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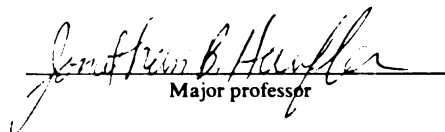


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USING AN ECOLOGICAL CLASSIFICATION
AND WILDLIFE HABITAT MODELS
IN FOREST PLANNING

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Gary John Roloff

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of the requirements for

PhD degree in Wildlife


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**USING AN ECOLOGICAL CLASSIFICATION SYSTEM AND
WILDLIFE HABITAT MODELS IN
FOREST PLANNING**

By

Gary John Roloff

A DISSERTATION

**Submitted to
Michigan State University
in partial fulfillment of the requirements
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ABSTRACT

USING AN ECOLOGICAL CLASSIFICATION SYSTEM AND WILDLIFE HABITAT MODELS IN FOREST PLANNING

By

Gary J. Roloff

The usefulness of integrating an ecological classification system (ECS) and wildlife habitat suitability (HSI) models into vegetation sampling and wildlife habitat assessments was evaluated. Vegetation attributes (required for HSI models) were measured using a variety of plot, line intercept, and subjective evaluation techniques from a stratified random sample design. Sampling was stratified by USDA Forest Service cover type and underlying ecological land type phase grouping (ELTPG)(a sub-level of the ECS hierarchy). Replicates were also categorized by: 1) Forest Service cover type, and 2) ELTPG. Relative efficiency tests were used to evaluate sampling efficiency between the 3 stratification schemes.

Results of the relative efficiency tests between cover type-ELTPG and cover type stratifications indicated ($P \leq 0.10$) that sampling precision generally increased when stratified by cover type-ELTPG. Overall, the ECS appears to be a promising tool for reducing the variance associated with sampling vegetation attributes.

Wildlife population indices for 11 species, including ruffed grouse (*Bonasa umbellus*), pileated woodpecker (*Dryocopus pileatus*), ovenbird (*Seiurus aurocapillus*), American redstart (*Setophagus ruticilla*), black-throated green warbler (*Dendroica virens*), eastern meadowlark (*Sturnella magna*), upland sandpiper (*Bartramia longicauda*), white-tailed deer (*Odocoileus virginianus*), gray squirrel (*Sciurus carolinensis*), wood duck (*Aix sponsa*), and eastern bluebird (*Sialia sialis*), were collected at systematically located points during 1991 and 1992. Cover type and ELTPG maps were digitized and HSI models were programmed into a

geographic information system (GIS); all of which were linked to the cover type-ELTPG database thereby automating HSI computations. HSI values were calculated for each sample point at home-range sized spatial scales, and Spearman's Rank Correlation tests were performed between population indices and associated HSI values. HSI values were also computed at other spatial scales (0.3-182.4 ha) to determine the appropriate spatial scale of model applicability.

Correlation coefficients varied by species and ranged from -0.427 to 0.644 and -0.223 to 0.514 for 1991 and 1992, respectively. For HSI models combined across years ($n=8$), all exhibited significant ($P \leq 0.10$), positive correlation coefficients, ranging from 0.131 to 0.307. The spatial scale at which the HSI models should be applied corresponded to a biologically meaningful scale (e.g. home range).

The integration and application of ECS and HSI models as coarse filter-fine filter approaches to landscape level planning are demonstrated. Implications for forest management are also discussed.

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Typical dissertation acknowledgements begin with gratitude for the funding agencies or faculty involved on the graduate committee. Although I am deeply appreciative of these contributions, I choose to begin my acknowledgements by thanking my wife and children. Thank you for 10 years of support, encouragement, understanding, and love. We were warned it would be tough, and indeed it was. There are few women who could raise a family through field seasons; solitary holidays, weekends, and evenings; graduate student salaries; and university housing and continue to support their husband's dreams. From the bottom of my heart, I thank you Kimberly for understanding. To my children, Joshua, Cassondra, and Jordan, the time has come for you to have a "real" father.

