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Management of Bottomland Hardwoods and Deepwater Swamps for Threatened and Endangered Species

by *Richard A. Fischer, Chester O. Martin, Kevin Robertson, William R. Whitworth, Mary G. Harper*

Prepared for Headquarters, U.S. Army Corps of Engineers

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Preface

The work described in this report was authorized by the Strategic Environmental Research and Development Program (SERDP) under the SERDP study, “Regional Guidelines for Managing Threatened and Endangered Species Habitats.” The technical monitor was Dr. Femi Ayorinde, Conservation Program. Mr. Brad Smith is Executive Director, SERDP.

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1 Introduction

Background

Bottomland hardwood forests (BLH) and deepwater swamps are forested wetlands that include stream and river floodplain forests and basin mixed hardwood forests throughout the southeastern United States. The ecology and management of these communities are reviewed here with an emphasis on land uses associated with Department of Defense (DoD) installations. The natural hydrologic processes associated with bottomland hardwoods and deepwater swamps contribute to clean water supplies and fertile soils and influence wildlife populations in the region. These communities also support several threatened, endangered and sensitive species (TES). Since bottomland ecosystems have been greatly reduced in extent and altered in function, examples of these communities on DoD lands merit careful management.

This report constitutes one of several in a series that is the product of an interlaboratory effort between the U.S. Army Engineer Waterways Experiment Station (WES) and U.S. Army Construction Engineering Research Laboratories (USACERL) to generate habitat-based management strategies for TES on DoD lands in the southeastern United States (*Strategic Environmental Research and Development Program [SERDP] work unit "Regional Guidelines for Managing T&E Species Habitats* (Martin et al. 1996). This effort is directed at developing strategies to manage TES and their habitats on a plant community basis using methods that apply to multiple species and military training lands across the southeastern United States. Any increase in understanding of the habitat requirements of listed TES will assist training and natural resource personnel in complying with the Endangered Species Act (ESA), while avoiding restrictions on the military mission.

Objectives

The objectives of this research were to compile information, identify gaps in knowledge, and stimulate future research efforts on the potential positive and negative effects of landscape planning, silviculture, military training, and other resource-based activities on BLH and deepwater swamps that serve as high-quality habitat for TES on military lands in the southeastern United States.

A range of management options was considered for areas that trainers and resource managers recognize as potential endangered species habitat. These options are not intended to constrain military training. Rather, management options were developed within the context of training requirements and should be considered only to the extent they are compatible with training. Many of the more restrictive land-use options identified in this report apply to lands (i.e., forested wetlands) already protected due to their sensitive nature. Training will continue to be the primary land-use concern, with training-land decisions being made on a daily basis with whatever information is available at the time. Flexibility in the management options identified in this and related reports will enable land managers to make more informed decisions and effectively support the training mission. Moreover, while management options within this report are not intended to be applied across entire DoD installations, they are presented as potential tools consistent with an ecosystem approach designed to provide healthy, functional communities.

This SERDP report, in particular, was undertaken to reduce duplication of effort in conservation of TES in BLH and deepwater swamp habitats. This review of information may be used to improve the ecological and economic effectiveness of TES habitat management. By understanding the ecological requirements of TES and the environmental resilience or sensitivity of BLH and deepwater swamps, installations may improve TES management and land-use decisions.

Approach

To identify potential impacts to BLH and management options to mitigate impacts, researchers reviewed the available literature and conducted interviews with community ecologists throughout the southeastern United States, with an emphasis on interviewing those people who have been involved in TES and plant community survey work on military installations. Site visits were made to several military installations. Potential impacts were also discussed with military natural resources personnel, botanists, community ecologists, and military contractors such as The Nature Conservancy (TNC) and associated state Natural Heritage Program (NHP) staff. A list of experts contacted is included in Appendix A. Information also was acquired from installation TES survey reports in which impacts and management were addressed. Land Condition Trend Analysis (LCTA) reports, Land Rehabilitation and Maintenance (LRAM) data, and academic and Federal agency literature on logging and recreational impacts to plant communities were also used.

Scope

Within the context of the larger DoD mission, TES populations can be maintained through the following framework: (a) identify mission requirements, (b) identify TES requirements, (c) identify ideal compromises for meeting both TES and mission requirements, and (d) pursue these compromises and develop realistic, workable approaches. The last step should be executed through professional management of TES populations, at the installation level, to reduce restrictions on the military mission. This document contributes to the total TES and land-management process. It provides information to assist in identifying the needs of TES (step b), and should assist in identifying options for compromise as well (step c).

This series of management reports (e.g., Trame and Harper 1997, Harper et al. 1997, Trame and Tazik 1995) focuses on plant communities because they provide habitat for numerous species. By managing at the community or ecosystem level, DoD has the opportunity to conserve multiple TES simultaneously. Plant communities are less ambiguous entities than complete ecosystems and have been variously described and cataloged for many decades by ecologists and biogeographers. They provide a useful basis for managing the natural systems that support military training and other land uses.

Bottomland hardwood communities support multiple uses, including DoD training and testing; TES conservation; and forest commodities (e.g., timber) production. This document provides a review of wetland ecology and recommended management practices for BLH and deepwater swamps. It is intended to provide current information for management of BLH on military installations that is compatible with the military training mission. Where feasible, management recommendations mimic natural disturbance patterns and provide suitable habitat for the diversity of species that inhabit the community, with an emphasis on TES. Because the focus of this report is on TES, the wealth of currently available literature addressing BLH management techniques for game species (primarily waterfowl) is not included.

Mode of Technology Transfer

This report is to be used by DoD natural resource policy makers, installation land managers, and the natural resources research community, in conjunction with associated documents produced under this SERDP work unit to (a) develop ecosystem-compatible approaches in describing natural communities and TES habitat within the context of military land management, (b) evaluate military-related effects on those communities, (c) develop community-based strategies for supporting both military land-use and TES habitat management, and (d) develop management solutions for military impacts to natural communities when management for TES habitat is a priority.

This report is available on the U.S. Army Engineer Research and Development Center (USAERDC) web page at http://www.wes.army.mil/el.

2 Ecological Description

BLH and deepwater swamps are forested wetlands[1] that include stream and river floodplain forests and mixed hardwood forests in basins of the southeastern United States. Bottomland hardwoods occupy the majority of natural riparian areas in the United States (Huffman and Forsythe 1981, Mitsch and Gosselink 1993); they are dominated by a variety of woody plant species adapted to survival in an environment where soils within the root zone may be either inundated or saturated during various times of the growing season (Sharitz and Mitsch 1993). These floodplain forests are characterized by high biomass, relatively high stem density of adult trees, and large individual trees forming a high canopy (Figure 1; Brinson 1990). On persistently inundated sites, BLH communities generally have low stem density (Sharitz and Mitsch 1993). Bottomland hardwoods were classified as Palustrine Wetlands in the National Wetlands Classification System and Inventory (Cowardin et al. 1979), as Riverine Wetlands by Brinson (1993), and are a type of riparian community (e.g., Turner, Forsythe, and Craig 1981; Taylor, Cardamone, and Mitsch 1990, Sharitz and Mitsch 1993).

Deepwater swamps are freshwater systems that occur throughout much of the range of southern BLH in depressions (e.g., abandoned river channels, elongated sloughs) that are inundated during most or all of the year (Figure 2). These sites are typically dominated by baldcypress (*Taxodium distichum)*, pondcypress (*T. ascendens)*, water tupelo (*Nyssa aquatica)*, and swamp tupelo (*N. sylvatica)* (Conner and Buford 1998). (Penfound 1952, Sharitz and Mitsch 1993). Deepwater swamps typically are highly productive because they usually are found along the floodplains of rivers having soils with ample nutrients (Conner and Buford 1998).

Eighty-six percent of forested wetlands in the southeastern United States are dominated by hardwood tree species (Tansey and Cost 1990), with the remainder dominated primarily by baldcypress. Most southern BLH occur in the Coastal Plain along small drainageways, but others occur adjacent to major rivers or in

---

[1] Wetlands are transitional habits between terrestrial and aquatic systems. They are defined as those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions (wetlands generally include swamps, marshes, bogs, and similar areas) (33 CFR328.3(b) (1984)).

Figure 1. Typical BLH forest, Cache River, Arkansas

areas with a clay hardpan (Tansey and Cost 1990). Southern BLH provide numerous ecosystem functions (Wilkinson, Schneller-McDonald, and Auble 1987) and support a diverse plant and animal community that includes many state and Federal threatened, endangered, and candidate species (TES)[1] (Ernst and Brown 1989). Kusler (1977) suggested that although only 3.5 percent of United States land area is wetland, approximately 35 percent of all rare, threatened, and endangered wildlife species depend on wetlands for survival. Bottomland hardwood ecosystems were once widespread and abundant throughout the southeastern United States. However, their distribution has been greatly reduced to the point where BLH ecosystems are now considered threatened (Noss, LaRoe, and Scott 1995).

Classification Systems

Southeastern BLH are generally similar in overall species composition throughout their geographic range (Sharitz and Mitsch 1993). Pronounced

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[1] The acronym "TES" will be used instead of "T&E Species" in this report to conform to standard DoD terminology. We include "Candidate Species" (former C1 species), defined as those plant and animal species that, in the opinion of the U.S. Fish and Wildlife Service (USFWS) or National Marine Fisheries Service, may qualify for listing as threatened or endangered pursuant to the Endangered Species Act; and "Species of Concern" (Former C2 species).

Figure 2. Deepwater swamp, southern Alabama

variations in species abundance and composition occur on a local scale, however, due to differences in hydrology and to the complex and dynamic nature of the floodplain environment (Sharitz and Mitsch 1993). Classification systems for forested wetlands, including BLH, have been developed in an attempt to describe this variation. Several of these systems classify BLH throughout the geographic range of the community. Others, such as the state classification systems developed for North Carolina (Schafale and Weakley 1990) and Florida (Ewel 1990), cover only a limited geographic area. Classification systems covering limited areas often work well on a local scale, but they have limited regional applicability (Kellison et al. 1998). Selected classification systems are discussed below.

In 1979 the USFWS published a national wetland classification system to be used in a new inventory of wetlands and deepwater habitats across the United States, to furnish units for mapping, and to provide uniformity in concepts and terminology (Cowardin et al. 1979). In the Cowardin et al. (1979) hierarchial classification, wetlands are defined based on the presence of hydrophytic plants, frequency of flooding, and hydric soils. The broadest grouping is the *system*, defined as "a complex of wetlands and deepwater habitats that share the influence of similar hydrologic, geomorphologic, chemical, or biological factors." Systems are marine, estuarine, riverine, lacustrine, and palustrine. *Subsystems* further define systems and consider primarily water depth, flow, and substrate gradient. Familiar subsystems include tidal, intermittent, limnetic, and littoral. Subsystems are composed of *classes*, which describe the general appearance of the system in terms of either the dominant vegetation or substrate. Vegetation is the basis for class description if the vegetative cover exceeds 30 percent. Substrate is the basis if vegetative cover is less than 30 percent. Classes are divided into *subclasses*, *dominance types*, and *modifiers*. These categories are further described in Cowardin et al. (1979) and provide progressively more detail on water regime and the dominant plants present.

Perhaps the most inclusive and consistent classification of southeastern floodplain forests is the zonation of floodplain communities established by Larson et al. (1981) (Table 1). These zones are defined by their location on a hydrologic gradient of soil saturation or inundation. Not all wetland ecologists have adopted this system because of its oversimplification of such a complex system and the variable nature of plant species occurrences in each zone (Sharitz and Mitsch 1993). Nevertheless, this system is useful in defining fairly distinct vegetation zones. Characteristic plant species occupy each zone based on maximum tolerance to soil moisture and hydrologic regime (Larson et al. 1981), although several tree species occur in more than one zone (Table 2). Zone I is an aquatic habitat with no woody vegetation and is therefore not considered here. Zones II and III are usually considered wetlands under most definitions, and inundation is nearly permanent, or at least present for a majority of the growing season. Zone II contains deepwater swamps typically dominated by baldcypress and/or water tupelo. Zone III also has cypress and tupelo but has a much more diverse overstory. Zone IV includes backwater areas and flats that are seasonally flooded or saturated, especially during the early growing season. Overstory species richness is even higher than in Zone III and may include several oak (*Quercus* spp.) species. Zone V occupies the highest portions of the active floodplain, and flooding occurs only for a very short duration during the growing season; the water table typically lies below the surface.

There is debate over the inclusion of Zone V in wetlands definitions, and Zone VI (higher areas with rare soil inundation or saturation) is typically not considered wetland. These zones often occur in sequence from water's edge to upland, but this should not always be assumed; floodplains sometimes do not rise uniformly from the river but are crossed by levees, meander scrolls, sloughs, and oxbow lakes that allow any of the different zones to occur in any floodplain location (Taylor, Cardamone, and Mitsch 1990). This classification system is beneficial because it recognizes hydrology and the related limitation of oxygen as the primary determinant of floodplain species composition (Wharton et al. 1982, Sharitz and Mitsch 1993), while avoiding the complexity of small-scale variation in relative species abundances characteristic of floodplain communities. Also, it was designed to help determine appropriate management practices for floodplain forest types. Classification systems for deepwater swamps are discussed in Conner and Buford (1998).

Table 1
Classification of BLH Forests[1]

Zone	Water Regime	Flooding Frequency	Flooding Duration	Dominant Canopy Tree Species[2]	Other Common Vegetation Species[2]	Comments
Zone II	Intermittently exposed	Near 100 percent	Continuous except during extreme drought periods	Baldcypress, water tupelo	Buttonbush, water elm, pondcypress, black gum, swamp privet, sweet bay, red bay	Surface water continuously present; vegetation is in saturated or flooded soil for entire growing season.
Zone III	Semi-permanently flooded	51 to 100 percent	>25 percent of the growing season	Black willow, silver maple, cottonwood, overcup oak, water hickory, red maple, green ash, river birch	Cabbage palmetto; several species of ash, maple, and birch, persimmon	Surface water or saturated soil persists for a major portion of growing season.
Zone IV	Seasonally flooded	51 to 100 percent	12.5 to 25 percent of the growing season	Laurel oak, willow oak, water oak, green ash, American elm, box elder, ironwood, sycamore, sweetgum, cottonwood	Willow oak, Nuttall oak, pin oak, other oaks, sugarberry, sycamore, box elder	Surface water or saturated soil present for extended periods, especially early in growing season.
Zone V	Temporarily flooded	11 to 50 percent	2 to 12.5 percent of the growing season	Swamp chestnut oak, cherrybark oak, white oak, hickory, spruce pine	Loblolly pine, Laurel oak, sweetgum, sweet bay, eastern red cedar, American holly, black cherry	Surface water or saturated soil for brief periods during growing season; water usually well below soil surface for most of season.
Zone VI	Intermittently flooded	1 to 10 percent	<2 percent of the growing season	Several species of oak, ash, and hickory	Upland tree species intolerant to inundation or soil saturation.	Soil inundation or saturation rarely occurs, but surface water may be present for variable periods without detectable seasonal periodicity.

[1] Source: after Larson et al. (1981)
[2] Scientific names are in Table 2

Range

Current distribution

Bottomland hardwood and deepwater swamps can be found within the Gulf Coastal Plain and Atlantic Coastal Plain from Virginia southward (Figure 3). The geographic distribution of these communities is as great or greater than most other southeastern forested communities (Sharitz and Mitsch 1993). According to the most recent National Wetlands Inventory (NWI) data, Georgia has the greatest area of palustrine forested wetlands among the southern states, followed

Table 2
Floodplain Forest Woody Vegetation Included in Ecological Zones[1]

	Zone				
Species	**II**	**III**	**IV**	**V**	**VI**
Acer negundo (boxelder)		X	X	X	X
Acer rubrum (red maple)		X	X	X	X
Alnus serrulata (common alder)	X	X	X		
Amorpha fruticosa (dull-leaf indigo)		X	X		
Asimina parviflora (dwarf paw paw)				X	X
Asimina triloba (paw paw)				X	
Baccharis glomeruliflora (groundsel)			X	X	X
Betula nigra (river birch)		X	X		
Bumelia reclinata (bumelia)			X	X	
Callicarpa americana (beautyberry)				X	X
Carpinus caroliniana (ironwood)			X	X	
Carya aquatica (water hickory)		X	X		
Carya cordiformis (bitternut hickory)				X	
Carya glabra (pignut hickory)				X	X
Carya illinoiensis (pecan)			X	X	X
Carya ovata (shagbark hickory)				X	X
Celtis laevigata (sugarberry)			X	X	X
Celtis occidentalis (hackberry)			X	X	X
Cephalanthus occidentalis (buttonbush)	X	X			
Cornus florida (flowering dogwood)				X	X
Cornus foemina (stiff dogwood)		X	X		
Craetegus marshallii (parsley haw)			X	X	
Craetegus viridis (green hawthorn)			X		
Diospyros virginiana (persimmon)		X	X	X	X
Euonymus americanus (strawberry bush)				X	X
Fagus grandifolia (American beech)				X	X
Forestiera acuminata (swamp privet)	X	X			
Fraxinus caroliniana (water ash)	X	X	X		
Fraxinus pennsylvanica (green ash)		X	X		
Fraxinus profunda (pumpkin ash)	X				
Gleditsia aquatica (water locust)		X	X		

[1] After Larson et al. (1981) and Wharton et al. (1982) with additions by Sharitz and Mitsch (1993).

