. Effects of intensive forestry on succession and wildlife in Florida sandhills. Proc. Annu. Conf. Southeast. Assoc. Game and Fish Comm. 28:686–693.
U. S. Army Corps of Engineers. 1980. A habitat evaluation system for water resources planning. Environ. Anal. Branch, Planning Div., Lower Miss. Valley Div., Vicksburg.
Wales, B. A. 1972. Vegetation analysis of north and south edges in a mature oak-hickory forest. Ecol. Monogr. 42:451–471.
Warren, R. C. and G. A. Hurst. 1980. Squirrel densities in pine-hardwood forests and streamside management zones. Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies 34:492–498.
Williamson, P. 1971. Feeding ecology of the red-eyed vireo and associated foliage-gleaning birds. Ecol. Monogr. 41:129–152.
Willson, M. F. 1974. Avian community organization and habitat structure. Ecology. 55:1017–1029.
Zeedyk, W. D. and K. E. Evans. 1975. Silvicultural options and habitat values in deciduous forests. Pages 115–127 *in* D. R. Smith, tech. coord. Proceedings of the symposium on management of forest and range habitats for nongame birds. U. S. Dep. Agric., For. Ser., Gen. Tech. Rep. WO-1, Washington, D.C.