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project, and I also thank him for his friendship. I appreciated and enjoyed our discussions. Although not part of my graduate committee, I would like to thank Dr. Henry Campa, III, for his advice, support, and friendship. In Dr. Haufler's absence, Dr. Campa was always available. I sincerely hope to continue collaborating and conversing with all of you, I am an immensely better biologist and person because of our interactions.

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Thanks to Bill Enslin, Michigan State Center for Remote Sensing, for his cooperation in obtaining geographic information system (GIS) coverages. Also, thanks to Ryan Simmons from the Michigan State Computer Center for his GIS technical support. Thanks are also due Peter O'Grady and Phil Huber, USDA Forest Service, Huron-Manistee National Forest, for their assistance and advise in data collection, management, and analysis.

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Heartfelt gratitude is due my father, James Roloff, and my grandfather, the late John Golz. In addition to exposing me to the outdoors, they taught me to be honest, hard-working, sincere, and forthright. At times I stray from their teachings, but their influence is undoubtedly expressed in this document. Long before I read the "Land Ethic" by Aldo Leopold, these individuals taught me to respect the land, recognizing that we were just one cog in the environmental wheel. They are truly good men, and I will continue to emulate their values.

Lastly, I wish to thank my parents, James and Jane Roloff. Thanks for tolerating the often bizarre behavior of a wildlife biologist in your home. I promise not to bring any more dead animals home! Your encouragement and support were invaluable. This document is as much a part of you as it is me, thank you.

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INTRODUCTION

Since the mid-1970's, legislation and social demands have necessitated dynamic, ecosystem-based natural resource plans. As a result, the U.S. Forest Service was mandated to manage for multiple-resource use with specific directives to maintain or improve biological diversity at the genetic, species, and ecosystem levels (Hunter 1990). The feature of "multiple-use" or "biodiversity" management that set it apart from the traditional single-resource, sustained yield paradigm of forestry is an ecosystem perspective (Behan 1989). Despite the general acceptance of "ecosystem" management, the concept remains ambiguous, and the slow development of ecological analysis techniques reflects the incomplete understanding of ecosystem structure and function (Hawkes et al. 1983, Hunter 1990).

To employ ecosystem management, we must be able to: 1) measure or estimate inputs and outputs of ecosystem materials, 2) describe processes involved in regulating ecosystems, 3) simulate these processes and their results at various spatial and temporal scales, and 4) respond to rapid changes in social, economic, and political forces (Hawkes et al. 1983, Boyce 1985, Kaufmann and Landsberg 1990). One of the major problems facing forest and wildlife professionals in meeting the demands of ecosystem management is incorporating these concepts into forest plans. Forest plans must (by law) address minimum legal wildlife conditions (Mealey et al. 1982, Gokee and Joyce 1992), and thus, tools are needed that evaluate the impacts of forest management policies on wildlife populations (Farmer et al. 1982, Mello and Haufler 1983).

Maintaining wildlife populations as viable, natural units requires management, and sometimes protection, of the entire support system. Yet, effective field applications of ecosystem-based wildlife habitat management is still in its infancy, although numerous techniques have been developed to estimate habitat conditions for resource planning (see reviews in Hawkes et al. 1983, Morrison et al. 1992). Wildlife-habitat models and ecological classification systems (ECS's) have the potential to provide ecosystem-based planning tools with predictive capabilities. Procedures to rapidly determine relative values of land as wildlife habitat are required by decision-makers, and wildlife-habitat models provide quantitative habitat values that relate to wildlife abundance or productivity and can also be used for monitoring the implications of management alternatives. Since integrated ECS's are simplified ecosystem maps, they appear to be well suited as a component of ecosystem-based wildlife habitat mapping (Kerr 1986, Hunter 1990). The ECS also provides predictive capabilities of ecosystem dynamics (e.g., succession).

Recent advances in computer technology have made geographic information systems (GIS's) available to resource planners. A GIS is capable of displaying, managing, and analyzing geo-referenced spatial data. With the increased need for location-specific, analytical, multi-purpose information bases (Davis 1980), the use of GIS's for ecosystem management is imminent.