(Sheet 1 of 3)

Table 2 (Continued)					
	Zone				
Species	II	III	IV	V	VI
Gleditsia triacanthos (honey locust)			X	X	X
Ilex decidua (possum haw)			X	X	X
Ilex opaca (American holly)				X	X
Itea virginica (Virginia willow)	X	X			
Juniperus virginiana (eastern red cedar)				X	X
Leucothoe axillaris (dog-hobble)	X	X			
Ligustrum sinense (privet)			X	X	X
Liquidambar styraciflua (sweetgum)			X	X	X
Lyonia lucida (fetterbush)	*X*	*X*	*X*		
Magnolia grandiflora (southern magnolia)				*X*	*X*
Magnolia virginiana (sweet bay)	*X*	*X*	*X*		
Morus rubra (red mulberry)			*X*	*X*	*X*
Myrica cerifera (wax myrtle)			*X*	*X*	*X*
Nyssa aquatica (water tupelo)	*X*	*X*			
Nyssa ogeche (Ogeechee tupelo)	*X*	*X*			
Nyssa sylvatica (black gum)				*X*	*X*
Nyssa aquatica (swamp tupelo)	*X*	*X*			
Ostrya virginiana (eastern hop-hornbeam)				*X*	*X*
Persea borbonia (red bay)	*X*	*X*	*X*		
Pinus glabra (spruce pine)			*X*	*X*	
Pinus serotina (pond pine)	*X*	*X*	*X*		
Pinus taeda (loblolly pine)				*X*	*X*
Planera aquatica (water elm)	*X*	*X*			
Platanus occidentalis (sycamore)			*X*	*X*	*X*
Populus deltoides (eastern cottonwood)		*X*	*X*		
Populus heterophylla (swamp cottonwood)		*X*	*X*		
Prunus serotina (black cherry)				*X*	*X*
Ptelea trifoliata (water ash)				*X*	*X*
Quercus alba (white oak)				*X*	*X*
Quercus laurifolia (laurel oak)		*X*	*X*	*X*	
Quercus lyrata (overcup oak)		*X*	*X*		
Quercus michauxii (swamp chestnut oak)			*X*	*X*	
Quercus nigra (water oak)			*X*	*X*	*X*
					(Sheet 2 of 3)

Table 2 (Concluded)

	Zone				
Species	**II**	**III**	**IV**	**V**	**VI**
Quercus nuttallii (Nuttall oak)			X		
Quercus pagodaefolia (cherrybark oak)				X	X
Quercus phellos (willow oak)			X	X	X
Quercus shumardii (Shumard's oak)				X	X
Quercus virginiana (live oak)			X	X	X
Rhododendron canescens (hoary azalea)			X	X	
Sabal minor (dwarf palmetto)			X	X	X
Sabal palmetto (cabbage palm)			X	X	X
Salix caroliniana (swamp willow)	X	X			
Salix nigra (black willow)	X	X			
Sambucus canadensis (elderberry)			X	X	X
Sassafras albidum (sassafras)				X	X
Serenoa repens (saw palmetto)				X	X
Styrax americanum (American snowbell)		X			
Taxodium ascendens (pondcypress)	X				
Taxodium distichum (baldcypress)	X	X			
Tilia americana (basswood)				X	X
Ulmus alata (winged elm)			X	X	X
Ulmus americana (American elm)			X	X	X
Ulmus rubra (slippery elm)				X	X
Viburnum obovatum (black haw)			X		

(Sheet 3 of 3)

closely by Florida. Shepard et al. (1998) reported that over 75 percent of wetlands in the South are wooded. For example, the 1992 National Resources Inventory (NRI) indicated that wooded wetlands in North and South Carolinas made up more than 90 percent of the palustrine wetlands in both of these states and composed more than 80 percent of wetlands in Georgia, Arkansas, Alabama, and Mississippi. The NRI is an inventory of natural resource conditions on non-Federal lands conducted by the Natural Resources Conservation Service. Although not mapped by the NRI, substantial Federal holdings of forested wetlands occur in the southeast (Shepard et al. 1998).

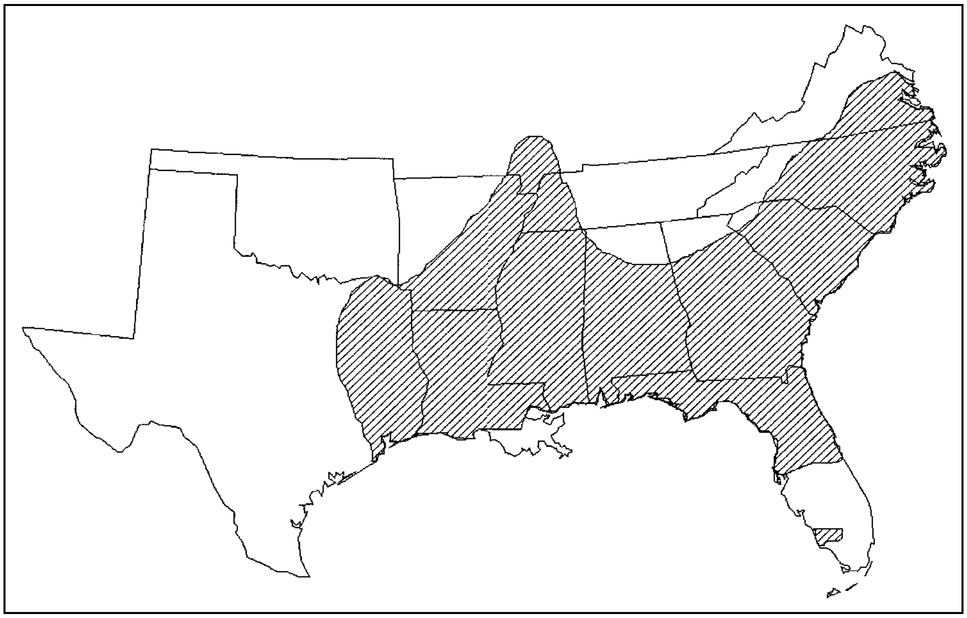

Figure 3. Approximate distribution of BLH forests in the southeastern United States

Distribution on military installations

Bottomland hardwoods and deepwater swamp communities are documented on at least 29 DoD installations in the southeastern United States (Table 3). Although the exact total acreage of these communities is uncertain, their occurrence on installations appears to represent a significant source of biological diversity and forest productivity. A standard terminology does not presently exist for classifying wooded wetlands. Therefore, Table 3 includes descriptions from installation documents and other sources.

Environmental Characteristics

Geomorphology

The topography of the floodplain forest environment is formed by the transportation, erosion, and deposition of sediment by the meandering river channel. The deposition of coarse sediment adjacent to the river results in the formation of a natural levee, usually steeply sloping toward the river and gradually into the floodplain. Where streamflow velocity decreases on the inside of river bends, sediment is deposited to form point bars. The river channel migrates as the outside bank is eroded and the point bar is aggraded (Leopold and Wolman 1950). "Meander scrolls" or ridges are left on the floodplain where prior levees existed. Generally, river bends are eventually "cut off" or bypassed by the formation of a more direct channel (Hicken 1974), leaving segments of abandoned channel called "oxbow lakes" (reviewed by Sharitz and Mitsch 1993). These are important to the water storage capacity of the floodplain

Table 3
Occurrence of BLH and Deepwater Swamps on Military Installations in the Southeastern United States

State	Branch	Installation	Name in Document	Zone(s)	Reference
AL	Army	Anniston Army Depot and Coosa River Annex	Forested Palustrine Wetlands	III - V	Godwin and Bailey (1994)
		Fort Rucker	Infrequently flooded Mesic Hardwood Forests	V	Mount and Diamond (1992)
		Redstone Arsenal	Mixed Bottomland Oak Forest	V	Jaryl Hilton, Personal Communication, 1994
		Fort McClellan, Main Post	Sweetgum-Mixed Bottomland Oak Forest	IV-V	Alabama Natural Heritage Program (1994a)
		Fort McClellan, Pelham Range	Sweetgum-Mixed Bottomland Oak Forest	IV-V	Alabama Natural Heritage Program (1994b)
AR	Air Force	Little Rock Air Force Base (AFB)	Bottomland Hardwoods		Woolpert, Inc. (1995)
FL	Air Force	Tyndall AFB	Floodplain Swamp	II	Florida Natural Areas Inventory (FNAI) (1994a)
		Hurlburt Field	Forested Wetlands		Labat-Anderson Inc. (1994)
		Avon Park AFB	Floodplain Swamp (potentially)		The National Conservancy (TNC) 1994
		Eglin AFB	Bottomland Forest Floodplain Forest		FNAI (1994b)
	Army	Camp Blanding	Bottomland Hardwood Forest Riparian area Swamp, bay, and riparian area Hardwood Swamp	IV-V II II-III IV	R. Brozka, Personal Communication, 1994
	Navy	Naval Air Station (NAS) Jacksonville	Wetland Bottomland Swamp	II-V	Anon. (1988a), Environmental Services and Permitting, Inc. (1990)
		NAS Cecil Field	Wetland Bottomland Swamp	II-V	Anon. (1988b), Environmental Services and Permitting, Inc. (1990)
		NAS Pensacola	Mixed Forested Wetlands	II	Anon. (1988c), FNAI (1988)
		NAS Whiting Field	Floodplain Swamps	II	Anon. (1991)
GA	Army	Fort Gibson	Floodplain Forest	III-IV	Moore and Giannasi (1992)
		Fort Stewart	Bald Cypress-Water Tupelo Swamp, Coastal Plain Small Stream Swamp Forest	II II-III	The Nature Conservancy (TNC) (1995)
		Fort Benning	Bottomland Hardwood Forest	III-IV	Gulf Engineers & Consultants and Geo-Marine, Inc. (1994)
	Marine Corps	Marine Corps Logistics Base (MCLB)	Blackwater Stream Riparian Forest	IV	Georgia Department of Natural Resources (DNR) (1994)

(Continued)

Table 3 (Concluded)					
State	**Branch**	**Installation**	**Name in Document**	**Zone(s)**	**Reference**
LA	Air Force	Barksdale AFB	Batture Forest	III	Nelwyn McInnis, Personal Communication, 1994
	Army	Fort Polk	Riparian Forest	III-V	R. Stewart, Personal Communication, 1995
		Camp Villerie	Species list only	III-IV	TNC (1993)
MS	Army	Camp McCain	Swamp Chestnut Oak-Cherrybark Oak Bottomland Forest	V	Wieland (1994)
		Camp Shelby	Mesic-hydric Forest	V	Wieland (1994)
NC	Army	Fort Bragg and Camp Mackall	Coastal Plain Bottomland Hardwoods, Blackwater Coastal Plain Levee Forest, Coastal Plain Small Stream Swamp	IV III-IV III-V	Russo et al. (1993)
	Marine Corps	Cherry Point Marine Corps Air Station	Coastal Plain Small Stream Swamp, Blackwater	II	LeBlond, Fussell, and Braswel (1994a)
		Camp Lejeune	Coastal Plain Small Stream Swamp, Blackwater Cypress-gum Swamp	II-IV II	LeBlond, Fussell, and Braswel (1994b), LeBlond, Fussell, and Braswel (1994c)
SC	Army	Fort Jackson	Bottomland Hardwood Forest		B. Pittman, Personal Communication, 1995
	Navy	Naval Weapons Station	Forested Wetlands	IV-V	Anon. (1989)
VA	Marine Corps	Quantico Marine Corps Base	Deciduous Forested Wetlands, Deciduous Scrub/Shrub Wetlands	II-IV II-IV	Fish and Wildlife Management Plan, Marine Corps Base, Quantico (1999)

(Brinson 1990). Sedimentation on the floodplain may be categorized as lateral or vertical. Lateral deposition occurs on point bars as the river meanders. Vertical deposition is that occurring on the floodplain, generally during floods (Leopold and Wolman 1950).

Hydrology

Processes in BLH are controlled by flood regime; the physical processes that drive productivity of these wetlands center around hydrological events upstream and in the watershed, and the subsequent groundwater levels (Fredrickson and Reid 1990, Sharitz and Mitsch 1993). O'Neil, Pullen, and Schroaeder (1991) stated, "Geomorphic and hydrologic actions are the primary driving forces that shape the character of a bottomland forest system through erosional and depositional actions of water moving through the floodplain..." In a particular location within the floodplain, the hydroperiod is the most important regulator of plant community dynamics and species composition (Lugo, Brinson, and Brown 1990), primarily because of its influence on the availability of oxygen to plants (Sharitz and Mitsch 1993) and microbes affecting chemical cycling and nutrient availability (Brinson 1990). The hydroperiod, or flooding regime, refers to the timing, frequency, depth, season, and duration of flood events. The hydroperiod is influenced by climate, topography, channel slope, soil characteristics (Gosselink et al. 1990), groundwater fluctuations (Patrick 1981) regional geology, and catchment size (the size of the watershed) (Lugo, Brinson, and Brown 1990). Catchment size is the most important determinant of duration and depth of floods, with larger catchment sizes resulting in deeper and longer floods, such as those of large rivers in the southern Coastal Plain (Brinson 1990). Soil depth and type are also critical, since deeper, more weathered soils provide greater water storage (Brinson 1990). Precipitation, in the form of rainfall, provides the basic water input to the hydrologic system (Williams 1998). Rainfall occurs primarily as winter frontal storms and summer convective thunderstorms; tropical cyclones often contribute large amounts of rainfall to coastal areas. Evaporation and transpiration are second to precipitation in importance to the water balance equation (Williams 1998).

The hydroperiod in the floodplain environment varies greatly along an elevational gradient, from nearly permanently inundated areas to those flooded once every few years. Therefore, hydroperiod characteristics and their associated plant species may be separated into zones roughly corresponding to an elevational sequence (Larson et al. 1981; Table 1). Because of low topographic relief, a very small change in elevation over a short distance may create very different hydrologic conditions, potentially resulting in different soils and plant communities (Sharitz and Mitsch 1993). The interspersion of wet and dry areas causes the vegetation community to vary on a small spatial scale because different plants have varying tolerances to flooding conditions and anaerobic soil conditions (O'Neil, Pullen, and Schroeder 1991). The differing hydrologic regimes in BLH produce a high interspersion of wet and dry areas that are very important for providing high quality wildlife habitat. These conditions can provide a greater variety of food and cover conditions that can support a more diverse animal community. This interspersion also may facilitate reproduction in species with limited mobility, such as amphibians (O'Neil, Pullen, and Schroeder 1991).

Another important characteristic of floodplain hydrology is the pattern and velocity of water movement, particularly during floods (Hupp and Osterkamp 1985). The velocity of water varies among rivers, along the river valley, and among geomorphic features of the river and floodplain, in relation to such variables as catchment size, slope, bank geometry, river planform, and floodplain topography (Sharitz and Mitsch 1993). The greatest kinetic energy and soil shear stress of floodwaters is near the main water column and on the outside banks of river bends (Hooke 1974). Since different plant species suffer different amounts of mechanical damage from water flow and have varying morphological and reproductive responses to that damage, water velocity has a large influence on plant population dynamics and community structure (Everitt 1968, Irvine and West 1979, Harris 1986).

Soils and nutrients

In general, southeastern floodplain forests are supported by Histosols[1] where soil is developed and Entisols[2] where sediment has been recently deposited (Boul, Hole, and McCracken 1989). Alluvial sediments may range from 5 to 80 m thick (Mitsch and Gosselink 1986). The coarsest grained soils are found on point bars and levees and finer soils are found farther from the floodplain (Hooke 1974, Sharitz and Mitsch 1993). Rates of deposition generally decrease with distance from the river channel. The texture of sediment as well as the amount deposited varies in relation to hydrodynamics and existing topographic features. The interaction of soil texture and hydroperiod influences properties of the soil that are important to plant communities, including availability of oxygen and rate of water table drawdown following floods (Sharitz and Mitsch 1993).

Southeastern floodplain forests are generally nutrient rich because of their dynamic nutrient cycling caused by changing hydrology (Brinson 1990) and the import of nutrients with deposited sediment (Sharitz and Mitsch 1993). They are characterized by open nutrient cycles with large inputs and outputs from frequent flooding (Sharitz and Mitsch 1993). Floodplain soils tend to have high nitrogen because of the large amount of organic matter and high phosphorous in the clay-rich soils (Patrick 1981).

Forested wetlands are high in soil organic content and have relatively high biomass, making them important in the storage of atmospheric carbon through photosynthesis (Lugo, Brinson, and Brown 1990). Reduction of chemicals by bacteria inhabiting these wetlands closes the biogeochemical cycles of nitrogen, sulfur, oxygen, and carbon (Lugo, Brinson, and Brown 1990).

Flooding patterns are linked to soil characteristics such as aeration, water holding capacity, and nutrient exchange dynamics. Oxygen levels in the soil are controlled by the rate at which excess water drains from the surface through the soil profile, which is related to soil characteristics such as clay content and distribution. Soil aeration is a major determinant of community composition and species distributions because it affects oxygen, water, and mineral absorption by roots (Harms 1973). In general, an air-filled volume of 15 to 20 percent of the total soil volume is needed to support a diverse bottomland community (Patrick 1981).

Fire regime

Few studies have addressed the influence of fire in riparian areas, but it is generally agreed that fires can be damaging to hardwood communities. Fire directly removes vegetation from the watershed and may indirectly affect

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[1] Soils that belong to the Histosol order have a very high content of organic carbon (more than one-half of the soil's thickness is organic) in the upper 32 in. of soil.

[2] A soil that reflects no major set of soil-forming processes belongs to the Entisol order. Entisols are able to support any vegetation and occur in any climate.

riparian and aquatic ecosystems by changing a watershed's hydrological and erosional characteristics (Hall 1988). In the southeastern United States, the landscape was historically dominated by mixed woodland and conifers with an interspersion of BLH that were too wet for the fires to encroach upon. However, fires can and do occur in BLH during drought, especially in the drier portions of the community. Frequent fires are more likely to occur as a result of training activities on military installations. Wetter areas of BLH usually do not have adequate understory or litter to carry fire unless extensive clearcutting has left a large amount of slash. When fires do occur in BLH, many of the tree species can be killed because most have thin bark; trees that survive fire may rot (Wright and Bailey 1982).

Biological Composition

The dominant trees found in natural BLH communities (Table 1) are a reflection of several variables, including the depth of water and the duration and timing of flood events. For example, deepwater swamps and the wetter portions of floodplain forests are usually dominated by baldcypress and/or water tupelo. Semipermanently flooded portions of the floodplain are typically dominated by black willow (*Salix nigra*), eastern cottonwood (*Populus deltoides*), silver maple (*Acer saccharinum*), overcup oak (*Q. lyrata*), water hickory (*Carya aquatica*), red maple (*A. rubrum*), green ash (*Fraxinus pennsylvanica*), and river birch (*Betula nigra*). Less mesic portions (seasonally to intermittently flooded) may be dominated by laurel oak (*Q. laurifolia*), white oak (*Q. alba*), American elm (*Ulmus americana*), willow oak (*Q. phellos*), water oak (*Q. nigra*), sweetgum (*Liquidambar styraciflua*), sycamore (*Platanus occidentalis*), sugarberry (*Celtis laevigata*), ironwood (*Carpinus caroliniana*), boxelder (*A. negundo*), and hickory.

The species composition and relative abundance of species in various communities within forested wetlands are extremely variable on a small spatial scale. This variation is due to the dynamic and spatially heterogeneous nature of the floodplain's environmental factors, including flooding, changes in geomorphology, and occurrence of tree-fall gaps (Sharitz and Mitsch 1993). Most BLH forests are marked by low density of shrubs and understory plants, particularly in wetter areas (Brinson 1990, Sharitz and Mitsch 1993). The exception is along the river channel, where shrub density is high due to increased light (Brinson 1990). Light limitation due to the thick canopy (Menges and Waller 1983), and oxygen limitation due to flooding, have been cited as causes of low plant density in the understory (Brinson 1990). Species composition and plant density are further complicated by the history of selective timber removal in many forests, as well as seemingly random circumstances, such as high seedling establishment associated with a particular hydrologic event (Brinson 1990). Wharton et al. (1982) and Sharitz and Mitsch (1993) agree that oxygen availability or the "anaerobic gradient" is the most important factor determining species composition and distribution.

If a floodplain is wide enough or opens into an adjacent low-lying area, alluvial bottomland communities can be closely associated with basin wetlands ecosystems (Brinson 1990). The gradient to surrounding uplands is typically to mesic hardwood forests (Brinson 1990). Küchler (1964) described these as oak-hickory forests in the Atlantic Coastal Plain and oak-hickory-pine forests in the Gulf Coastal Plain (Sharitz and Mitsch 1993). With increasing proximity to the Atlantic Ocean or the Gulf of Mexico, associated plant communities often are herbaceous marshes, which may vary along salinity gradients (Sharitz and Mitsch 1993). Bayheads and white cedar (*Chamaecyparis thyoides)* can also occur immediately adjacent to BLH, which in turn can grade into flatwoods and pocosins (Christensen 1988).

Successional Relationships

Succession of riparian plant communities is integrally tied to the associated stream dynamics. Because of the high frequency and spatial heterogeneity of stream disturbances in the floodplain environment, true succession is often difficult to recognize since the pattern of seral stages is not predictable (Sharitz and Mitsch 1993). Moreover, it is difficult to completely define succession in BLH, and few quantitative studies are available. The process initiating and driving primary succession is sediment deposition (Nanson and Beach 1977, McBride and Strahan 1983, Jones et al. 1994). It is the sequence of floods and shifting sediments that create new surfaces and deliver the seeds of colonizing species (Davis et al. 1996). Plant species in BLH differ in their ability to tolerate the constantly shifting influences of stream migration, soil erosion, and deposition (Sharitz and Mitsch 1993). The ability of plants to colonize under these changing conditions also differs among species. Timing of flooding affects seed germination and establishment. Successional processes in riparian forests and deepwater swamps are both allogenic (caused by abiotic factors such as flooding) and autogenic (caused by biological factors such as competition for light).

Though the species composition of observed primary successional sequences varies considerably among floodplain environments, common dominant trees of the first sere on point bars include eastern cottonwood, black willow, river birch, and silver maple (Sharitz and Mitsch 1993). Also occurring in early stages on poorly drained sites are sycamore, red maple, green ash, American elm, winged elm (*U. alata*), sweetgum, sugarberry, and hackberry (*C. occidentalis*). Later, successional stages in areas with the shortest hydroperiod are dominated by oaks and hickories; baldcypress and water tupelo dominate in mature stands with long hydroperiods (Sharitz and Mitsch 1993). Climax communities are rare, except where hydroperiods remain very stable, because of the dynamic nature of the ecosystem (Sharitz and Mitsch 1993). Natural and anthropogenic disturbances, including high floods, wind-throw of trees, drought, logging, and climatic or anthropogenic changes in hydrology, influence the successive habitation by plant species and may be considered to initiate secondary succession in some cases (Kangas 1990).

In deepwater swamps, logging of baldcypress has resulted in a shift to bay forests or mixed hardwood swamps (Hamilton 1984). Surface fires following logging may reduce regeneration of shrubs, but severe fires can result in conversion of cypress forests to scrub habitats (Hamilton 1984). Drainage also may allow establishment of species that could not tolerate sustained flooding (Marois and Ewel 1983).

3 Ecological Quality

Ecosystem Functions and Contributions to Biodiversity

Forested wetlands are considered to be an integral component of landscape diversity (Wigley and Lancia 1998). For example, in agricultural or urbanized landscapes, wetland forests may be the largest contiguous blocks of forest land in the area, and they are often the last remaining forested habitats in the landscape. These remaining forests often provide refugia for area-sensitive species where adjacent woodlands have been fragmented into smaller and more isolated stands. Large alluvial floodplains serve as regional and continental migration corridors for waterfowl, raptors, and songbirds, and smaller corridors of wetland forests may facilitate dispersal and other movements in an otherwise treeless landscape. A mosaic of habitat patches within a landscape can enhance biodiversity and improve sites for species that require several habitats to meet their life requisites (Wigley and Lancia 1998).

Despite the relatively small percentage of the landscape occupied by floodplain forests, they play a critical role in maintaining water quality by filtering agricultural runoff (Lugo, Brinson, and Brown 1990) and sediment from upstream disturbances (Brinson 1990). Floodplain forests are important in moderating downstream flooding by storing floodwaters, as well as conserving water during drought periods (Lugo, Brinson, and Brown 1990). Riparian areas can serve to buffer the effects of disturbance or land-use practices occurring in uplands, via biogeochemical processes that influence water quality, the aquatic ecosystem, and riparian-vegetation productivity (Green and Kauffman 1989). For example, riparian forests can remove large amounts (more than 65 percent) of dissolved nitrate occurring in runoff from adjacent-agricultural lands (Lowrance et al. 1984, Peterjohn and Correll 1984).

Riparian areas can also disperse suspended sediments and attenuate downstream sedimentation. During flooding, riparian vegetation can reduce water velocity, allowing the riparian zone to function as an area for the deposition of sediments (an important mechanism for the fertilization of floodplain soils) and other materials that would otherwise degrade water quality (Lowrance et al. 1984, Cooper et al. 1987). Riparian vegetation also reduces the erosional capacity of water in the floodplain, provides erosion control of river banks, and influences sedimentation processes necessary for the functioning of the ecosystem.

Although BLH forests typically comprise a very small proportion of a total landscape, they provide a variety of wildlife habitats, ranging from permanently flooded swamps to infrequently flooded forests, beaver ponds, and shrub communities. The complex pattern of topographic features associated with high-order river bottoms (such as berms, levees, sloughs, steep banks, shallow depressions, and wet benches) supports a diverse plant community that, in turn, supports a diverse wildlife community (Wharton et al. 1981, 1982; Wigley and Lancia 1998). Both the vertical structure and distribution of riparian vegetation contribute to the multiplicity of ecological niches available to wildlife species (Davis et al. 1996). In many floodplain forests, the vertical profile is multilayered and dominated by deciduous trees that may reach very large diameters (Wigley and Lancia 1998). Also, alluvial floodplains often contain important habitat features such as abundant detritus, hard and soft mast, ground-level vegetation, arboreal cavities, snags and downed woody debris, as well as a diverse aquatic and terrestrial invertebrate community. Nearly 30 percent of animal TES at least partially depend on BLH and other riparian habitats (Brinson et al. 1981). Plant and animal TES documented in BLH on military installations are listed in Tables 4 and 5. A comprehensive listing of all BLH fauna is too extensive for this report, but several reviews are available (e.g., Fredrickson 1979; Wharton et al. 1981, 1982; Brinson et al. 1981; Wigley and Lancia 1998).

As is the case for vegetation, researchers have debated and continue to debate whether the mere presence of a particular animal species in a habitat can be an indication of habitat quality (reviewed in Landres 1988, Croonquist and Brooks 1991). Neal and Jemison (1990) suggested a number of animal species that might qualify as functional indicators for southeastern bottomland hardwood habitats, including wood duck, green heron (*Butorides striatus*), American swallow-tailed kite (*Elanoides forficatus*), prothonotary warbler (*Protonotaria citrea*), gray squirrel, river otter (*Lutra canadensis*), swamp rabbit, Mississippi mud turtle (*Kinosternon subrubrum*), mud snake (*Farancia abacura*), canebreak rattlesnake (*Crotalus horridus*), and mole salamander (*Ambystoma talpoideum*). Gosselink and Lee (1989) recognized a greater variety of mammalian species as characteristic of mature BLH communities, including the black bear, red wolf, muskrat, raccoon, golden mouse (*Ochrotomys nuttalli*), mink, southern flying squirrel (*Glaucomys volans*), beaver, and southern short-tailed shrew (Blarina carolinensis).

Just as the presence of a particular animal species might suggest high-quality habitat, the recurrent presence of so-called “negative indicator species” (Gosselink and Lee 1989) in BLH communities can suggest that some level of degradation has occurred or may be occurring. Negative indicator species often include exotics and animals of open spaces and highly fragmented habitats such as the coyote and brown-headed cowbird (*Molothrus ater)*.

Table 4
Federally Listed and Candidate TES Animal Species and Animal Species of Concern (SOC) Documented in BLH Forest Communities on Military Installations in the Southeastern United States

Common Name	Scientific Name	Installation	Fed. Status	Habitat
Mammals				
Bat, Gray	*Myotis grisescens*	Fort McClellan, AL	E	Feed on insects associated with water or wetland vegetation and follow corridors of trees from roosts to feeding sites. Summer colonies inhabit areas in which open water and the banks of streams, lakes, or reservoirs are near suitable caves (LaVal et al. 1977, Tuttle 1976, Mitchell 1998).
Bat, Indiana	*Myotis sodalis*	Fort Knox, KY Jefferson Proving Ground, IN Picatinny Arsenal, NJ	E	During summer, Indiana bats require closed canopy, riparian forests for foraging, and hardwood stands with open to partially closed canopies for roosting.
Bear, Florida Black	*Ursus americanus floridana*	Eglin AFB, FL Camp Blanding, FL	SOC	A variety of forested wetlands, Florida scrub, and upland hardwood forests.
Myotis, Southeastern	*Myotis austroriparius*	Tyndall AFB, FL Fort Gordon, GA Fort Jackson, SC	SOC	Primarily in mature cypress-tupelo swamps and mature floodplain forests. Have also been observed in oak-pine and longleaf pine vegetation zones of East Texas. Most reproduction occurs in cave habitats. Winter in cave roosts but also hollow hardwood trees of floodplain forests.
Panther, Florida	*Felis concolor coryi*	Camp Blanding, FL	E	Frequently occupy hardwood hammocks and mixed swamp forests during the day (USFWS 1987).
Squirrel, Sherman's Fox	*Sciurus niger shermani*	Camp Blanding, FL	SOC	Primarily inhabits longleaf pine-turkey oak sandhills, mature longleaf pine forest, and mixed pine-hardwoods forest, but there is a marked seasonal use of bottomland forest during winter (Jordan 1995).

(Sheet 1 of 3)

Note: E = Endangered
T = Threatened
SOC = Species of Concern

Table 4 (Continued)				
Common Name	**Scientific Name**	**Installation**	**Fed. Status**	**Habitat**
Birds				
Eagle, Bald	*Haliaeetus leucocephalus*	Camp Robinson, AR Pine Bluff Arsenal, AR Jacksonville NAS, FL Key West NAS, FL Eglin AFB, FL Camp Blanding, FL MacDill AFB, FL Avon Park Air Force Range, FL Tyndall AFB, FL Fort Benning, GA Fort Stewart, GA Fort Gordon, GA Aberdeen Proving Ground, MD Picatinny Arsenal, MD MOT Sunny Point, NC Fort Jackson, SC Charleston NWS, SC Fort Belvoir, VA Fort A.P. Hill, VA Fort Eustis, VA Fort Pickett, VA Marine Corps Base, Quantico, VA	T[1]	Nesting habitat is almost always associated with creeks, rivers, and large bodies of water. Wintering bald eagles are most often associated with riparian and open water areas that provide an ample food supply and have adequate nocturnal roost sites.
Falcon, Peregrine	*Falco peregrinus*	Redstone Arsenal, AL Avon Park AFR, FL Camp Blanding, FL Tyndall AFB, FL Aberdeen Proving Ground, MD Picatinny Arsenal, MD MCB Camp Lejeune, NC Fort Eustis, VA	E[2]	Peregrines hunt in meadows, grasslands, wetlands, and open, early successional habitat types. Wetlands support the majority of the species preyed upon by peregrines.
Sparrow, Bachman's	*Aimophila aestivalis*	Fort Rucker, AL Camp Robinson, AR Little Rock AFB, AR Avon Park AFR, FL Tyndall AFB, FL NAS Cecil Field Camp Blanding, FL Eglin AFB, GA Fort Stewart, GA Fort Gordon, GA Fort Benning, GA MCLB Albany, GA Fort Polk, LA Camp Shelby, MS Fort Bragg, NC MCB Camp Lejeune, NC Fort Jackson, SC Fort Pickett, VA Fort A.P. Hill, VA	SOC	A year-round resident of pine savannas in the Southeastern coastal plain. In winter, Bachman's sparrow inhabits scrub oak, open broom sedge fields, fencerows and wet upland edges of river swamps, and saltwater shores.
				(Sheet 2 of 3)

[1] At press time, U.S. Fish and Wildlife Service had determined that the Bald Eagle would be delisted. However, the Bald Eagle will continue to be protected by the Migratory Bird Treaty Act and Bald Eagle Protection Act.
[2] Proposed for delisting.

Table 4 (Concluded)				
Common Name	**Scientific Name**	**Installation**	**Fed. Status**	**Habitat**
Birds				
Stork, Wood	*Mycteria americana*	NAS Mayport, FL Camp Blanding, FL Fort Benning Fort Stewart, GA Fort Gordon, GA	E	Inhabit fresh and brackish wetlands, primarily nesting in cypress or mangrove swamps. Feed in freshwater marshes, narrow tidal creeks, or flooded tidal pools. Particularly attracted to depressional marshes or swamps where fish become concentrated when water levels fall (USFWS 1996b).
Warbler, Cerulean	*Dendroica cerulea*	Camp Robinson, AR Fort Pickett, VA	SOC	Breeding cerulean warblers prefer, and are most common in, large and contiguous hardwood forest tracts, including BLH in the southern portion of their range.
Reptiles				
Alligator, American	*Alligator mississippiensis*	Redstone Arsenal, AL Fort Rucker, AL Pine Bluff Arsenal, AR Camp Blanding, FL Fort Benning, GA Louisiana AAP, LA Fort Jackson, SC MOT Sunny Point, NC Longhorn AAP, TX	T	Found in swamps, lakes, ponds, sloughs, drainage canals, and sluggish streams. Although most populations have fully recovered, this species is listed due to its similarity in appearance to the endangered American Crocodile.
Snake, Southern Hognose	*Heterodon simus*	Eglin AFB, FL Avon Park AFR, FL Tyndall AFB, FL Fort Benning, GA Fort Stewart, GA Fort Gordon, GA Fort Bragg, NC MCB Camp Lejeune, NC Fort Jackson, SC	SOC	The southern hognose snake is primarily a species of xeric, upland habitats; however, it is also known from dry river floodplains and hardwood hammocks.
Turtle, Alligator Snapping	*Macroclemys temmincki*	Fort Chaffee, AR Camp Robinson, AR Little Rock AFB, AR Fort Rucker, AL Eglin AFB, FL Tyndall AFB, FL Fort Benning, GA Fort Polk, LA	SOC	Winters in hibernacula such as undercut river banks and deep holes in bayous and lakes (Pritchard 1989; Harrel, Allen, and Hebert 1996). Nests are usually located near water on high and well drained sites (Pritchard 1989) such as natural or artificial berms bordering aquatic environments (USFWS 1991).
				(Sheet 3 of 3)

Table 5
Federally Listed Endangered and Candidate Plant Species of BLH Forests and Deepwater Swamps on Military Installations in the Southeastern United States

Common Name	Scientific Name	Installation	Fed. Status	Habitat
Woody Plants				
Ashe's Magnolia	*Magnolia ashei*	Eglin AFB, FL	SOC	Narrow creek bottoms and sandy woods near streams (Kral 1983).
Buckthorn	*Bumelia thornei*	Fort Stewart, GA	SOC	Small hardwood nonalluvial swamp (TNC 1995) where soil is normally saturated for long periods and woods bordering pond and creeks where some surface water stands during wet seasons (Godfrey 1988).
Forbs				
Alabama Anglepod or Alabama Milkweed	*Matelea alabamensis*	Fort Rucker, AL Eglin AFB, FL	SOC	Bottomland hardwood forests, upland hardwood forests (FNAI 1994b).
False Dragon Head	*Physostegia leptophylla*	Fort Stewart, GA	SOC	Swamp woodlands, river edges and inlet banks, and coastal sloughs (Kral 1983), typically found in wet muck or peat, often in shallow water (Godfrey and Wooten 1981)
Macbridia	*Macbridea caroliniana*	Fort Gordon, GA	SOC	Bottomland hardwood forest (L. Gaywin, Personal Communication, 1996), bottomland woodlands, marshes, bogs (Godfrey and Wooten 1981), alluvial woods (Radford, Ahles, and Bell 1969).
Southern Lady's Slipper	*Cypripedium kentuckiense*	Fort Polk, LA	SOC	Riparian Forest (Nelwyn McInnis, Personal Communication, 1995)
Texas Trillium	*Trillium texanum*	Barksdale AFB, LA	SOC	Acid hardwood bottoms, a shade plant, in association with bottomland hardwood trees (Kral 1983).
White Fringeless Orchid (Monkey-faced Orchid)	*Platanthera integrilabia*	Fort McClellan, AL	SOC	Boggy deciduous forested ravine woods and streambanks (ANHP 1994a).
Grasses, Sedges and Rushes				
Rhynchospora	*Rhynchospora decurrens*	Avon Park AF Range, FL	SOC	Swamp forests (Radford, Ahles, and Bell 1969, Godfrey and Wooten 1981).

Birds

Southern BLH forests support a diverse avian community (Wigley and Roberts 1994, Wakeley and Roberts 1996, Wigley and Lancia 1998), including breeding and wintering species, and birds that "stop over" during migration. This community supports more bird species than adjacent upland forests in the same area (Harris and O'Meara 1989; Lugo, Brinson, and Brown 1990) and numerous studies have documented the enormous importance of BLH for providing bird habitat. Smith, Hamel, and Ford (1993) reported that 200 of the 236 land birds (85 percent) in eastern North America can be found in Mississippi

Alluvial Valley (MAV) forests during at least some part of the year. Klimas, Martin, and Teaford (1981) estimated that approximately 100 species of breeding and wintering birds species inhabit BLH forests in the lower MAV. Pashley and Barrow (1993) stated that approximately 70 species regularly breed in BLH and about 30 of those are neotropical migrant birds. Many of these species, primarily neotropical migrant land birds, are undergoing declines in abundance and distribution concurrent with decreases in forest area (e.g., Burdick et al. 1989). Cerulean warblers (*Dendroica cerulea*), a former candidate species (C2) for listing by the USFWS, have experienced a more precipitous decline in abundance in North America than most other breeding songbirds. This species prefers, and is most common in, large and contiguous forested hardwood tracts (Hamel 1992); within their range in the southeast, they often are found in BLH. Although cerulean warblers may not breed on many southern DoD installations, the species may use hardwood stands on these installations as stopover habitat during spring and fall migration between North and South America (Evans and Fischer 1997). Game birds such as the wood duck (*Aix sponsa*) and other waterfowl, wild turkey (*Meleagris gallopavo*), and American woodcock (*Philohela minor*) rely extensively of BLH habitat as wintering, foraging, and nesting habitat (Wigley and Roberts 1994).

The bald eagle (*Haliaeetus leucocephalus*) is often associated with riparian zones near rivers and lakes, and usually nests near the bodies of water where it feeds. In the southeastern United States, most nests are constructed in dominant or codominant pines or cypress (USFWS 1996a). The wood stork (*Mycteria americana*) is North America’s only native stork and is a federally endangered species restricted to marshes, bottomland swamps, and other freshwater and brackish wetland communities in the extreme southeastern United States. Current estimates place the population level at 4,000 to 5,000 breeding pairs in the United States, with documented and potential occurrences being reported from numerous military installations in Florida, Georgia, and North Carolina.