The goal of this research was to evaluate the utility and precision of combining wildlife-habitat models (U.S. Fish and Wildlife Service 1981; Schamberger and Krohn 1982) with an ECS (Barnes et al. 1982, Pregitzer et al. 1983, Pregitzer and Barnes 1984, and Spies and Barnes 1985) for forest planning using a GIS. Justifiably, concerns regarding the usefulness of incorporating ECS into wildlife habitat assessments and wildlife-habitat model validity were expressed by management personnel. Therefore, the overall objectives of this

study were to:

1. evaluate the use of ECS in habitat evaluation and management,
2. evaluate the utility of wildlife-habitat models for forest planning, and
3. demonstrate landscape (forest-wide) applications of this approach.

Specific objectives are outlined in each chapter.

This document consists of 3 chapters. Chapter 1 evaluates the effects of incorporating an ECS into measures of vegetation structure and composition. Specifically, sampling precision was compared for 3 landscape classification schemes. In Chapter 2, the validity of wildlife habitat models is evaluated. Also, the appropriate spatial scale(s) at which to apply habitat models is documented. A conceptual approach of integrating ECS and wildlife habitat models into forest planning across a landscape is presented in Chapter 3.

STUDY AREA

Study sites were denoted as "Hoist Lakes Recreation Area & vicinity" (Hoist Lakes), "Reed Lake Recreation Area" (Reed Lake), and "Sandpiper Fields" and totaled approximately 6,000 ha (15,000 ac) in southwest Alcona County, Michigan (Fig. 1). Portions of Hoist Lakes and Reed Lake are currently designated "non-motorized, semi-primitive" and the last timber harvest occurred in 1992. The U.S.D.A. Forest Service manages these sites and emphasizes providing visitors with remote, recreational experiences (Lantz 1976).

The Sandpiper Fields are part of the Forest Service's Kirtland's warbler (*Dendroica kirtlandii*) management areas. These areas are characteristically jack pine (*Pinus banksiana*) plains subjected to frequent disturbance. The Forest Service emphasizes habitat management for Kirtland's Warbler which typically involves whole-tree harvests followed by jack pine planting.

The climate of this region is temperate. Average winter and summer temperatures are -6.6°C and 21°C, respectively (U.S. Department Commerce 1979). Annual precipitation ranges from 71-81 cm during a normal year, and the greatest amounts of precipitation occur during May, June, and September (U.S. Department Commerce 1979). Average annual snowfall accumulation is approximately 152 cm (U.S. Department Commerce 1979).

Landforms of the areas are characteristic of changes caused by continental glaciation. The study sites are located in the morainal outwash plains (Sommers 1984). Relatively steep, hilly moraines predominate the northern latitudes, whereas flatter, outwash plains are found

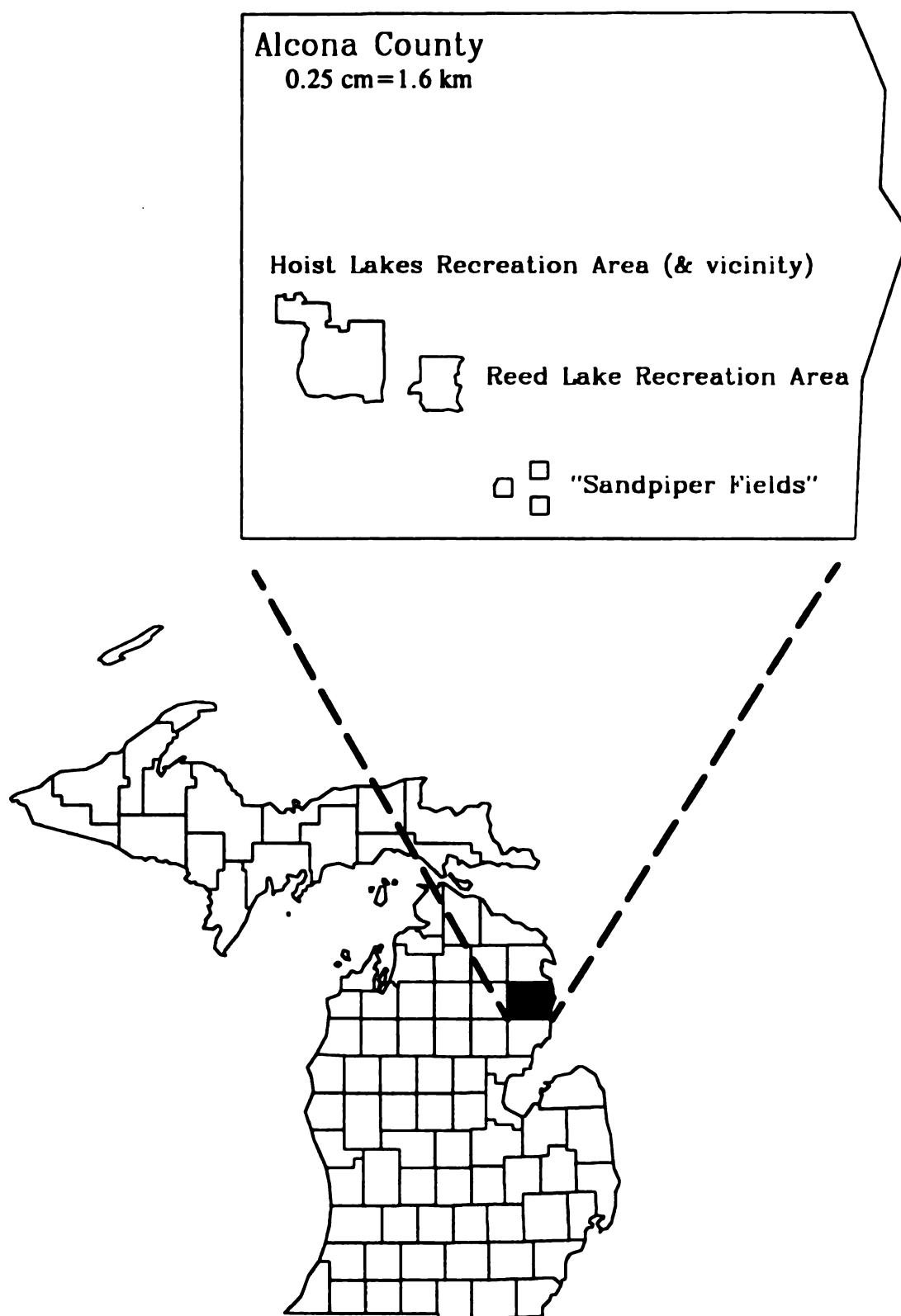


Figure 1. Location of study sites, Alcona County, Michigan.

progressively south (Lantz 1976). The underlying bedrock is composed of shale, limestone, and sandstone (Sommers 1984).

Soils typically have high sand concentrations of the Grayling-Rubicon series (Lantz 1976, Sommers 1984). Some sites, particularly Reed Lake and the northern part of Hoist Lakes, consist of a sandy-clay to loam-clay layer, causing variation in site fertility (Lantz 1976). Overall, the soils are classified as low in moisture content, low in humus, medium acid, and low in fertility (Lantz 1976). Isolated areas of peat and muck are also distributed throughout the areas.

Different vegetation associations are interspersed throughout the areas as results of past forest management and soil conditions. Major vegetation associations (Forest Service vegetation classification) include (in decreasing order of predominance) quaking aspen (*Populus tremuloides*), northern red oak (*Quercus rubra*), jack pine (*Pinus banksiana*), mixed oaks (*Quercus* spp.), sugar maple/beech (*Acer saccharum*/*Fagus grandifolia*), big-tooth aspen (*P. grandidentata*), and red pine (*P. resinosa*).

Understory vegetation consists of numerous species including dogwoods (*Cornus* spp.), viburnums (*Viburnum* spp.), cherries (*Prunus* spp.), witch hazel (*Hamamelis virginiana*), beaked hazel (*Corylus rostrata*), and serviceberry (*Amelanchier* spp.). Ground cover is similarly diverse with abundant species including blueberries (*Vaccinium* spp.), brambles (*Rubus* spp.), bracken fern (*Pteridium aquilina*), and sedges (*Carex* spp.).