Mammals

A variety of large and small mammals are found in BLH (Zwank et al. 1979, Wharton et al. 1982, Taylor, Cardamone, and Mitsch 1990). Although they are an important component of the ecosystem, many species (especially small mammals) have not been well studied (Wigley and Roberts 1994). Small mammal communities in alluvial floodplains often are dominated by a few species that sometimes vary among hydrologic zones (Wigley and Lancia 1998). Common large mammals in BLH include white-tailed deer (*Odocoileus virginianus*), beaver (*Castor canadensis*), raccoon (*Procyon lotor*), swamp rabbit (*Sylvilagus aquaticus*), marsh rabbit (*S. palustris*), gray squirrel (*Sciurus carolinensis*), muskrat (*Ondatra zibethicus*), nutria (*Myocaster coypus*), river otter (*Lutra canadensis*), and mink (*Mustela vison*). Many of these species are closely associated with wetter zones (Wigley and Lancia 1998).

Several mammalian TES inhabit BLH, either seasonally or as a primary habitat. Summer maternity colonies of Indiana bats (*Myotis sodalis*) are most often located in floodplain deciduous forests or upland stands adjacent to riparian or floodplain forests. Indiana bats require closed canopy, riparian forests for foraging and hardwood stands with open to partially closed canopies for roosting. Adult females establish maternity roosts in hollow trees and under the loose bark of various tree species (e.g., cottonwood, shagbark hickory [*C. ovata*], bitternut hickory [*C. cordiformis*], and green ash) (Humphrey, Richter, and Cope 1977; Cope, Richter, and Searley 1978). Optimal roost sites occur beneath the bark of dead trees with adequate spaces to allow for air circulation and for bats to change position on the trunk (Garner and Gardner 1992). Gray bats (*M. grisescens*) forage primarily over water along rivers or lake shores where flying insects are abundant, and depend on the associated riparian vegetation (Tuttle 1976, 1979; LaVal et al. 1977). Summer colonies of gray bats inhabit areas in which open water and the banks of streams, lakes, or reservoirs are reasonably close to roosting sites and maternal caves (Mitchell 1998). Although southeastern myotis (*M. austroriparius*) primarily inhabit caves, a maternity colony in Illinois was found in a hollow tupelo tree within a mature cypress-tupelo swamp. Southeastern myotis use a variety of habitats for feeding but often have been reported to forage along water courses (Reynolds and Mitchell 1998). Schmidly et al. (1977) observed them feeding over narrow, slow-moving creeks in wooded areas of eastern Texas, and they have been reported foraging in forested wetlands in Illinois (Gardner et al. 1992) and Tennessee (Graves and Harvey 1974).

Fox squirrels (*S. niger*), including Sherman's subspecies (*S. n. shermani*; former C2 candidate species for listing), primarily use upland pine-oak habitats but often use edge habitats, including bottomland forest, during winter (Jordan 1995). The Florida panther (*Felis concolor coryi*), restricted to southern Florida, often occupies mixed swamp forests and hardwood hammocks during daylight hours to avoid detection (USFWS 1987). Similar to the panther, key deer (*O. virginianus clauvium*) are restricted to southern Florida and use BLH as cover, forage, and bedding areas (USFWS 1985).

The American black bear (*Ursus americanus*) is the most abundant and widespread bear in North America, but there is concern for both the Florida and Louisiana subspecies, which require very large, contiguous tracts of habitat (including BLH). Due to their tremendous home range (up to 124 km^2 for males and 26 km^2 for females) and secretive nature, their occurrence on many of the military installations in the southeast is poorly documented. Eglin AFB, Florida, has one of the five major populations of the Florida subspecies. Like many animal TES, habitat loss is the main reason for population declines (Nowack 1986), although illegal hunting of black bear (to obtain their gall bladders, meat, etc.), relatively low reproductive potential, and intolerance to human disturbance (Hellgren and Vaughan 1989) have contributed to their decline. Habitat loss has resulted from reservoir construction that flooded extensive areas of former bottomland forest while agricultural development, urbanization, and road development has further fragmented the remaining habitat. Black bear adults do not reach reproductive maturity until 3 to 4 years of age, and, if in good physical condition, they produce an average litter of two young every 2 years thereafter. This low reproductive potential has made it difficult for populations to sustain the increased mortality from vehicle-bear collisions (the leading cause of death for bears in Florida), harassment from pets (dogs), and other conflicts associated with urbanization.

The endangered red wolf (*Canis rufus gregorii*) once ranged throughout southeastern BLH, especially in coastal areas. However, habitat loss and the spread of coyotes (*C. latrans*), which interbreed with wolves, spurred the USFWS to capture remaining wolves for a captive-breeding program (Harris and O'Meara 1989). Red wolves are currently being released into the wild at selected sites in the southeastern United States.

Reptiles and Amphibians

Bottomland hardwoods have a diverse herpetofauna that inhabit the array of flood/habitat conditions (Wharton et al. 1982, Wake 1991, Fredrickson and Batema 1992), and they may be good indicators for these ecosystems (Wake 1991). In portions of the southeast, reptiles and amphibians may constitute as much as 45 percent of the native fauna, excluding fish (Vickers, Harris, and Swindel 1985). Amphibians associated with BLH habitat tend to use the lower zones for reproductive purposes but may exploit drier or seasonally flooded sites for other needs (Clark 1979). Many reptiles use lower BLH zones for food and cover and migrating to more xeric sites to lay eggs (Wigley and Roberts 1994). Standing water following flood events and heavy rains is important for the reproductive cycle of many amphibians in alluvial floodplains (Wigley and Lancia 1998). Amphibians often are most abundant in moist conditions provided by a closed canopy and abundant leaf litter (Rudolph and Dickson 1990), and reptiles usually are most abundant where understory vegetation is dense and there is an abundant prey base (Wigley and Lancia 1998). See Wharton et al. (1982), Clawson, Lockaby, and Jones (1997), and Phelps and Lancia (1995) for further discussion on BLH herpetofauna.

The alligator snapping turtle (*Macroclemys temmincki*) is a species of concern found in the southcentral and southeastern United States throughout the Mississippi River Valley and Gulf Coast states (Lane and Mitchell 1997). Sloan and Taylor (1987) reported that alligator snapping turtles in Louisiana wetlands preferred aquatic habitat consisting of bayou channels bordered by baldcypress or lakes with floating mats of dense herbaceous vegetation associated with baldcypress or buttonbush (*Cephalanthus occidentalis*). Nests are usually located near water on high and well-drained sites (Pritchard 1989) such as natural or artificial berms bordering aquatic environments (USFWS 1991). Floodplain forest is among the wide variety of terrestrial sites used for nesting by alligator snapping turtles (Ewert 1976).

Vegetation

The tree species diversity of floodplain forests varies widely. Approximately 70 tree species occur in southeastern BLH because of the variety of soil conditions and physical settings (Malac et al. 1981). Lectman et al. (1983) (cited in Brinson 1990) noted as many as 31 tree species along the Appalachicola River in Florida, and as few as 1 tree species in areas dominated by cypress, tupelo, or silver maple. On a local scale, the number of tree species present generally follows a hydrologic gradient, with fewer species in the wetter environments (Brinson 1990, Sharitz and Mitsch 1993).

Habitat Loss and Degradation

During European settlement, approximately 80 million hectares (ha) of forested wetlands existed in the conterminous United States (Gosselink and Lee 1989). By the mid-1970's, this amount decreased to between 20 and 29 million ha (Frayer et al. 1983, Abernethy and Turner 1987, Gosselink and Lee 1989, Dahl and Johnson 1991), with more than one-half of this amount occurring in the 12 southern states (Turner, Forsythe, and Craig 1981). Shepard et al. (1998) reported that approximately 55 percent of the total wetland loss in the nation from 1982 to 1992 occurred in the 12 southern states.

Estimates of the extent of southern forested wetlands vary because of differing methods and definition of habitat. Historically, BLH forests were one of the dominant types of forested wetland ecosystems of the United States. However, approximately 80 percent of original southeastern BLH has been converted to other land uses (Turner, Forsythe, and Craig 1981; Mitsch and Gosselink 1986; Haynes and Moore 1988). Recent estimates suggest that there are between 6.6 and 13 million ha of BLH remaining. The greatest loss of BLH occurred early in the 20th century, but from 1960 to the mid-1970s alone, an estimated 2.6 million ha of southern BLH were lost (Turner, Forsythe, and Craig 1981). These losses have been greatest in the MAV, where 78 percent of forested wetlands (mostly BLH) were estimated to have been lost (MacDonald, Frazer, and Clauser 1979). Bottomland hardwoods along the lower Mississippi River were still being cleared for agriculture during the 1980s in tracts as large as 12,000 ha (Gosselink and Lee 1989). In other areas of the southeast, 85 percent of forested wetlands have been lost in the Tensas Basin, Louisiana (Gosselink et al. 1990), and 82 and 60 percent of BLH has been lost in Oklahoma (Brabender, Master, and Short 1985) and Tennessee (Pyne and Durham 1993), respectively. McWilliams and Faulkner (1991) reported that overall losses of BLH continue at approximately 65,000 ha/year. Future losses of BLH are projected to continue into the 21st century; the USDA Forest Service (1988) projected that the area of BLH will decrease from 12.2 to 10.6 million ha by 2030. These tremendous losses of forested wetlands in the southeast are of great concern because of the many functions these ecosystems provide (Table 6). Bottomland hardwoods play an important role in landscape processes, because

Table 6
Potential Functions and Values Provided by BLH and Deepwater Swamp Communities [1]

Functions	Description
Primary productivity	High natural productivity supports a complex wetland food web.
Litterfall and decomposition	High productivity of litterfall and subsequent organic decomposition provide matter for the aquatic foodchain.
Organic export	Detritus exported to aquatic organisms outside the immediate wetland environment.
Sediment deposition	Deposit of sediment across floodplains when streamwaters leave the channel.
Retention of nutrients and toxins	Many nutrients and toxins may be at least temporarily retained in BLH.
Biochemical transformations	Anaerobic transformations in standing water and sediments.
Surface water storage	Results from the pulses of high water that occur seasonally in BLH.
Groundwater storage	Groundwater supports BLH forests during dry periods and serves to reduce surface waterflow during floods.
Fish and wildlife habitat	BLH constitutes a transition habitat between aquatic and upland communities, providing food, cover, and water to a diverse array of species including numerous TES; nearly 30% of all TES are at least partially dependent upon riparian habitats.
Groundwater discharge	Extends the period of streamflow in some regions.
Values	**Description**
Timber harvest	Proper harvesting can enhance wildlife habitat and provide commercially valuable timber for construction, furniture, firewood, and pulp.
Fish and wildlife harvest	Among the most productive habitats for game and commercially important species such as waterfowl, furbearers, crayfish, white-tailed deer, and turkey.
Water quality protection	Can be effective in removing some nutrients, sediments, and metals from surface waters; forested wetlands inhibit eutrophication by converting inorganic nutrients to their organic forms.
Erosion control	Woody vegetation is effective in binding soil, stabilizing banks, and reducing water velocities, resulting in reduced sheet erosion within the wetland and streambank erosion downstream.
Flood storage and control	Reduces economic losses from flooding and promotes human use of downstream areas.

[1] After Taylor, Cardamone, and Mitsch (1990).

they are linked to upland and upslope terrestrial ecosystems as well as downstream aquatic ecosystems (Walbridge and Lockaby 1994).

Most major streams and rivers in the southeastern United States have been manipulated by human activities (e.g., dams, channelization, dredging, pollution). Similarly, floodplains associated with these streams and rivers have also been affected (e.g., levees, ditches, timber harvest, agriculture) (Malac et al. 1981). Bottomland hardwoods are sensitive to disturbances and some management practices, and TES may be influenced by the type and scale of change. Livestock grazing, agriculture, large-scale timber removal, road-building, urban development, and recreation are among the uses that have mainly negative effects on riparian and aquatic ecosystems and their associated species (Budd et al. 1987; Meade, Yuzyk, and Day 1990; Medina 1990). Most BLH clearing in the eastern United States has been for agricultural production (Turner, Forsythe, and Craig 1981), but clearing for housing developments, urbanization, and industrialization (Neal and Jemison 1990) has also occurred. Another problem that arises from urbanization and recreation is conflict between humans and wildlife.

Indicators of Community Quality

Decisions regarding TES and other land-use priorities can be guided by site classification based on ecological quality. Site quality initially can be assigned using baseline data but should be augmented by a monitoring program that evaluates the effects of land-use decisions. One such ranking system developed for Eglin AFB, Florida, was introduced in the companion document by Harper et al. (1997). Determination of community quality has obvious benefits for TES conservation planning, as low-quality communities are less likely to support sustainable TES populations and therefore should be treated differently in terms of protection, restoration efforts, and allowable land uses. Use of a quality ranking system for management purposes results in priority protection being given to the higher quality TES habitats and restoration activities for communities that have the greatest potential to become high-quality TES habitat with minimum restoration efforts. Alternatively, this system ensures that limited resources are not misdirected toward the restoration of low-quality communities or those with limited potential for TES habitat. Finally, plant communities on military lands are subject to multiple uses, and use of a quality ranking system in combination with an assessment of impacts of various land uses can provide managers an objective means to determine which activities are appropriate in which communities, based on the potential to provide quality habitat for TES.

High-quality sites

Species characteristic of BLH communities are described in Chapter 2, under "Biological Composition." Documentation of the plant species present on a site does not in itself identify high-quality habitat. Rather, the combined use of visual and structural indicators (both qualitative and quantitative) has been a common approach to provide the broader context in which to evaluate the composition data and assess habitat quality. High-quality BLH typically is described as mature, late-successional forest. For example, in describing what constitutes quality bottomland habitat in major alluvial (red river) floodplain forests of the south, Kellison et al. (1998) write:

> With large crowns and clear boles, upper canopy species of well-developed stands clearly dominate the community, forming "cathedral-like" canopies that commonly reach heights more than 30 m (100 ft). Seen from outside the stand, the understory of many first bottoms appears thick with vegetation and impenetrable. This impression results from seeing high-density understories and vines along stand edges; it is not typical of stand interiors. Inside well developed [high quality]

> stands, the vegetation layers below the upper canopy are sparse, owing to flooding and lack of understory tolerant species. The lower canopy invariably contains Ironwood, possumhaw (*Ilex decidua*), American holly (*I. paca*), hawthorn (*Crataegus* spp.), Virginia willow (*Itea virginica*)...The vine layer contains poison-ivy (*Toxicodendron radicans*) and crossvine (*Bignonia capreolata*). The herb layer contains false-nettle, violets, giant cane, sedges, and uniola grass. Most of the ground surface is covered with a thin layer of leaf litter.

Low-quality sites

Severe disturbance and/or colonization of unvegetated sites such as point bars and oldfields can initially result in low-quality alluvial floodplain sites. These areas are characterized by reduced species richness or pure stands of early-successional shade-intolerant species. In contrast, disturbance by harvesting floodplain forests often yields young stands of equal diversity (Kellison et al. 1998). Succession following harvest in deepwater swamps, oxbow lakes, and sloughs is different than in alluvial floodplains since the dominant species of baldcypress, swamp tupelo, water tupelo, and green ash replace themselves through sprouting. On the wetter, most poorly drained sites, and on those where the harvested timber is too old to readily sprout, the site becomes dominated by invading species such as cattail (*Typha* sp.), bulrush (*Scirpus* spp.), and alligator-weed (*Alternanthera philoxeroides*) followed by black willow and red maple (Kellison et al. 1998). None of these communities are considered as high in quality for TES as the floodplain forests or deepwater swamps prior to harvest.

4 Impacts and Management Recommendations

Management of BLH has traditionally focused on maintaining or restoring a stable zone of vegetation adjacent to the aquatic system for the enhancement of water quality and wildlife habitat (Gore and Bryant 1988). Some strategies allow BLH to function naturally (or in their present condition based on current hydrology) with minimal direct management, while others are managed with specific techniques to diversify the landscape and create habitat for various wildlife species (Malac et al. 1981). Evaluation, design, and implementation of management strategies in these areas depend on many considerations, such as geographic location, soils, water regime, topography, existing vegetation, and fauna. Malac et al. (1981) stated that management strategies in BLH systems are influenced primarily by hydroperiod, physical factors, and groundwater characteristics; hydroperiod was considered the most significant factor affecting management options regardless of whether the forest was being managed for timber, wildlife, recreation, or water quality. Physical characteristics that can affect management decisions include soil type, existing and potential plant communities, size and shape of the forested area, and presences of gullies, oxbow lakes, sloughs, and old point bars. Kellison et al. (1998) emphasized the importance of considering the natural disturbance regime (e.g., flooding) when managing BLH.

Because at least 80 percent of all original BLH has been cleared, it is important to maintain remaining areas in the highest quality condition possible, to maintain vital ecosystem functions and TES habitat. Although DoD lands exist to support the military mission, there is opportunity to conserve valuable ecosystems and rare species as well.

The following paragraphs discuss management recommendations that support the military mission on training lands in addition to providing guidelines for TES conservation based on community management. This information is based on literature review, contacts with endangered species experts, and guidelines extracted from installation reports.

Fragmentation and Land-Use Conversion

Impacts

The size, configuration, and arrangement of BLH tracts strongly influences species richness and animal community composition. Discontinuous patches of BLH may not be suitable as management units for species that require large, unfragmented blocks of habitat (Malac et al. 1981), and highly fragmented stands provide poor habitat for many TES (Harris and O'Meara 1989). Thus, maintaining large, contiguous tracts of BLH is critical to many animal TES. Although both large and small habitat patches can have high species richness, smaller patches tend to have highly mobile habitat generalists whereas larger tracts favor specialists (Blake and Karr 1984), which are those species most in need of conservation (Noss 1983). Moreover, Wilcox and Murphy (1985) assert that habitat fragmentation "...is the most serious threat to biological diversity and is the primary cause of the present extinction crisis." Harris, Sullivan, and Badger (1984) and Sharitz and Mitsch (1993) list the primary effects of fragmentation on native fauna: (a) loss of wide-ranging species, (b) loss of interior or area-sensitive species, (c) erosion of genetic diversity within rare species, and (d) increased abundance of weedy plant species.

Urban and agricultural encroachment has especially been responsible for fragmenting forested wetlands, which has resulted in significant habitat loss and degradation in the Southeast (Harris 1984). This pattern is expected to continue. Equally relevant but less obvious are the more indirect impacts to TES associated with urbanization on the remaining BLH habitat. Bottomland hardwoods are frequently used for recreational activities such as picnicking, camping, boating, backpacking, hunting, birdwatching, and use of all-terrain vehicles, each potentially having significant impacts on TES plants and animals. Areas around campgrounds, for example, often appear denuded of standing vegetation, with dead wood being quickly removed for camp fires. In intensively used recreational areas, vegetation diversity may decrease with increasing use, and tree regeneration may be reduced or eliminated (Marnell, Foster, and Chilman 1978; Trumbull et al. 1994). Recreational impacts to most TES are largely unknown and appear to vary greatly by vertebrate group, age, season, and species (reviewed by Knight and Gutzwiller 1995). Most species exhibit some degree of negative physiological response to disturbance. Black bears, for example, may be more likely to abandon dens if disturbed (Goodrich and Berger 1994). Disturbance to birds during the breeding season can result in abandonment of established nests and increased egg and fledgling mortality. There can be recreational impacts on wildlife in the nonbreeding season as well. Although the impacts are more studied in northern species, repeated disturbance to feeding activities can reduce the buildup of fat reserves needed to attain breeding status, migrate, or survive periods of prolonged climatic stress (Owens 1977; reviewed by Knight and Gutzwiller 1995).