Chapter 1: Using an Ecological Classification System in Vegetation Sampling

From 1900 to the late 1960's, forest management in the United States was concerned mostly with fire protection and the efficient harvesting of timber (Carmean 1975).

Throughout the 1970's and 80's, however, an increasing environmental awareness by the public and government confirmed the need for a structured ecological database to meet multiple use/biodiversity forest objectives. This "new" direction of forestry focused on the idea that complex ecosystems could be described by structure and how the component parts were connected (Boyce 1985).

Ecosystem components include climate, landform, soils, biota, and associated ecological processes (e.g., nutrient cycling, competition). The importance of individual components to resource management has been documented; including relationships between soils and forest productivity (see reviews in Carmean 1975, Harvey and Neuenschwander 1991), soils and wildlife (e.g., Albrecht 1944, Denny 1944, Crawford 1950, Hundley 1959), topography and vegetation (Pregitzer et al. 1983), and climate and vegetation (Moore 1976, Brubaker and Cook 1983). These single-component classifications proved to be ineffective multiple-use/biodiversity management tools (Lewis 1967, Van Lear and Hosner 1967, Broadfoot 1969, Murphy and Porath 1969, Carmean 1975, Barnes et al. 1982, Jones 1989), thus, the need for multi-factor, ecological approaches to resource management was imminent.

It has long been recognized that ecosystems represent a combination and integration of all biotic and abiotic elements (Tansley 1935). As a means of addressing ecosystem organization for resource management, ecological classification systems (ECS's) have received considerable support (Hirsch et al. 1978, Nelson et al. 1978, Rowe 1978, Barnes et al. 1982, Bailey 1985, Spies and Barnes 1985, Bazzaz and Sipe 1987, Swanson et al. 1988, Barnes 1989, Jones 1989). An ECS unites land elements; including vegetation, soils, water, climate, and landform; to form a coordinated entity (Hirsch et al. 1978, Rowe 1978, Driscoll et al. 1984, Pregitzer and Barnes 1984, Bailey 1987) capable of predicting ecosystem processes (e.g., succession; Host et al. 1987, Jones 1989, Johnson et al. 1991).

Wildlife populations are subject to limits imposed by ecosystems (Farmer et al. 1982), yet ecosystem-based approaches to wildlife habitat management have been rare. Single-factor classifications have historically been used by managers to quantify and display wildlife habitat (e.g., "cover" type maps), but due to previous silviculture activities, land use practices, and disruptions in disturbance regimes, natural relationships between ecosystem components have been disassociated. Also, existing forest inventories seldom contain the information necessary to effectively manage wildlife habitat. For example, 53 input variables from 12 wildlife habitat suitability models reviewed for this study were related to vegetation structure or composition. The Forest Service's Corporate Database System could only account for 36% of these input variables.

The ECS provides a multi-factor, ecosystem-based tool that can be used in wildlife habitat assessment. Numerous examples of ECS development and use in resource planning have been published (e.g., Wiken 1982, Van Lear and Jones 1987, Mengel and Tew 1991, Hornbeck and Swank 1992), but no studies have quantified the usefulness of ECS for wildlife habitat management. The goal of this chapter is to evaluate the use of ECS in habitat

evaluation and management. Specific objectives are:

1. to compare sampling precision of wildlife habitat data stratified by: 1) Forest Service cover typing and an underlying ECS, and 2) Forest Service cover typing.
2. to evaluate the use of ECS to stratify wildlife habitat sampling.