Loss of BLH has contributed to the decline or extinction of several BLH obligate species. For example, Bachman's warblers (*Vermivora bachmanii*) once inhabited southeastern BLH, nesting near the ground in open patches.

Habitat loss and, possibly, brown-headed cowbird nest parasitism, probably contributed to its demise (Harris and O'Meara 1989). Fragmentation of BLH leads to an increase in "edge," which decreases suitable breeding habitat for many species considered "interior" forest species. Cerulean warblers, as well as many other neotropical migratory songbirds that appear to be in decline, require large contiguous tracts of mature, deciduous hardwoods for suitable breeding habitat (Hamel 1981; Robbins, Fitzpatrick, and Hammel 1992). The decline of cerulean warblers is due in part to loss of extensive floodplain forests of the central and eastern United States and habitat fragmentation on both the wintering and breeding grounds (Flaspohler 1993). Loss of floodplain forests has resulted in limited cerulean warbler nesting habitat for an already stressed population, and the resulting fragmentation and isolation of large tracts of mature deciduous species has exacerbated the problem by allowing an increase of brown-headed cowbird nest parasitism. The bald eagle was once widely distributed throughout the southern United States, but its breeding habitat, which largely consists of BLH, is highly fragmented in the southeastern coastal plain (Harris and O'Meara 1989).

Florida and Louisiana subspecies of black bear require large, contiguous blocks of BLH. For example, Cox et al. (1994) reported that a relatively stable population of black bears in the southeast would require a contiguous block of habitat approximately 2,000 to 4,000 km^2. Management recommendations for black bears have typically focused on the acquisition and protection from development of large blocks of forest, maintenance of adequate dispersal corridors between existing blocks, ensuring a sufficient pool of potential den trees, and forest manipulation to promote mast production. Of these, the latter two options may be the most practical for military land managers. Fragmentation creates unsuitable habitat for both the bear and Florida panther and has led to a high incidence of vehicle-induced mortality of black bears on roadways in Florida (Sharitz and Mitsch 1993). Cox et al. (1994) further report much of bear management in Florida is focusing on connecting large Federal and state landholdings in an attempt to provide movement corridors and provide larger blocks rather than relying on within-habitat modifications.

Management recommendations

Further fragmentation of existing BLH habitats should be discouraged. Installation managers should use aerial photos, stand attribute data, and if available, spatial data contained in geographic information system (GIS) databases to identify the location and ecological condition of BLH stands on the landscape and identify where TES actually or potentially occur in or near these stands. These data also will assist in identifying stands that need habitat improvements or stands that have the potential to be restored following degradation or conversion to other uses.

Installation managers should strive to connect smaller, fragmented BLH stands to produce larger contiguous tracts. This can be accomplished by establishing linkages using natural or artificial regeneration, or with proven restoration measures. Restoration can link higher-quality BLH stands or provide a buffer zone around existing stands. Linkages between existing stands could potentially serve as important corridors for genetic interchange, seasonal movements, or habitat. When restored stands mature, small clearcut harvesting could occur within them to provide timber revenues. Single-tree or small-group selection harvests that emulate natural disturbance regimes could be conducted in the core areas of higher quality habitat (S. King, Personal Communication, 1998). Installations should strive to provide as much high-quality, mature BLH forest as possible; however, providing BLH stands of different successional stages on the landscape will provide a range of habitats that support increased biodiversity.

Silvicultural Activities

Impacts

Floodplain forests are attractive to the timber industry because of their tree biomass and rapid rate of tree growth (Brinson 1990). BLH provide approximately 17 percent of timber in the southeast region (USDA Forest Service 1988); oaks, gums, and baldcypress are primary species harvested. The most common silvicultural practice in southeastern BLH is clear-cut harvesting followed by natural regeneration during dry portions of the summer (Walbridge and Lockaby 1994, Wigley and Roberts 1994) (Figure 4). In commercial stands, clearcutting is often followed by intensive silvicultural plantings (generally of one or a few species that produce an even-aged harvest) or conversion to agriculture; in either case, the habitat for most floodplain plants and animals is significantly altered. Clearcutting at least temporarily decreases evapotranspiration and productivity and may disrupt natural nutrient cycling (Brinson 1990). Selective logging appears to have a negligible effect on ecosystem processes but may have a large impact on species composition and regeneration dynamics (Brinson 1990).

The impacts of logging on river and stream water quality include nutrient release from increased organic matter breakdown, erosion and sedimentation in aquatic ecosystems, disruption of streambanks and streambeds from skidding trails, deposition of slash in streams, accidental fuel or lubricant spills, and removal of shading cover (Irland 1985). Road construction for hauling timber away from harvesting operations is a common activity that can also lead to such impacts. Depending on their design, these roads can alter hydrologic relationships by constricting floodplains and subsequently change floodwater velocity via roadbeds and culverts (Gosselink et al. 1990, Walbridge and Lockaby 1994).

Figure 4. Timber harvesting in and adjacent to BLH is among many land-use practices contributing to loss of habitat

In deepwater swamps, natural regeneration of baldcypress was poor to nonexistent following logging operations (Conner, Toliver, and Sklar 1986), because swamps remain flooded for much of the year (Conner 1994). Baldcypress seeds cannot germinate in standing water (Demaree 1932) nor do they grow tall enough to survive frequent flooding (Conner 1994).

The threat of watershed erosion is increased after logging in upland areas (Figure 5). Vegetation loss from clearcutting or road construction can have direct effects on both aquatic and terrestrial ecosystems. Loss of plant biomass in the watershed decreases transpiration, subsequently increasing the amount of runoff entering riparian and aquatic ecosystems (Gore and Bryant 1988). Following logging, sediment continues to enter riparian and aquatic ecosystems for many years (Harr and Nichols 1993). Excessive siltation resulting from upslope logging operations has caused damage or destruction of rare plant populations in bottomland and deepwater forests. Increased silt deposits following upslope logging operations have been reported to be the primary threat to deepwater swamps at Fort Stewart, Georgia (TNC 1995). Losses of populations of monkey-faced orchid (*Planthera integrilabia*) and Texas trillium (*Trillium texanum*) have been attributed to soil disturbance associated with timber harvest and subsequent siltation in bottomland and riparian forest areas as

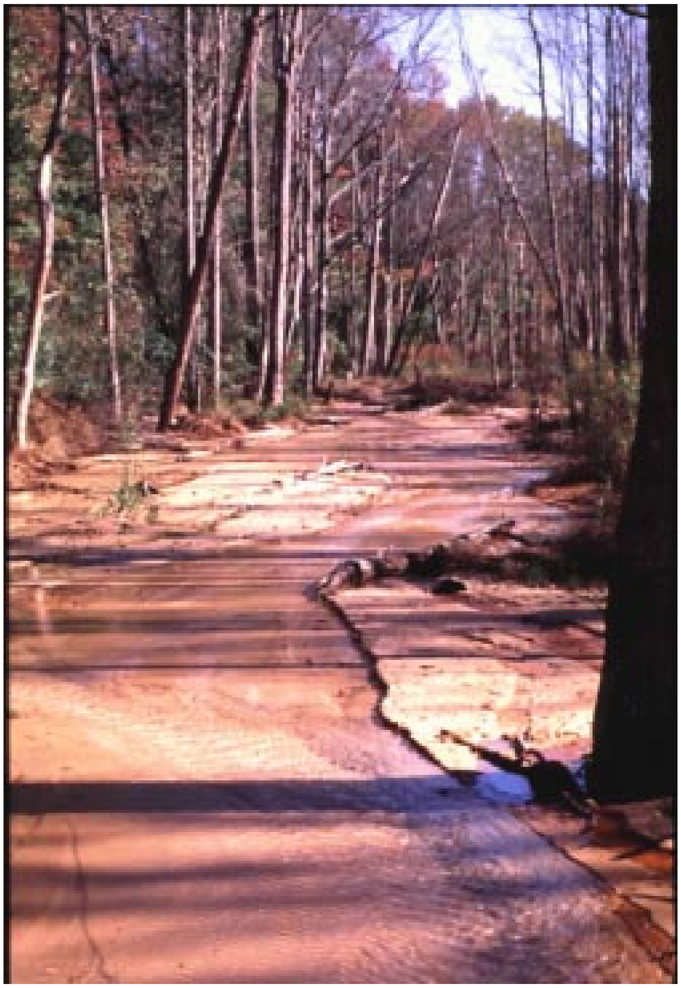

Figure 5. Land-use practices in uplands can increase sedimentation in bottomland hardwoods, leading to tree mortality

well as changes in hydrology due to rutting by machinery (Shea 1992; McInnis 1994; Zettler, Ahuja, and McInnis 1996). Logging also can decimate the herbaceous layer and tree seedlings (McInnis 1994). Populations of buckthorn (*Bumelia thornei*) and Alabama anglepod (*Matalea alabamensis*) are also threatened by siltation associated with logging activities (Mount and Diamond 1992, TNC 1995).

Most logging on wet soils today involves mechanized felling and removal by rubber-tired skidders. In wet sites, conventional felling machines may cause a significant amount of soil damage and alteration of drainage patterns. Some operations use modified rubber-tired carriers with wide or dual tires to increase mobility, but there may be a large amount of visible damage to the site (Jackson and Stokes 1991). Dual-tire skidders are cost effective under wet conditions and are able to work in harsh conditions but may leave the site with high levels of disturbance (Jackson and Stokes 1991).

Detrimental changes in the natural light conditions required by rare plants, including decreases and increases in light, have been associated with logging. Although logging may improve light conditions for monkey-faced orchid and Alabama anglepod at first, subsequent encroachment by shrubs following cutting can reduce light to levels lower than before cutting (Shea 1992, Mount and Diamond 1992). Texas trillium, on the other hand, requires shade, and is damaged by opening of the canopy caused by logging (Nelwyn McInnis, Personal Communication, 1994). Although the specific influence of logging activities has not been documented for many rare plant species, it is probable that most species would be similarly influenced by direct damage, soil disturbance, changes in hydrology, siltation, and light levels associated with tree harvest. The objectives of any harvest should be viewed in light of the potential for negative effects on plant TES present on the site.

According to Wigley and Roberts (1994), wildlife habitat components (i.e., food, cover, water) can potentially be affected by tree harvesting, but the magnitude of the effects depends on the intensity of the harvest. Some negative impacts were noted as well as potential benefits of proper harvesting techniques to enhance biodiversity (Table 7). Habitat changes will continue to occur after the harvesting operation as plant succession proceeds. The direction, magnitude, and rate of habitat changes depend on factors such as (a) structure and composition of the residual stand and woody vegetation, (b) flooding regime, (c) browsing by herbivores, and (d) subsequent management activities. Under most circumstances, tree species composition will tend to progress toward that of the harvested stand (Wigley and Roberts 1994). Generally, even-aged clearcutting leads to a predictable succession of habitat types, including (a) an impenetrable thicket of weeds, briars, vines, and tree seedlings during approximately the first 10 years, (b) a sapling stage characterized by tree seedlings and sprouts (10 to 15 years), (c) a pole stage during which competition leads to the development of a somewhat thinned forest of young trees (15 to 25 years), (d) a small sawlog stage of maturing trees (25 to 40 years), and (e) a large sawlog or mature forest stage that can be managed in perpetuity by proper silvicultural techniques. Each of these stages of forest development have characteristic fauna associated with them; the thicket stage and mature forest stage likely have the highest diversity of animals (Kellison et al. 1998).

Forest management practices can alter stand-level habitats for mammals by affecting the availability of mast, browse, invertebrates, ground-level vegetation, arboreal cavities, vertical structure, and downed woody material (Forsythe and Roelle 1990). Reducing canopy cover often lowers the immediate availability of hard mast and cavities and the amount of vertical structure (Wigley and Roberts 1994). Lowering the availability of these features over large, contiguous areas will decrease habitat suitability for mammals such as the gray squirrel, southern flying squirrel, and raccoon. Densities of small mammals typical of BLH communities also can decline following intensive timber harvesting (McComb and Noble 1980), but populations of some small mammals often increase due to greater food production and cover from ground-level vegetation and cover from logging slash. McComb and Noble (1980) reported higher capture rates for small mammals in harvested BLH tracts than in uncut stands in Louisiana and

Table 7
Potential Beneficial and Detrimental Consequences of Silvicultural Activities on Wildlife Populations

Potential Benefits
Foliage height diversity may be enhanced through thinnings that allow vegetation to respond to sunlight penetration (Beck and Harlow 1981).
Production of ground-level vegetation and soft mast generally increases following harvest.
Invertebrate availability at the ground level may increase in response to changes in ground-level vegetation.
Downed woody cover usually increases following harvest if logging slash is not piled and burned.
The rotting wood from decomposing logging slash (in clearcuts) and downed logs (within interior forest habitats) harbors beetles, grubs, and other invertebrates that are protein-rich food sources for many wildlife species (Weaver et al. 1990).
Potential Negative Consequences
Stand-level availability of hard mast, arboreal cavities, and some foraging substrates (e.g., canopy layers) often are reduced immediately after harvest.
Removal of snags can reduce populations of species that forage, perch, or nest in them, such as the prothonotary warbler (*Protonotaria citrea*), wood duck (*Aix sponsa*), and bald eagle (Pashley and Barrow 1993).
Tree removal could potentially be harmful to some bats, such as the southeastern myotis, that often roost in large, hollow hardwood trees.
Birds may be affected by a variety of impacts on habitat components associated with timber harvest. These include changes in stand structure and the availability of hard and soft mast, ground-level vegetation, invertebrates, snags, and arboreal cavities. Removing overstory trees will have immediate stand-level impacts on some species. Removal of spanish moss (*Tillandsia usneoides*) and large/old trees, particularly baldcypress, can negatively impact northern parula (Parula americana), yellow-throated warbler (*Dendroica dominica*), yellow-throated vireos (*Vireo flavifrons*), and other BLH associated neotropical migrant species (Pashley and Barrow 1993).
Initially, harvested areas generally have higher water tables and higher soil temperatures because of vegetation changes and soil structure changes from skidder traffic. Habitat changes continue long after the harvest and depend upon factors such as the structure and composition of the residual stand and woody regeneration, flooding regime, browsing by herbivores, and subsequent management actions

Mississippi. Hurst and Smith (1987) determined that forage and cover for the swamp rabbit were generally more abundant in young clearcuts and thinned stands than in closed-canopy forests. Few studies are available regarding effects of timber harvest on BLH herpetofauna (Wigley and Roberts 1994). In Louisiana, Raymond and Hardy (1991) found that increased insolation from clearcutting a pine-hardwood stand resulted in higher soil temperatures and greater evaporative water loss from the soil and understory and hypothesized that these changes caused reduced survival in mole salamanders. Rudolph and Dickson (1990) found relatively few amphibians and reptiles in narrow (0 to 25 m) streamside management zones with open overstories and midstories, dense shrub layers, dense herbaceous vegetation, and little leaf litter. In contrast, they believed that higher abundance of herpetofauna observed on wider streamside zones were due to the closed canopy and leaf litter layer characteristic of wider (50 to 95 m) streamside zones.

Management recommendations

Harvest management. Timber harvest is often the primary focus in BLH management because of the potential economic benefit from logging. Silvicultural practices vary considerably from region to region and according to forest type. Both even- and uneven-aged timber management practices are used, but the most common silvicultural practice in BLH is clear-cut harvesting followed by natural regeneration during dry portions of the summer to produce an even-aged forest (Walbridge and Lockaby 1994, Wigley and Roberts 1994). However, clearcutting usually is not a standard practice in BLH on military installations.

Sustained timber production in BLH can be compatible with sound management for plant and animal TES if silvicultural guidelines and restrictions are followed. Even if BLH are included in an installation's commercial forestry program, various silvicultural practices should be prohibited, such as large-scale, even-aged timber management, widespread application of insecticides and herbicides, and skidding practices that run parallel to or in the streambed (Fredrickson and Reid 1990; also see Table 8). These restrictions should apply not only to corridors along permanent rivers and streams, but also along branches and intermittent streams with permanent pools. If timber is harvested, Fredrickson and Reid (1990) recommend that uneven-aged and single tree removal methods be used. Kellison et al. (1998), however, stated that uneven-aged systems are difficult to successfully implement in forested wetlands. Even though the value of intermediate stand management (i.e., thinning of stands of intermediate age) is well recognized for increasing timber quality in alluvial floodplains, the practice is difficult to carry out because of flooding during some portions of the year. Some depressions remain wet and mucky for long periods, and the use of logging machinery causes damage to tree boles and root systems. Because of this, thinning alluvial floodplain stands is often discouraged unless loggers use extreme care in the process (Kellison et al. 1998).

There are several alternatives for low-impact harvesting systems on wet soils. The following are taken from Jackson and Stokes (1991).

Felling. Mechanized felling can be done by swing feller-bunchers on tracks. Although costly, disturbance is reduced by limiting the amount of travel on the site and through the use of wide tracks. In extremely wet sites, mats can be used to increase feller-buncher mobility and reduce site disturbance. Felling technology is now available that includes lightweight, long-reaching machines that combine high production with little disturbance. Using grapple-saws would increase the flexibility of the feller-buncher, since it would reduce the weight on the end of the boom and allow the felling machine to perform limited bucking and topping. Such a machine can cut the trees, cut off the tops and some of the

Table 8
General Silvicultural Guidelines for Wooded Riparian Areas when TES Management is a Priority

Suggested Guideline	Purpose
Maintain a “No-Cut” buffer zone of sufficient width adjacent to streams. Width will depend on site-specific conditions such as slope and type of soils. Widths of 15 m generally protect water quality but much wider buffers are needed for wildlife (i.e., at least 50 m).	Protects aquatic and wetlands systems from nonpoint source runoff, provides habitat and movement corridors for wildlife.
Select uneven-aged management practices over even-aged methods when practical.	Uneven-aged management provides more habitat structure and heterogeneity for wildlife.
Manage stands for maximum rotation periods.	Provides for more mature forests and diverse forest structure. Older trees become senescent and become snags that provide food and nesting cavities for many species.
Minimize or prohibit applications of herbicides and pesticides near aquatic areas.	To reduce potential for toxic runoff that can pollute water supplies, potentially harming aquatic species.
Modify skidding practices to avoid erosion damage (i.e., skidding practices that run parallel to or in the streambed).	Reduces destruction of habitat and runoff of sediments into streams.
Prohibit fuelwood harvest, including collection of downed timber and live branches, in all riparian areas.	Source of coarse, woody debris essential for invertebrate habitat and terrestrial and aquatic nutrient cycling. Live branches utilized by nesting birds and helps control soil erosion.
Protect bottomland hardwoods from fire.	Most BLH trees do not have adequate protection against fire and are either seriously damaged or killed. Prevent loss of coarse, woody debris from forest floor.
Avoid dramatic regeneration cuts (e.g., clear-cutting, seedtree, shelterwood).	These cuts have greater impacts on natural plant communities than single-tree or uneven-aged group-selection cuts.