METHODS

Ecological Classification System

The ECS for this study was founded on the Baden-Württemberg model used in Germany and is based on the works of Barnes et al. (1982), Pregitzer et al. (1983), Pregitzer and Barnes (1984), Spies and Barnes (1985), and Cleland et al. (1993). Early emphasis (early 1970's) of this ECS was to obtain relatively rapid, inexpensive inventories for broad level planning purposes (Russell and Jordan 1991). Forest plan implementation, however, required a finer level of land and ecosystem stratification, thus, ecological land type phases were delineated for operational forest planning (Russell and Jordan 1991). For this study, similar ecological land type phases (as delineated by ECS field manuals and Cleland et al. (1993)) were grouped into "ecological land type phase groupings" (ELTPG's) (that typically encompassed multiple vegetation types) (Table 1). ELTPG's used in this study were intermediate to the ecological land type phases and ecological land types delineated in Cleland et al. (1993) (Table 1; Appendix A). The finest level of ecological stratification was the ecological land type phase, however, field logistics did not permit the acquisition of suitable sample sizes at this level. The intermediate ELTPG stratification was chosen as a spatial and logistic compromise between using ecological land type phases or ecological land types. The statistical procedures employed by Cleland et al. (1993) to delineate the ECS support the contention that some ecological land type phases were similar and could be combined without

Table 1. Components (terminology and numeric coding following Cleland et al. (1993)) of ecological land type phase groupings (ELTPG's) used in this study (also see Appendix A for explicit definitions).

Ecological Land Type Phase Grouping (ELTPG)	Cleland et al. (1993)	
	Ecological Land Type Phase(s)	Ecological Land Type
Outwash I	1	Outwash Plains
Outwash II	10,11	Outwash Plains
Outwash; Ice	13	Outwash Plains
Ice; Moraine I	20,21	Dry Ice-Contact and Sand Hills
Ice; Moraine II	23,25	Dry Ice-Contact and Sand Hills
Moraine I	30,31,33	Mesic Ice-Contact Sand Hills
Moraine II	35	Mesic Ice-Contact Sand Hills
Moraine III	37,38	Mesic Ice-Contact Sand Hills

significant effects on the sample variance of vegetative parameters (pers. comm., David Cleland, Lake States Area Ecologist, U.S. Forest Service, Rhinelander, WI). The relatively smaller spatial scales obtained by grouping similar ecological land type phases (typically 1 to 10's of contiguous hectares) as opposed to stratifying by ecological land types (10's to 100's of hectares) was also more conducive to quantifying vegetative structures for operational level assessments.

Eight ELTPG's were sampled ranging from "white oak (*Quercus alba*)/*Deschampsia* on well-drained sands of outwash plains" (termed Outwash I) to "red oak-red maple (*Acer rubrum*)/*Desmodium* on moderately well-drained sandy loams" (Moraine III; Fig.2). Similar ecological land type phase(s) within each ELTPG contained like plant associations and overall soil characteristics. Often, primary differences between similar ecological land type phases were soil texture and morphology in the substratum (Appendix A). Numerous Corporate Database cover types were typically sampled in each ELTPG (Fig. 2).