After Thomas et al. (1979), Oakley et al. (1985), Fredrickson and Reid (1990).

larger limbs, buck logs, and pile stems. Integrating limited processing and piling into the felling function can reduce subsequent extraction impacts.

Extraction. Where climate is favorable, logging on frozen ground during colder months or dry ground during warmer months can reduce erosion from road building, skidding, and soil disturbance. Planning (e.g., directional felling of trees) and careful training of loggers, equipment operators, and field personnel can significantly reduce disruption to stream courses (Peterson 1983, Irland 1985). Logged watersheds can be successfully rehabilitated by decommissioning logging roads, removing stream crossings, recontouring slopes, and reestablishing natural-drainage patterns (Harr and Nichols 1993).

Wide tires are an important option for reducing soil and ground-cover disturbance during extraction. Extra-wide tires, 1.3 m and 1.7 m (50 and 68 in.), have been used in the South. Such tires exert about 8.8 kg per sq cm (3 psi) of pressure on the soil and are still relatively maneuverable. They have better flotation and lower damage to the residual stand. Mellgren and Heidersdorf's (1984) list of advantages of extra-wide tires included increased productivity, fuel savings, reduction in ground disturbance, less soil compaction, smaller machine requirements, smoother ride, improved stability, and increased access to timber. Disadvantages were high price, reduced maneuverability, and the need for specialized repair and maintenance equipment. Flexible tracked skidders have been reintroduced; design changes supposedly decrease operating costs to the point that such machines may be cost effective. Advantages of track skidding over tire skidding include lower ground pressure and higher traction. These skidders have been observed to have lower overall soil impacts in peat soils (D. Stewart, Personal Communication, 1996).

Cable systems that are properly implemented have little impact. The best way may be to give the logs a high lift, even to the point of keeping them completely off the ground. Very large, highly mobile yarders may be required. Another requirement may be portable tail holds for quick set up after moving. On large, float tracts with an in-place road system, such a system may be economically feasible. Cable systems may require intermediate supports to keep the logs off the ground. This method may be the only means of removing trees from many sites, except with a helicopter.

Transport. Since roads are more disturbing to soils than harvesting and are expensive to build and maintain, the use of special equipment that can haul on lower-quality roads or transport the wood further without using roads may reduce soil disturbances. Also, central tire inflation systems that allow the use of low-pressure tires on logging trucks can permit them to operate on low-quality roads and reduce road maintenance. Special matting and mat-handling equipment may help access more difficult and low-quality roads, reduce needed earthwork, and reduce residual disturbance (Jackson and Stokes 1991).

Snag management. Snags (standing dead or dying trees) are extremely important as wildlife habitat, and approximately 85 species of North American birds use snags for nesting, roosting, perching, and other activities (Scott et al. 1977). Additionally, sensitive bat species, such as the southeastern myotis, are often found roosting in hollow trees in BLH forests of the southeastern United States (e.g., east Texas BLH (Horner and Mirowsky 1996), mature bald cypress-tupelo swamps in South Carolina (Clark 1994), and floodplain forests of Tennessee (Graves and Harvey 1974)).

Snag management should be considered an essential element of any timber management program in riparian zones. Snag objectives are highly variable, depending on the wildlife species of concern and management goals for the area. Several snag characteristics should be considered when managing forest stands for selected species or communities. These include snag density, forest type, species composition, longevity, and preference by species of wildlife

(Cunningham, Balda, and Gaud 1980). Many species of wildlife require snags situated near water. Waterfowl such as wood ducks (*Aix sponsa)* and hooded mergansers (*Lophodytes cuculatus*) prefer to nest in cavities directly over standing water. Raptors that feed primarily on fish, such as bald eagles and ospreys (*Pandion haliaeetus*), utilize snags near water for nesting, perching, and feeding. A variety of neotropical migrant songbirds use cavities in riparian snags. Where snags are limited, the construction of nest boxes can be an effective management tool.

For many species that depend on snags, the proximity of the snag to water is a determinative factor in choosing sites for nesting or other activities. For example, bottomland hardwoods in oak-hickory forests are especially beneficial not only due to the generally high percentage of cavity nests but also because bottomland tree species tend to grow more quickly than trees on other sites, thus making cavity substrates available sooner (Brawn, Tannenbaum, and Evans 1984). Cline, Berg, and Wight (1980) suggested maintaining old-growth stands as buffer strips within riparian zones to enhance diversity in both aquatic and terrestrial habitats, as well as to protect water quality. Furthermore, since timber harvesting is often restricted in riparian zones, snag management in these areas is appropriate.

Snag management objectives are highly variable, depending heavily on the wildlife species of concern and the resource management goals of the forest. Common factors for analysis include snag abundance, forest types, and species considerations. Generally, the overriding factor affecting the abundance of cavity-dwelling species is snag density. Several researchers have examined snag characteristics for various cavity nesters, which have been used to determine the snag densities necessary to sustain various population levels of cavity-nesting species. Various techniques can be used to determine the density of snags in a stand; once the number of snags is determined, density should be monitored over time to ensure against the loss of snag abundance. This can be accomplished by periodic censusing, or by using predictive models (e.g., Bull and Meslow 1977). However, models require extensive information on the snags in a stand.

Thomas et al. (1979) hypothesized that if the habitat requirements were met for primary nesters, snags would not be a limiting factor for secondary nesters. Under this assumption, a model was constructed using snag requirements of primary nesters to determine snag densities necessary to maintain these nesters at various percentages of their carrying capacities. The number of snags required per 40 ha to sustain populations was determined by the following equation $N = (P)(M)$, where N = the number of snags; P = the potential maximum population; and M = the percent of the population desired to be sustained.

Forest type is of primary importance in determining snag density requirements for cavity users. Forest composition affects such snag characteristics as diameter, height, and rate of decay. Snag diameter should be a major management consideration. While species such as the downy woodpecker (*Picoides pubescens*) and the Carolina chickadee (*Parus carolinensis*) might nest in snags with diameters as small as 15 cm dbh (Evans and Conner 1979), other species will require much larger diameters. For example, the red-headed woodpecker (*Melanerpes erythrocephalus*) requires a 70-cm dbh snag for nesting (Conner 1978). Average nest height, while often variable, should be considered in managing cavity-nesting wildlife. Conner (1978) found pileated woodpeckers (*Dryocopus pileatus*) to nest at least 5 m above the ground.

The tree species available as potential snags is also important and will affect species use. Therefore, preference for particular snag species should be taken into consideration when managing for target wildlife species. Also, some snags form cavities more readily than others. Robb and Bookhout (1995) found that American beech (*Fagus grandifolia*), red maple (*Acer rubrum*), and American sycamore (*Platanus occidentalis*) produced 72 percent of available cavities for wood ducks (*Aix sponsa*) in a study site composed of upland and bottomland hardwood habitats in south-central Indiana; however, these species composed only 28 percent of the basal area of the forest. Bottomland hardwoods in oak-hickory forests are especially beneficial, not only due to the high percentage of cavity nests, but because bottomland trees tend to grow more rapidly than trees on upland sites, thus providing cavity substrates at a faster rate (Brawn, Tannenbaum, and Evans 1984).

The species of wildlife under consideration for management should play an integral role in the management approach taken for snags. Management practices that alter the snag resource to benefit species of concern should include altering rotation length, leaving snags where they would normally be removed, killing trees to create snags, and creating artificial snags and nests. Thomas et al. (1979) suggested managing stands on rotation lengths long enough to permit dominant trees to grow considerably larger than 50.8 cm (20 in.) dbh, the size required for nesting by the pileated woodpecker. Renken and Wiggers (1993) found that large trees and huge snags were important features in pileated woodpecker habitat in Missouri; these features were most often associated with bottomland forests. Study areas in Missouri with greater amounts of bottomland forest typically have greater densities of snags ($\geq$54 cm dbh), which is a critical habitat component for nesting pileated woodpeckers. Evans and Conner (1979) and Renken and Wiggers (1993) recommended managing for an even distribution of huge snags, because a forest with clustered pockets of huge snags will likely not hold as many woodpeckers as a forest with a dense, uniform distribution of snags.

Robb and Bookhout (1995) concluded that management of natural cavities for nesting wood ducks should encourage old-growth (>200-year) stands, including upland forest, within 2.0 km of potential breeding pair and brood habitat. Silvicultural practices should exclude culling of undesirable timber trees (e.g., beech) that provide cavities, and rotation ages should be extended to the economical maximum ($\geq$80 year). Cavity production in branches could be increased through retention of trees in large (>80-cm) dbh classes. Cavities in live trees remain suitable as nest sites longer, thus live trees should be emphasized in forest management for wood ducks (Robb and Bookhout 1995).

Riparian buffer zones and corridors. There is increasing interest in habitat corridors and buffer strips, since retaining a riparian vegetation buffer strip of proper width along water courses can effectively minimize erosion and nonpoint source pollution[1] (NPSP), attenuate stream sedimentation, provide connectivity among habitat patches, provide noise abatement and visual screening, and provide fish and wildlife habitat. Bottomland hardwoods and other riparian communities can serve as effective buffers between uplands and aquatic systems, and numerous studies have addressed the influence of riparian area width in removing or buffering runoff containing nutrients, sediments, and other nonpoint pollutants (e.g., Lowrance et al. 1984, Peterjohn and Correll 1984, Lowrance, Leonard, and Sheridan 1985, Pinay and Decamps 1988, Osborne and Wiley 1988, Delong and Brusven 1991). Unfortunately, when decisions are made to retain buffer strips adjacent to streams, the basis for determining strip width has been almost completely dominated by surface runoff considerations (Harris and Gosselink 1990); few studies have addressed the compatibility of recommended buffer strip widths with other important ecological functions, especially their ability to sustain native faunal and floral species. It should also be noted that although riparian areas are effective buffers from NPSP, they cannot protect aquatic ecosystems from point sources or materials entering waters upstream (Risser 1990).

GIS can aid in the design of buffer zones to minimize stream sedimentation (Hemstrom 1989). Because interactions between aquatic, riparian, and terrestrial ecosystems are a function of valley-floor morphology, digitized GIS data on valley-floor morphology aids in delineation of specific areas where erosion potential is high (e.g., where streams flow through alluvial deposits) or low (e.g., through bedrock). Thus, critical areas for buffer strips can be identified before impacts are visible. Valley-floor morphology typically remains constant over long-time periods, and knowledge of the valley-floor types provides important information regarding types of channels and riparian processes likely to be present in a given area (Hemstrom 1989).

A corridor's suitability as wildlife habitat varies depending on such factors as width, length, degree of fragmentation, and dominant vegetation present. To encourage use by area-sensitive fauna, such as black bears and forest-interior neotropical migrant songbirds, corridors should be as wide as possible, be relatively free from improved roads and human settlements, and contain an abundance of large trees (greater than 84-cm diam, breast height (dbh)) for use as potential bear den sites (Oli, Jacobson, and Leopold 1997). However, establishing corridors that connect two or more large blocks of BLH will not provide equal benefits to all species. For example, a forested corridor 60 to 100 m wide may be heavily used by edge species such as white-tailed deer and wild turkey, but largely ignored by area-sensitive or forest-interior species such as Louisiana waterthrush (*Seiurus motacilla*), pileated woodpecker *(Dryocopus*

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[1] Nonpoint source pollution includes potentially harmful chemicals such as herbicides, insecticides, toxic metals, petroleum products, acid rain, and other direct pollutants of rivers (Brinson 1990). These chemicals may directly or indirectly affect floodplain plants, animals, (including TES) and microbes essential to chemical cycling processes (Brinson 1990).

pileatus), yellow-throated vireo, and prothonotary warbler. Specifically, Harris and O'Meara (1989) reported Louisiana waterthrush did not occur consistently in buffer strips less than 60 m wide; pileated woodpeckers, hairy woodpeckers (*Dendrocopus villosus*), and Acadian flycatchers (*Empidonax virescens*) were rarely observed in buffer strips less than 50 m wide; northern parula warblers occurred only in the widest corridors. Both Hodges and Krementz (1996) and Triquet, McPeek, and McComb (1990) recommended buffer strips of at least 100 m wide to provide habitat for Neotropical migrant birds. Kilgo et al. (1998) investigated breeding bird communities in BLH stands of varying widths in South Carolina and concluded that although narrow strips can support an abundant and diverse avifauna, vegetated buffer zones at least 500 m wide are necessary to maintain the complete avian community of BLH. Managers should consider managing for corridors and buffer strips that are at least 100 m wide. This recommendation applies to either side of the channel in larger river systems and to total width for lower-order streams and rivers.

Habitat width is not always the primary consideration when managing BLH corridors and buffer zones for TES. For example, bald eagles and American alligators (*Alligator mississippiensis*) more readily use relatively narrow BLH corridors as nesting and feeding habitat. Similarly, water level management and protection of rookeries appear to be more important than corridor width for wood stork survival (Kahl 1964).

Changes in Hydrology

Impacts

The historical destruction of wetlands has been so extensive in the United States that all watersheds have been degraded to some degree and few have retained their natural hydrology or productivity (Fredrickson and Reid 1990). Natural hydrographs of most large North American rivers exhibit a rise in water levels due to high-elevation spring snowmelt or fall rains (Rasmussen 1996). The highest primary productivity occurs in BLH that experience seasonal hydrologic pulsing, resulting in habitat that is neither too wet or too dry (Mitsch and Gosselink 1986). These natural-flow regimes are altered when streams and their adjacent watershed are modified. Impacts to streams that may alter hydrology include damming; stream channelization; river constriction; diking and draining; impounding water for flood storage and control, water supply, or hydroelectric power; or diverting water for irrigation. Impacts to watersheds that influence hydrology include row-crop farming, timber harvesting, urbanization, and draining/filling of wetlands (Satterlund and Adams 1992). These impacts have eliminated seasonal inundation of floodplains in many rivers.

The hydroperiod can be altered through clearing of vegetation on uplands (e.g., clearcutting or agricultural conversion). After clearing, the increased runoff from the upland leads to higher frequencies and intensities of flood pulses. Any activity that affects water flow in the watershed may affect the hydrology of the associated bottomland ecosystem (Harris and Gosselink 1990). Such changes in the hydroperiod will affect sedimentation on the floodplain and in the stream. The topography of the floodplain, natural ponding areas or sloughs, downed wood, vegetation, and other sources of surface heterogeneity contribute to sediment retention.

Changes in water regimes often have indirect effects on wildlife by altering the distribution of plant species and substrate materials downstream (Klimas, Martin, and Teaford 1981, Johnson and Carothers 1987). For example, impoundment may trap up to 99 percent of a river's sediment load and stabilize its hydrology (Brinson 1990). This causes downstream downcutting, bank erosion, lowering of the water table and loss of wetland area (Brinson 1990). Impoundment of rivers has been linked to changes in water chemistry (Hannon 1979 in Brinson 1990) and temperature (Fraley 1979 in Brinson 1990). The hydrology of the river segment downstream tends to be stabilized (TNC 1995), which transforms the floodplain forest into an aquatic ecosystem (Brinson 1990) and subsequently alters natural processes of forest regeneration (Green 1947, Hall and Smith 1955, Sharitz and Mitsch 1993). In a natural floodplain ecosystem, water fluctuations are necessary to provide the proper moisture and aeration characteristics for plant regeneration in the following growing season (Broadfoot 1967). In deepwater swamps, changes in hydrology that result in permanent inundation will cause reduced growth and eventual death of water tupelo and cypress trees (Penfound 1949, Eggler and Moore 1961).

Diking and draining for flood control and for conversion to agriculture has altered the hydrology of vast areas of previously forested wetlands, especially along the Mississippi River (Sharitz and Mitsch 1993). This has the effect of preventing deposition of fresh sediment, reducing the nutrient input, and allowing for degradation of the land (Brinson 1990). Increased oxidation of drained organic soil increases the release of carbon to the atmosphere (Lugo, Brinson, and Brown 1990). Drainage of deepwater swamps may cause reestablishment of species that could not tolerate prolonged flooding (Marois and Ewel 1983), reduction in growth rates of trees, and thinning of the overstory canopy (Conner 1994).

Constriction of rivers by levees also causes alteration of the hydrology within the levees, excess sediment deposition, and homogenization of the habitat, resulting in reductions of aquatic animal diversity (Brinson 1990) and nutrient loss (Trush, Conner, and Knight 1989). River channelization deepens, widens, and straightens rivers to improve downstream flow of water from poorly drained areas. This increases the channel slope, causing sharper hydraulic pulses, gulleying, and transport of sediment downstream, resulting in further degradation (Brinson 1990). Channelization, with its attendant destruction of streamside vegetation, increases erosion rates (Thorne and Osman 1988, Trush, Conner, and Knight 1989, Hupp 1992) and is detrimental to riparian wildlife communities. Channelization affects streamside habitats in at least these ways: (a) it alters the structure and/or composition of the vegetation (Hehnke and Stone 1979, Barclay 1980), (b) it reduces the acreage or linear extent of riparian habitat when meandering streamcourses are straightened (Barclay 1980), and (c) it alters the flooding regime, initiating long-term changes in floodplain plant communities (Klimas, Martin, and Teaford 1981). Furthermore, channelization drops the water table and reduces flooding of surrounding lands, which promotes the encroachment of agriculture and urbanization into the riparian area (Barclay 1980).

Greentree reservoirs. Greentree reservoirs (GTR's) are artificially impounded BLH managed primarily for waterfowl. The GTR concept is to flood areas of BLH forest during the dormant period of the trees to provide habitat for migrating and wintering waterfowl, then to draw down the water level again just before the growing season. Although GTR's are, at least temporarily, beneficial for some species of waterfowl (e.g, mallards and wood ducks), it is not clear how subsequent winter floods may influence long-term survival and growth rates of trees (Malac et al. 1981). Flooding in GTR's often does not mimic natural flooding regimes, since these sites are usually flooded earlier and/or later and to a greater depth than would normally occur under natural conditions (Conner 1994). These alterations in hydrology may affect the ecological structure and function of forests (Fredrickson and Batema 1992). Impoundment of water causes low nutrient turnover due to anoxic conditions, nitrogen limitations, and lowered pH during the period of inundation (Brown, Brinson, and Lugo 1979), but flooding for short periods may or may not have a significant effect in the same way. Flooding during the dormant season has caused no effect on tree growth in some short-term studies (Broadfoot 1967, Broadfoot and Williston 1973, Conner 1994). Long-term studies, however, have shown that tree growth may be adversely affected by this type of management (Francis 1983, Schlaegel 1984, Rogers and Sander 1989, King 1995).

Land managers reported that the problems most often associated with GTR's were lower regeneration of seedlings and saplings of desirable species relative to naturally flooded hardwood stands, excessive tree mortality, wind throw, and crown die-back (Wigley and Filer 1989). Because seedlings of floodplain trees in the southeast germinate in early March or later (Streng, Glitzenstein, and Harcombe 1989; Jones et al. 1994), inundation would not directly effect seedlings if drawdown of floodwater were conducted in February. However, managers often are precluded from completing water drawdown by the target date because of beavers and natural flooding, natural brush or log blockage, and human interference (Wigley and Filer 1989). Creation of GTR's has become controversial, and the practice is presently being investigated on at least one military installation (Nelwyn McInnis, Personal Communication, 1996).

Impacts to fauna. Impacts of changes in hydrology have been investigated for several TES. Alligator snapping turtle habitat has been severely impacted throughout much of its range by changes in hydrology resulting from human-induced alterations (e.g., dam construction, channelization and ditching, recreation, pollution) (Pritchard 1989, USFWS 1991). Researchers attribute the decline of the highly colonial wood stork to excessive loss of suitable feeding habitat (shallow depressions in freshwater marshes and swamps, brackish wetlands, narrow tidal creeks, and flooded tidal pools (Kahl 1964)) and to the degradation of cypress and mangrove swamps typically used as nesting sites

(Ogden and Patty 1981). Specifically, man-made levees, floodgates, and canals (primarily in the South Florida Everglades) have greatly changed the hydrology of the region, contributing to the estimated 35 percent decline in potential stork feeding habitat since the turn of this century (Ogden and Patty 1981). Despite protection of known rookeries, wood stork populations in the United States continued to decline nearly 5 percent per year during the 1980s (USFWS 1986).

Rare plant species occurring in BLH may also be sensitive to changes in hydrology. Seeds of herbaceous plants, including rare species, that overwinter in the seedbank may respond positively or negatively to winter impoundment. Any negative impact is important because rare plant species seem to more often reproduce from seed than dominant perennials (Kirkman and Sharitz 1994). Populations of rare herbaceous plants that occur in this habitat are generally described as sensitive to changes in hydrology and may be reduced or eliminated by drastic alterations of the natural hydrologic regime (Kral 1983; TNC 1995; Zettler, Ahuja, and McInnis 1996). The monkey-faced orchid is particularly sensitive to changes in hydrology, especially decreases in the water table (Shea 1992, Zettler, Ahuja, and McInnis 1996). One population of orchids, for example, was eliminated from an area when beaver dams were removed to drain the site. Maintenance of the naturally high moisture level of the soil is important for keeping the understory clear of moisture-intolerant shrubs to provide light levels necessary for this species (Shea 1992) and probably other light-limited floodplain species. Drainage of wet soils has also been cited as a threat to Texas trillium, buckthorn, and false dragon-head (*Physostegia leptophylla*).

Management recommendations

Efforts should first be made to prevent any further alteration of natural hydrologic regimes (e.g., by drainage or unnatural flooding) and then to restore those regimes that already have been altered in BLH and deepwater swamps. Consideration of impacts should include that of both large-scale watershed processes and activities such as logging and use of heavy equipment that influence specific sites. Determination of the proper hydrologic regime may be approached by considering both the needs of the plant and animal populations of interest, including herbaceous and tree components, and researching the historical hydrology of a site. The hydrologic needs of particular herbaceous and shrub plant populations often are not known. Therefore, monitoring populations for their increase or decline in different environmental conditions is critical to making, and possibly reevaluating, management decisions.

Several methods can be used to restore hydrology to a site. Intensive hydrological management may include the use of water control structures to modify water delivery into or out of a site (Malac et al. 1981). This may allow managers to mimic (control) the natural hydroperiod by flooding timber at various frequencies, depths, and durations. Proper water management can allow the restoration of various wetland functions (Drayton and Hook 1989).

Management emphasis should be on restoring natural areas and ecosystem processes rather than on game (sustained yield) or single-species management. In cases where artificial impoundments are planned, the area should be surveyed for rare plant populations that may be impacted or lost to adequately assess the tradeoffs of this management practice.

Greentree Reservoirs. GTRs should not be flooded during consecutive years in the same area because of the potential impacts to regeneration. Rather, Giudice and Ratti (1995) recommend flooding of GTRs once every 2 to 3 years to simulate natural events, promote nutrient cycling, aid seedling establishment, and prevent a species shift toward a more mesic community than would otherwise develop. Survival and regeneration of herbaceous plant populations of concern should be monitored in established plots following purposefully flooded verses unflooded dormant seasons to determine the influence on the viability of the seedbank (Kirkman and Sharitz 1994). Drawdown of flood water should be executed before spring seed dispersal, since these seeds generally have low rates of viability in flooded conditions and require unflooded substrate to become established (Jones et al. 1994). Current management recommendations to maintain or enhance productivity in GTRs include clearcut harvests in small blocks or patches to create openings that promote regeneration and thinnings in the mid- and understory to increase desirable species (Rogers and Sander 1989; Moorhead, Hodges, and Reinecke 1991). However, the adverse effects of clearcutting in these areas should be considered. More research is needed on the influence of GTRs on BLH ecosystems.

Grazing/Animal Damage

Impacts

Livestock grazing is one of the most controversial issues in riparian habitat management; poor management and uncontrolled livestock grazing have caused severe riparian damage and habitat degradation, especially in the western United States. However, grazing is also a serious problem in streamside areas in other regions of the country, including the southeast. Livestock are attracted to riparian areas because of available water, shade, thermal cover, and quality and quantity of forage (Kauffman and Krueger 1984). Although grazing may not be a major problem in most larger BLH communities, it can become a problem in narrower floodplains where the stream channel is proximal to grazing areas. Lowrance and Vellidis (1995) suggested that cattle in BLH can cause loss of and/or damage to native vegetation, directly introduce waste products into the stream channel, reduce infiltration and increase erosion rates, and compact wetland soils through trampling. Cattle also can inhibit regeneration of trees in floodplain forests by eating their seedlings (Brinson 1990). The reduction of herbaceous material may also increase erosion, cause water quality problems (Buckhouse, Skovlin, and Knight 1981), and alter habitat structure for wildlife. Grazing and tuber herbivory by deer and rooting by feral hogs (*Sus scrofa*) have been recognized as threats to populations of the monkey-faced orchid. Deer have also damaged a population of buckthorn at Fort Stewart, GA (TNC 1995).

Management recommendations

Livestock grazing should be prevented in areas managed for rare plants and/or timber production. Use of BLH for grazing should take into consideration the large impact on the herbaceous layer of the community as well as tree regeneration. Where cattle can be given supplemental feeding sites away from BLH, the impact of cattle can be reduced. Managers at Eglin AFB, Florida, recommended extending the season and bag limits to reduce populations of wild hogs, which can cause damage to the community by extensive rooting (FNAI 1994b).

Agricultural Activities

Impacts

Bottomland forests in the southeast have been extensively cleared for agricultural production (Turner, Forsythe, and Craig 1981; Taylor, Cardamone, and Mitsch 1990; Sharitz and Mitsch 1993). Land clearing and channelization associated with agriculture can be especially detrimental to bottomland habitats. Floodplains are often converted to cropland or pasture, which includes removal of natural plant cover and often modification of the hydrology of the area, including both surface and subsurface flow of water. Besides immediate disruption of the riparian zone, agricultural lands themselves may also become degraded, especially from soil erosion and decreased water quality and quantity (Malanson 1993).

In many cases, stream and river channelization to deepen streams and facilitate water movement have reduced flooding and encouraged clearing for agriculture. Alternatively, water-diversion techniques, such as levees and ditches, are constructed to maintain cropland in floodplains. Channelization involves the alteration of rivers and streams by removing natural meanders, clearing streambanks, increasing channel depth and width, and disposing of dredged materials (Taylor, Cardamone, and Mitsch 1990).

When lands adjacent to BLH habitat are converted to agriculture, farming practices can result in a variety of pollutants being introduced into the system. Because of their location on the landscape, BLH and floodplain communities tend to accumulate chemicals from both upslope runoff and flood waters (Sharitz and Mitsch 1993). Point-source pollution is derived from a single source (e.g., industrial waste, municipal-treatment plants, surface-mine drainage, animal confinements) and is concentrated but easily controlled because of restricted-source location. NPSP are derived from large areas and are not well-defined (e.g., pesticides, herbicides, and fertilizers from broad-scale agriculture and urban runoff, silvicultural-induced erosion, mining). They are neither highly concentrated nor easily controlled because of the large source area (Gore and Bryant 1988). Both sources of pollutants are currently a problem in riparian and aquatic ecosystems (Neal and Jemison 1990, National Research Council (NRC) 1992, U.S. Environmental Protection Agency (USEPA) 1992).

NPSP, originating from both agriculture/irrigation runoff and polluted rainwater during precipitation peaks, can have negative impacts on aquatic ecosystems. Agriculture is the single largest contributor to NPSP problems in the United States and is the largest source of impacts to rivers, lakes, and wetlands (USEPA 1992). Croplands and urban lands probably release more NPSP per hectare on average than forests and rangelands (NRC 1992). In 1985, agricultural practices were the primary causes of NPSP in 64 percent of affected river miles (Council of Environmental Quality and the Interagency Advisory Committee on Environmental Trends 1989). Agricultural pesticides are another NPSP to riparian and aquatic ecosystems. Approximately 51 percent of all pesticides used are in agricultural fields and crops (Taylor, Cardamone, and Mitsch 1990). This group of pollutants has not been investigated as thoroughly as sediments, but the transport of pesticides through riparian areas is a function of the particular chemical used (e.g., persistence, adsorbability to soil particles) (Lowrance, Leonard, and Sheridan 1985). Pesticides reaching streams and rivers are absorbed and incorporated into organisms, resulting in biomagnification of toxic substances (NRC 1992).

Many listed herbaceous plants depend on pollinators for sexual reproduction, so if chemical pollution reduces populations of pollinating insects, there could be impacts to the listed populations as well. It is known that reduction in pollinator populations can be correlated with reduced seed set and reproduction in plant populations. Endangered species are particularly vulnerable to extinction when there are no verified pollinators and reproductive mechanisms are unknown (Zettler, Ahuja, and McInnis 1996). Thus, it is important to make observations of pollinators and pollinator status within populations of endangered species. Reduction in numbers of pollinators has been attributed to the decline of the white fringeless orchid (monkey-faced orchid), a former candidate species (C2) for listing by the U.S. Fish and Wildlife Service (Zettler, Ahuja, and McInnis 1996).

Pollutant impacts on seedbank viability is another important yet poorly researched concern to military land managers tasked with risk assessment and rehabilitating damaged wetland communities. It is reasonable to suspect that chronic exposure of seedbanks to pollutants could limit the ability of BLH and swamp communities to naturally revegetate themselves in the wake of fire, changing river course, decreasing water levels, military training, and other environmental perturbations.

Management recommendations

Agriculture is never the dominant land use on military installations, but natural resources managers should be aware that farming practices on lands adjacent to the installation and those upstream of rivers and streams that flow through the installation can have major impacts on water quality. Vegetative buffer strips and forested wetlands are frequently identified as an effective means to reduce the levels of pollutants, organic matter, and nitrogen runoff from both entering into aquatic systems in agricultural regions (Schlosser and Karr 1981, Lowrance et al. 1984, Walbridge and Lockaby 1994). Where agriculture does represent a significant activity on military lands, managers should ensure that adequate buffer zones are retained between cropland and wetland/aquatic resources. In some cases, restoring existing agricultural lands back to forest could eventually provide more revenue through timber harvests or fees from hunting permits. Restored lands also will provide improved wildlife habitat over time.

Military Training

Impacts

Army, Air Force, Marine Corps, and Naval facilities are scattered throughout the southeastern region, and many installations include significant natural resources. Table 9 summarizes a variety of common training activities that potentially impact BLH and riparian communities in the southeast. Activities common to many military installations include the occupation of areas (bivouacking), off-road driving and orienteering, air drops (soldiers/equipment) into maintained drop zones, firing munitions (small arms to large artillery), the construction and maintenance of infrastructure and equipment, physical fitness training, tactical maneuvering, and digging a variety of offensive and defensive trenches. Some heavy earth-moving training activities may be conducted in gravel pits or "constructed" sites but often must be in natural terrain or under the cover of a forested canopy. Munitions of all sizes are fired into designated impact areas. These areas receive frequent contamination from mortar, flares, artillery, and small arms fire and are subjected to frequent ground fires. Impact areas are not used for vehicle maneuvers nor subjected to any off-road use; activities generally are restricted to road and target maintenance.
Clearly, the specific mission(s) of a military installation has implications regarding the type and extent of training activities that occur within BLH communities, and the amount of management effort required to protect, maintain, and restore functional BLH habitat for TES. The following discussion is restricted to water-based training activities likely to occur within BLH and deepwater swamps.

River crossings are an integral part of land warfare. They are a means to project combat power across water obstacles such as lakes, rivers, and swamps and are broadly classified into three categories (Headquarters, Department of the Army (HQDOA) 1978). Hasty crossings are characterized by speed, decentralized control, minimum loss of momentum, use of local materials and existing equipment, and weak enemy forces on opposing banks. Crossing occurs over a broad front. Deliberate crossings are characterized by detailed planning and centralized control, requirement for additional bridging/rafting equipment,

Table 9
Typical Military Activities Potentially Impacting BLH Communities on Military Installations in the Southeastern United States

Activity	Description
Infantry training	In file on established route; moving cross-country; orienteering; attack, escape, and evasion training; hasty river crossing
Tracked, tactical vehicle operation	In file on established route or moving cross-country; crossing streams; tactical maneuvering, tactical concealment
Wheeled, tactical vehicle operation	In file on established route or moving cross-country; crossing streams; tactical maneuvering; transport of petroleum, oils, and lubricants (POL), and other supplies, tactical concealment
River crossing operations	Troop and equipment transport via bridge and watercraft; bridge building; watercraft reconnaissance of enemy positions and stream bottom conditions
Munitions	Small arms firing; medium and heavy weaponry; missile, rocket, and artillery firing; use of incendiary devices
Pollution, intentional and accidental	Tactical use of obscurants and other smoke products, training gases; incidental POL emissions from vehicles, aircraft, boats, and other equipment
Earth-moving activities	Construction of obstacles, fortifications or emplacements; engineer heavy equipment operations
Miscellaneous activities	Firefighting, camouflage, bivouacking, assembly and staging activities

proximity of stronger enemy forces, and the infeasibility of a hasty crossing. Crossings are made at fewer sites along a front. Retrograde crossings are defensive in nature and are characterized by enemy control of maneuver initiative, detailed planning and centralized control, and high risk to friendly forces.

The logistics of crossing any water obstacle can involve one or all of the following methods: swimming, fording, rafting, bridging, and use of assault boats and other watercraft (HQDOA 1988). In general, the three desired characteristics of a crossing site include adequate concealment cover to minimize detection by the enemy, low bank height, and a current velocity of the river of 0 to 1.5 m per sec. Fording of water obstacles requires a firm riverbed and banks with maximum slopes of 50 percent for light to heavy armored vehicles, and 100 percent for foot traffic. Maximum water depths at potential fording sites are 1 m for foot traffic, to 1.10 m for medium/heavy vehicles. Crossing sites where water current exceeds 3 m per sec is generally unacceptable.

Training exercises may require the construction of temporary bridges because of conditions unsuitable for fording or to accommodate a large advancing force. On narrow obstacles, a bridge spanning from bank to bank is most desirable.

However, on many water sites, a floating-type bridge is more appropriate. Light tactical bridges (e.g., ribbon rafts) basically consist of on/off ramps with a series of connectable deck sections in between, all of which are supported by pontoons (pneumatic floats). Bridging materials can be transported to the site via vehicle or helicopter. All military bridges must be anchored to both banks with cables or some other system. The impacts of bridge construction on BLH communities has not been well documented.

BLH and deepwater swamp communities generally are not used for mechanized military activities because of the high density of large trees, mucky soils, and concerns for water quality management. It is likely that a tracked vehicle would not be successful in crossing even a narrow band of floodplain or deepwater swamp, so these communities usually are avoided by mechanized troops and do not experience the soil disturbances and loss of vegetation that mechanized activity can produce (J. Nelson, Botanist, University of South Carolina, personal communication, 10 May 1995; B. Pittman, Community Ecologist, South Carolina Natural Heritage Program, personal communication, 10 May, 1995; R. Stewart, personal communication with Mary Harper and Ann-Marie Trame, 9 May 1995; M. MacRoberts, Botanist, Bog Research, personal communication, 24 July 1995). However, if mechanized activity did occur in a way that altered flow regime, sediment load, vegetation patterns, or groundwater levels, it could have a significant impact on the alluvial forest (Ward 1989). If activities such as mechanized maneuvers remove vegetation and leave unstable, rutted soils, these soils will likely erode into the streams during flood events.

The river floodplain is an open system that is sensitive to events upstream and in adjacent uplands. Activities in adjacent uplands tend to affect small streamside communities more than large bottomlands. Erosion from sandy uplands due to creation of a drop zone, off-road mechanized maneuvers, or occupation exercises may lead to significant sedimentation in smaller streams and bury sensitive wetland plants (A. M. Trame, Ecologist, USACERL, personal communication 1996). Changes to groundwater flow may be less visible but potentially could affect TES plants just as severely. Many bottomland hardwood communities exist on alluvial deposits that are underlain by older permeable strata, which creates a shallow aquifer. Changes in infiltration, percolation, lateral seepage, or subsurface channelized flow due to deep ruts or gully erosion could be damaging to sensitive plant species (Malac et al. 1981).

Alluvial forests can sustain moderate soil impacts from orienteering or cross-country marches. Yorks, West, and Mueller (1993) hypothesized that floodplain species' adaptation to saturated, low-oxygen soil conditions may make them more resilient to stress caused by soil compaction from human activities. However, the understory vegetation may be less resilient to aboveground structural damage than fire-adapted upland species. Alluvial forest understory plants can be strongly associated with specific hydrologic conditions and, thus, can be very sensitive to changes in hydrology (LeBlond, Fussell, and Braswell 1994b). Any activity that creates erosion from uplands or changes soil moisture conditions will threaten TES in floodplain areas. Threats to Chapman's sedge

(*Carex chapmanii)*, southern lady's slipper (*Cypripedium kentuckiense)*, and hairy-peduncled beaked rush (*Rhyncospora crinipes*) included disturbances to soil and hydrology associated with logging activities, vehicle activity, stream crossings, and upslope military training (Hart and Lester 1993; LeBlond, Fussell, and Braswell 1994b; Jordan, Wheaton, and Weiher 1995). Thus, mechanized military training would threaten understory TES if it occurred in or directly adjacent to alluvial forests.

Management recommendations

It is important to capture any sediment in runoff from uplands before it is deposited on the floodplain, to maintain the integrity of alluvial forest communities and to protect stream quality (Hart and Lester 1993). This is particularly important for small streams that are tightly linked to processes occurring in the sandhills community. A buffer zone around small high-quality streams will reduce sedimentation and should reduce changes in groundwater flow from uplands as well. In hilly areas, it may be necessary to avoid vehicle use within the entire drainage area, to the top of the slope and along the stream itself. Transient foot traffic can occur without significant negative impact, but tactical land vehicles should not be permitted to move through these areas. Nearby roads and firebreaks that could lead to erosion and sedimentation should be abandoned and revegetated to the extent possible (LeBlond, Fussell, and Braswell 1994a).