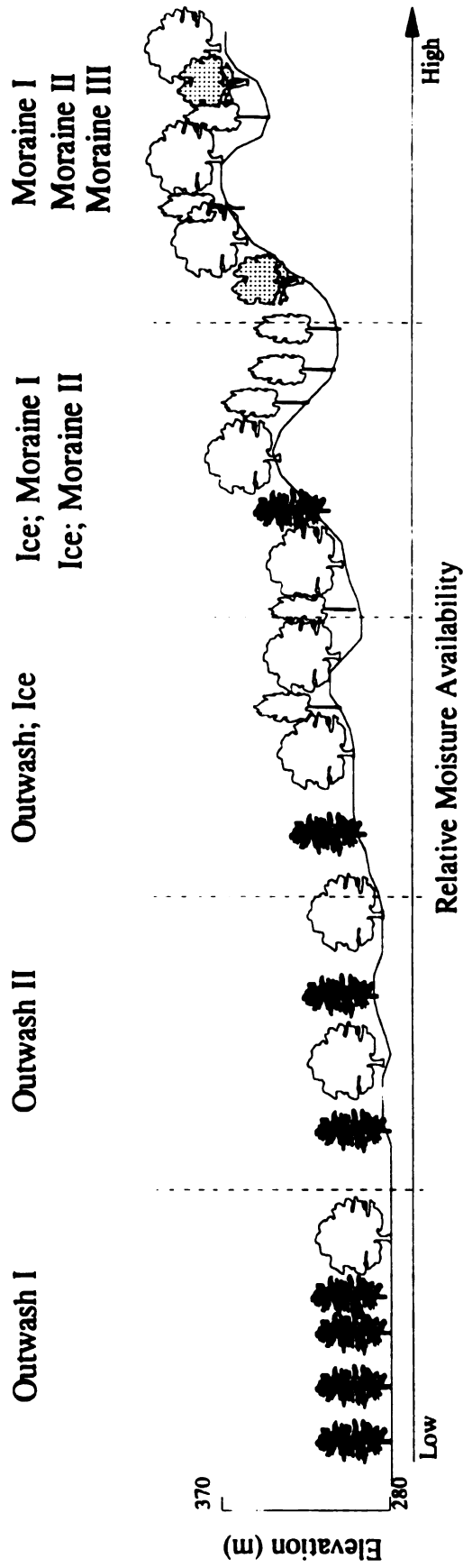
Vegetative Cover Typing

The U.S. Forest Service's Corporate Database vegetation classification was used. The Corporate Database system classifies vegetation into "stands" according to dominant commercial tree species, stocking density, and age (U.S. Forest Service 1988). Nine cover types at various stocking densities were sampled for this analysis (Fig. 2).

Sampling and Analyses

Vegetation sampling was conducted from 1989-1992 throughout the Alcona County study sites (Fig. 1). Vegetation variables (as delineated by wildlife habitat suitability models: Table 2) were measured using a variety of plot, line intercept, pacing, and subjective evaluation techniques (Table 2; see Hays et al. 1981) from randomly located points within

Ecological Land Type Phase Groupings & Commonly Associated Overstory



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Forest Service Corporate Database Cover Types Sampled

<i>Pinus banksiana</i>	<i>Pinus banksiana</i>	<i>Pinus banksiana</i>	<i>Pinus banksiana</i>
<i>Pinus resinosa</i>	<i>Pinus resinosa</i>	<i>Pinus resinosa</i>	<i>Pinus resinosa</i>
<i>Quercus velutina</i>	<i>Quercus rubra</i>	<i>Quercus velutina</i>	<i>Quercus velutina</i>
	<i>Populus tremuloides</i>	<i>Quercus rubra</i>	<i>Quercus rubra</i>
		<i>Quercus spp. (mixed)</i>	<i>Quercus spp. (mixed)</i>
		<i>Fagus/Acer spp.</i>	<i>Fagus/Acer spp.</i>
		<i>Populus tremuloides</i>	<i>Populus tremuloides</i>
		<i>Populus grandidentata</i>	<i>Populus grandidentata</i>
		Openings	Openings

Figure 2. Ecological land type phase groupings, naturally associated overstory, and corresponding Forest Service Corporate Database cover types sampled.

Table 2. Summary of vegetation parameters, associated sampling techniques (see Hays et al. 1981), and data transformations (Sokal and Rohlf 1981).

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Vegetation Parameters	Sampling Technique		Transformation
Overstory¹			
Total Basal Area	10 X 25 m (0.06 ac)	Plot	None
DBH ²	10 X 25 m (0.06 ac)	Plot; Biltmore Stick	None
Height	10 X 25 m (0.06 ac)	Plot; Haga Altimeter	None
No. Tree Species	10 X 25 m (0.06 ac)	Plot	None
Density	10 X 25 m (0.06 ac)	Plot	None
Understory¹			
Percent Cover > 3 m Tall	10 X 25 m (0.06 ac)	Plot; Ocular Estimation	Arcsine-Square Root
Percent Cover > 5 m Tall	10 X 25 m (0.06 ac)	Plot; Ocular Estimation	Arcsine-Cubed Root
Deciduous Height	10 X 25 m (0.06 ac)	Plot; Ocular Estimation	Square Root
Density	2 X 25 m (0.01 ac)	Plot	Cubed Root
Deciduous Density	2 X 25 m (0.01 ac)	Plot	Cubed Root
Percent Soft Mast Cover	25 m (82 ft)	Line Intercept	Arcsine-Square Root
No. Soft Mast Species	25 m (82 ft)	Line Intercept	None
Dead & Downed¹			
Snag Density (≥ 11.4 cm DBH)	10 X 100 m (0.26 ac)	Plot	Cubed Root
Snag DBH	10 X 100 m (0.26 ac)	Plot	Square Root
Log Density	10 X 100 m (0.26 ac)	Plot	Logarithmic (Base 10)