Intensive uses such as occupation and maneuver training are generally impractical in large floodplains due to the mucky soils and high tree densities. However, roadbed stream crossings can be very damaging and should be designed to prevent erosion and ponding (LeBlond, Fussell, and Braswell 1994a). Tracked and wheeled vehicles should stay out of BLH except at limited crossings; training activity can be funneled into a single crossing point, using telephone poles along the ground to direct troop movements. These limited crossings can be hardened with concrete or rock. Check dams can also be used on both sides of the streambed to minimize sedimentation from upslope areas (A. Henry, State Biologist, North Carolina Natural Resources Conservation Service, Personal Communication, 1995). To keep damage localized, degraded areas should be rehabilitated and reused so high-quality natural areas that serve as TES habitat can be avoided altogether (Russo et al. 1993).

5 Bottomland Hardwood Restoration

Although BLH continues to be degraded or converted, a number of state and Federal agencies and private landowners support and/or are actively attempting to restore large acreages via reforestation. Army Regulation 200-3 (HQDOA 1995), for example, states the Department of the Army "...will take a progressive approach towards protecting existing wetlands, rehabilitating degraded wetlands, restoring former wetlands, and creating wetlands in an effort to increase the quality and quantity of the nation's wetlands resource base." Restoration guidance within AR 200-3 intentionally focuses on the *end* more than the *means*, leaving specific restoration methods to the discretion of individual managers who are more aware of unique local conditions. Guidelines on BLH restoration are provided in Haynes and Moore (1988), Hook (1988), and Allen (1990). A recommended approach to restoration planning is summarized in Table 10. Several important considerations for BLH reforestation are discussed below.

When restoring BLH habitats on Army training lands, restoration and other fish and wildlife habitat improvements must be compatible with the installation's Integrated Natural Resource Management Plan (INRMP) (HQDOA 1995). The INRMP is a tripartite agreement among the installation, the state natural resource agency, and the U.S. Fish and Wildlife Service. Compatibility with the INRMP also ensures that restoration management and goals are not in conflict with current or projected military training requirements.

Site Selection

Hydrology, soils, and existing vegetation are the three basic components of wetlands that will affect establishment and growth of desired vegetation. Existing vegetation is a reliable indicator of factors limiting onsite plant growth but may also be a limiting factor itself because of competition with desired plantings. In addition, vegetation establishment is affected by land uses and offsite influences that can create adverse growing conditions (Davis 1993).

Table 10 Recommended Steps for Wetland Reforestation
1. Use the combined expertise of foresters, ecologists, wildlife biologists, soil scientists, hydrologists, and engineers to design and execute creation and restoration ventures.
2. Determine exactly what type of forested wetland is desired.
3. Choose a suitable site (the soils and climate of the site must be compatible with the tree species to be established).
4. Design a hydraulic system that will maintain the desired hydroperiod with minimum energy input.
5. Determine how the new water regime is likely to interact with the local and regional water tables.
6. Determine which method of regeneration will be best for the site (i.e., direct seeding, planting of seedlings, or natural regeneration).
7. Determine the best time to attempt reforestation.
8. Arrange for seed collection of desired species (make sure seeds are compatible with the site and arrange to have seedlings grown in a local nursery).
9. Screen seedlings before planting for desirable root morphology.
10. Remove all competing vegetation and be prepared to control competing vegetation until seedlings are large enough to dominate the site.
11. Manage water regime to favor the desired tree species.
12. Monitor projects and fine-tune where needed.
Source: Hook (1988)

When evaluating potential sites for restoration, managers should identify limiting conditions and collect information that can be used in the wetland project plan development and implementation (Davis 1993). For example, plant species growing on natural wetland sites in the area can provide a good basic list of potential species for use on the project; these species are adapted to local conditions and are most likely to be successfully established and maintained. Baseline site assessments should include the collection of appropriate information on (a) topography (e.g., elevations, slope, presence of depressions, and other physical features), (b) hydrology (water quality and quantity), (c) soils (soil texture, nutrients, pH, cation exchange capacity, erosion potential, and presence of dense layers such as rock, clay, or mineral deposits in the soil profile), (d) vegetation (species dominance and/or abundance for all strata - i.e., canopy, shrub, and herbaceous, maps of existing vegetation associations), (e) wildlife (dominant species, identification of herbivores that may limit restoration efforts), and (f) TES already occupying the area.

Hydrologic surveys of restoration sites should include an assessment of water quantity and quality. It is desirable to plan hydrologic regimes with seasonal water level fluctuations similar to local natural wetlands. This enables the placement of local wetland plant species in hydrologic conditions similar to where they occur naturally. Water quality is a secondary factor that determines wetland plant distributions. Site evaluations of water quality usually include nutrients, pH, alkalinity, and turbidity, as well as salinity and toxins, where appropriate. Water quality parameters are important for defining site-specific conditions for which tolerant plant species must be selected. Since most rooted plants acquire their nutrients from the soil, water chemistry is very important when considering submergent aquatic plants or potential eutrophication problems (Davis 1993).

Baseline site assessments will help determine whether site preparation is necessary and will define which site preparation methods are most appropriate to meet project goals. Site assessments should include historical, physical, chemical, and biological information that must be considered for successful establishment and management of wetland vegetation. A basic familiarity with preproject conditions will improve project plans and the chances of attaining project goals (Davis 1993). Additionally, a better knowledge of existing conditions will allow the manager to evaluate alternative designs and select the plan with the greatest potential for success. Additional guidelines for site selection may be found in Kusler and Kentula (1990), Hammer (1992), and U.S. Department of Agriculture Soil Conservation Service (USDASCS) (1992).

Species Selection

Successful reforestation is directly dependent on the tree species chosen for planting. Factors affecting choice of species include which species are capable of growing on the site, the availability of planting stock, and objectives of reforestation (Allen and Kennedy 1989). Historical accounts of the region, if available, often provide insight into what species naturally occurred there. In a Mississippi project, Allen and Kennedy (1989) emphasized planting mast-producing tree species, particularly oaks and sweet pecan, to maximize benefits towards game species (e.g., deer, turkey, waterfowl, and squirrel). Species selection may just as easily be weighted toward TES if species currently, or could potentially, occupy a restorable site.

Following tree harvest, natural regeneration in most BLH stands will be composed of the same tree species as those in the overstory (Johnson and Shropshire 1983), but relative abundances may differ, depending on site conditions. For BLH, procedures for establishing specific tree species are available (Kennedy 1984; Johnson and Krinard 1988, 1989; Allen and Kennedy 1989). Seeding with desirable species is suggested as a tool to restore BLH communities whose ecological value has been reduced or is known to become reduced by changes in species composition due to logging or other land use. Planting trees after clearcutting allows better control over reestablishment of desired species (Gresham 1985, Conner 1994). In deepwater swamps, satisfactory results have been obtained with cypress but there has been little success in planting tupelo (DeBell et al. 1982). Cypress seedlings, 1 yr old, at least 1 m tall, and larger than 1.25 cm at the root collar, have become established (Faulkner, Zeringque, and Toliver 1985) even in standing water (Conner and Flynn 1989). Replanting of cypress may be useful for habitat restoration and wildlife management in areas that have been cut over for cypress but the cypress have not regenerated, such as large areas within Dare County Air Force Range, North Carolina (Fussel et al. 1995).

Propagation Method

Three general approaches to BLH restoration are natural regeneration, seeding, and direct planting. A successful restoration effort often requires a combination of these techniques. Natural regeneration is the least costly and disruptive to the soil, flora, and fauna and should be considered if sufficient natural regeneration of desired species is occurring or likely to occur. Controlled flooding is one method of promoting natural regeneration; streamflow is the primary mechanism of seed transport and vegetation establishment. Timing of floodwaters is critical, with consideration being given to the short seed viability periods for some species, and avoidance of periods of high noxious weed content (Manci 1989). Sandrik and Crabill (1983) report successful natural regeneration of red maple, wax myrtles (*Myrica cerifera*), and bay species on disturbed bottomland sites in west-central Florida. Conversely, desired plant diversity on 12 National Wildlife Refuges in the southeast was not achieved on reforestation sites through natural regeneration alone (Haynes and Moore 1988).

Direct planting of saplings is often the desired technique to stabilize the soil and speed community succession. However, direct planting can require relatively intensive site preparation and the use of commercial nursery stock and is therefore the more expensive technique. McLeod et al. (1995) reported reasonable success in establishing baldcypress, water tupelo, and green ash plantings in 30 to 60 cm of water in South Carolina using both commercially balled-and-burlaped stock and by placing a bare-root sapling in soil and a burlap bag to lower cost. The use of transplanted cuttings from native plants can also be considered, although those started in a nursery survive better than direct plantings in the field (Anderson and Ohmart 1985).

Traditionally, restoration of a BLH site with oak species has been accomplished with bare-root seedlings or direct seeding with acorns (Humphrey 1993). However, several problems arise when flooding occurs during the planting season: (a) the site may become inaccessible, (b) newly planted seedlings may become inundated, and (c) poor stock quality may result from unavoidable, long-term storage (i.e., propagules may be damaged by mold, mildew, and dry rot). Storage often is unavoidable because nursery operators must harvest seedlings before seedbed preparation for the following year’s crop. If planting occurs before flooding, seedlings must tolerate flooding during the growing season and survive drought conditions during hot summer months. A stock that can be planted after spring floods, yet survive the anticipated summer drought, is needed for successful reforestation of frequently flooded areas (Humphrey 1993).

Container oak seedlings may alleviate planting problems encountered with bare-root seedlings and direct seeding on flooded sites (Humphrey 1993). A major benefit is that growth in containers promotes a more fibrous root system as well as a higher root-to-shoot ratio. The root system of a planted bare-root seedling consists of only a few primary and secondary roots because harvesting and subsequent pruning can result in the loss of a large portion of the roots. In contrast, the root system of a container seedling is bound to the media until planting, resulting in no root damage or loss from harvesting or pruning. This allows the planting of an undisturbed fibrous root system with a large surface area, which increases absorption capacity for water and nutrients in drought conditions and oxygen in hypoxic conditions (Humphrey 1993). The use of container seedlings also allows the planting season to be extended (Graber 1978, Yeiser and Paschke 1987), which provides flexibility in the planting schedule and eliminates storage problems encountered with bare-root seedlings and seed. Seedlings remain in the containers and receive water and nutrients until optimum planting conditions occur (Humphrey 1993).

Potential problems associated with growing container seedlings are maintaining moisture within containers and the leaching of fertilizer. Also, container seedlings have not been commonly used in the South due to high cost and unavailability of large quantities of propagules. Seedlings are now more readily available from a greater number of sources, but the initial cost may still be extremely high. However, the difference in seedling cost must be balanced with the potential for increased survival. Preliminary data from a field study at Lake George, MS, show a 75-percent seedling survival for container stock versus 45 percent for bare-root stock (Humphrey 1993). The selection of a tree species suitable for the site, use of seedlings grown in containers, and an extended planting season may allow the reforestation of frequently flooded sites that otherwise would be difficult or impossible to replant.

Seeding is an alternative to direct planting of commercial stock and is considerably less expensive (Allen and Kennedy 1989), although restoration time is lengthened and the probability of successful regeneration lower (Allen 1990). Johnson (1981) reported satisfactory results after seeding oaks at a rate of 3,705/ha and a spacing of 0.8 m × 3.7 m. Seeds are particularly susceptible to predation by squirrels and other rodents, and thus, should be monitored carefully to detect loss.

Manci (1989) and Haynes and Moore (1988) identified many of the typical factors that contribute to poor restoration success. Some of the more prevalent factors are low native regeneration potential, use of species poorly adapted to the hydrologic regime, late freeze or drought after planting, standing water or high temperatures on sites with young seedlings, and poor-quality seed or nursery stock. Site conditions that can limit plant growth in wetland restoration projects are listed in Table 11. Careful planning can minimize loss and improve the chances of successful stand establishment. Local nurseries should be consulted on the availability of native stock; use locally derived stock whenever possible.

Table 11 **Potential Adverse Site Conditions that Limit Plant Growth in Wetland Restoration Projects**
- Unfavorable season and duration of inundation - Unfavorable water depths - Wind and current action - Excessive turbidity - Unstable substrate - Steep slopes - Compaction and cementation of substrate - Extremes of surface temperature - Low nutrient status - Excessive stoniness and absence of fine, soil forming material - Broken, uneven surfaces - Sheet and gully erosion - High levels of potentially toxic elements - Absence of soil micro-organisms and soil fauna - Presence of invasive or nuisance vegetation - Harmful levels of herbivory
Source: Davis (1993)

Warren, Howard, and White (1994) published a directory of commercial plant sources, sorted by state and city, which may be useful in restoration projects. Regardless of the source, saplings should be large and vigorous enough to compete with other species present on the site (Lea 1988, Clewell and Lea 1990). Manipulation of overstory light levels through limited cutting to benefit present shade-intolerant hardwoods and improve sapling growth may be necessary. Johnson (1978) recommended postplanting cultivation for 1 to 5 years for many species including sweetgum, sycamore, and cottonwood (Johnson 1978). Contrary to prevailing opinion, Reed, Barnett, and McLeod (1995) determined that controlling competitive vegetation was not needed on their study area, believing that selecting the most appropriate species for the hydrologic conditions was the single most important factor in restoring the disturbed bottomland. Mammalian herbivory can significantly affect seedling height and can increase mortality (McLeod et al. 1995); protective measures (e.g., fencing) may be desirable on seasonally flooded sites that typically support high numbers of herbivores.

Pollinator Management

Further research is needed to determine which pollinators visit specific plant TES. For an excellent study providing methods for determining pollinators of a plant species, see the Zettler, Ahuja, and McInnis (1996) study of pollinators of the monkey-faced orchid. Once pollinators have been determined, they may be monitored, and management may be conducted in the proximity of specific populations of concern to favor pollinators. For example, in the case of the monkey-faced orchid, which is pollinated by certain Lepidoptera (butterflies and moths) (Zettler, Ahuja, and McInnis 1996), caterpillars of the desired species may be introduced if it is known what food they prefer. For management of herbaceous plants, it would be advisable to introduce naturally occurring caterpillars that eat tree leaves and would not damage the herbaceous layer. In the case of the monkey-faced orchid, increased light levels resulting from caterpillar herbivory would be beneficial to the plant population. Certain species of plants can be introduced to attract specific pollinators (Compton et al. 1995), but artificial introductions of understory plants may be harmful, unless they are native components of the community. The importance of insects as pollinators underscores the need to consider insect populations in general in management practices, as has been done in management recommendations for Dare County Air Force Range, North Carolina (Fussel et al. 1995).

Monitoring the Progress of Restoration

Bottomland hardwood reforestation is not unlike any other land rehabilitation project; the chances of building a functional association are improved if a comprehensive monitoring program is in place. Clewell and Lea (1990) caution that the relatively slow growth rates of woody plants provide an excellent opportunity for encroachment by invading or nontarget species that can slow or prevent stand establishment. Monitoring not only allows these and other problems to be avoided or corrected early, but the “lessons learned” can save considerable time and money on the next effort. Invariably, there is a need to replace individual plants that do not survive for one reason or another. More importantly, regular inspections are more likely to reveal the colonization or expanded use of the area by TES and other so called “indicator species” associated with the community. Their presence may be viewed as progress toward successful restoration, and their absence may suggest that community recovery has not yet occurred.

The objectives of reforestation projects are rarely stated in quantifiable terms, making it difficult to determine the project’s success. A paired comparison with control sites having similar response potentials and pretreatment characteristics may be the most useful means of evaluating management impacts and restoration progress. Most often, plant survival, species composition, and growth rates are used as measures of short-term success. The Washington State Department of Transportation, for example, determines a wetland restoration project as successful if 50 percent of the vegetation has survived 1 year; restoration after 5 years is determined successful if areal coverage by wetland species is 90 percent of that exhibited by adjacent natural areas (Miller 1988). However, reestablished BLH habitat may take 40 to 60 years to become self-regenerating (Haynes and Moore 1988), indicating that a longer monitoring commitment is necessary. Although many of the aforementioned measures of success focus on floral components, other elements are necessary for functional ecosystems. Specifically, an increasing number of Federal and state agencies are incorporating aquatic vertebrate and macroinvertebrate surveys and various water-quality parameters (e.g., dissolved oxygen, turbidity, dissolved and suspended solids, biological oxygen demand, pH, nitrogen, and metals) to more fully evaluate restoration success and assess wetland functional levels (Manci 1989).

6 Summary

Bottomland hardwoods and deepwater swamps are wetland plant communities dominated by woody species adapted to moist soil conditions or complete saturation or inundation during part or all of the year. These unique habitats contain elements associated with both aquatic and terrestrial systems. They provide a variety of functions, including storage of surface and groundwater, maintaining water quality by filtering out agricultural-based pollutants and sediments from various upstream activities; and providing habitat and movement corridors for a variety of plant and animal species. Hydrological characteristics of the site strongly influence community composition of woody plants, with baldcypress-water tupelo-sweet bay associations typical on seasonally to semipermanently inundated sites, to oak and oak-pine dominated stands on the drier sites. Bottomland hardwood habitats also support social and economic values by providing public recreational opportunities, flood control, commercially valuable timber, and reduction in erosion potential. Approximately 35 percent of animal TES are at least partially dependent on BLH and other riparian habitats, the majority of which are fish, reptiles, and amphibians.

Once widespread and abundant throughout the southeastern United States, BLH communities have steadily declined in distribution and are now considered by many as a threatened ecosystem. BLH habitat occurs on at least 28 DoD installations in the Southeast. Because almost 90 percent of all original BLH has been cleared, it is important to maintain remaining areas in the highest quality condition possible, to maintain vital ecosystem functions and TES habitat.

Although DoD lands have, and will continue to be, managed primarily in support of the military mission, there is opportunity to conserve unique ecosystems and their rare and sensitive species as well. Fortunately, many potentially destructive military training activities are generally restricted or prohibited within BLH habitats. Indirectly, however, training activities can significantly impact these communities through hydrological alteration (e.g., water quality, flow, depth) in upstream areas and at other points within the watershed. Earth-moving, road/levee construction, mechanized training, and other hydrology-altering activities should continually be evaluated at the local and watershed levels to identify and minimize any impacts. Indirect impacts to wildlife and animal TES from other military activities (overflights, smokes/obscurants) appear to be less of a concern than direct habitat alteration, although further research is clearly needed in this regard. Management options provided in this report are designed to support the military mission, local economic growth, and TES conservation through community-based management.

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Appendix A
Personal Communications

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Bottomland hardwood forests (BLH) and deepwater swamps on military installations support multiple uses beyond Department of Defense (DoD) training and testing missions, including threatened and endangered species (TES) conservation, public recreation, and forest commodities production. These unique communities contain elements associated with terrestrial and aquatic systems and are important in terms of ecological functions and social value.

This report documents mission-compatible strategies to manage TES and their habitats on a plant community basis, using methods that apply to multiple species and that have broad applications to military lands located throughout the southeastern United States.

Known occurrences of plant and animal TES associated with BLH habitats on DoD lands are reported, in addition to known and potential impacts to the integrity of BLH communities. Management options are provided within an ecosystem-based, adaptive management context.

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