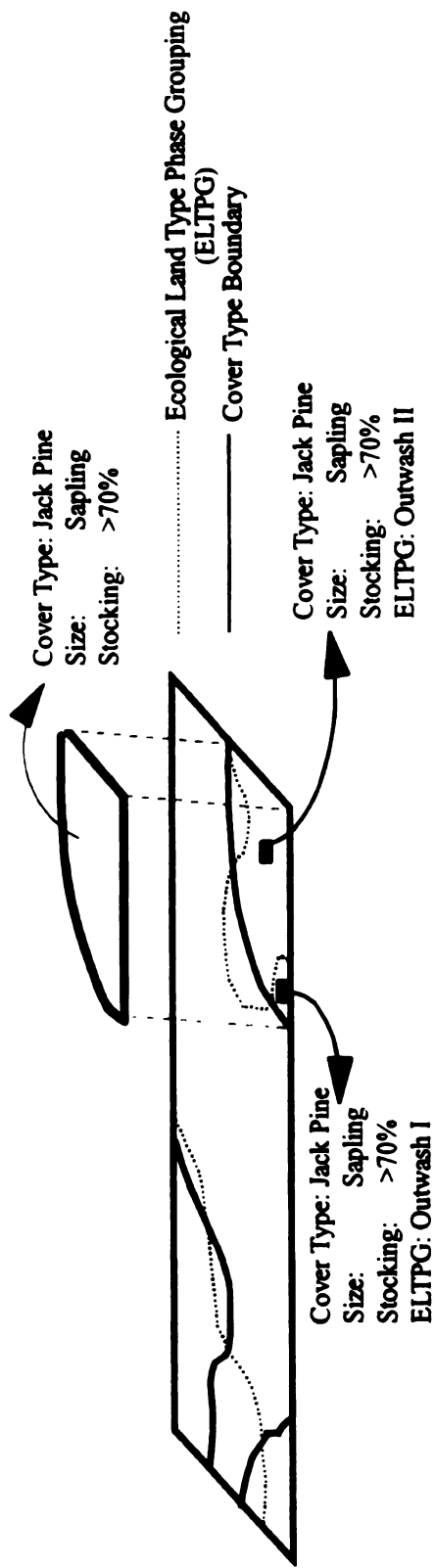
¹ Overstory = All trees ≥ 11.4 cm (4.5 in) DBH; Understory = All woody stems < 11.4 cm (4.5 in) DBH; Dead and Downed = Logs ≥ 18 cm (7.1 in) DBH, free standing snags ≥ 11.4 cm (4.5 in) DBH.

² DBH = Diameter at breast height.

cover type-ELTPG combinations. Each sample point was verified as to cover type and ELTPG. Throughout the analyses, Forest Service cover types on ELTPG's were the replicates, and each replicate had a minimum of 3 samples. Vegetation variables were thus described by means. Only non-zero measures of vegetation variables (excluding count data and percentages) were used in the analyses, consistent with Forest Service data management (pers. comm., Peter O'Grady, U.S.D.A. Forest Service, Systems Analyst, Cadillac, MI).

The parametric assumption of data normality was evaluated for each vegetation variable using frequency distributions, probability plots (Sokal and Rohlf 1981), and Shapiro-Wilk's W test ($P \leq 0.05$) (SAS Institute Inc. 1985, D'Agostino et al. 1990). Parameters that exhibited non-normal distributions were analyzed for outliers using Dixon's test statistic ($n \leq 25$) or Grubb's test statistic ($n > 25$; $P \leq 0.05$) (Sokal and Rohlf 1981). Data transformations were performed as needed to normalize vegetation parameters (Sokal and Rohlf 1981) (Table 2).

Parametric relative efficiency tests (Sokal and Rohlf 1981) were performed to evaluate sampling precision. Relative efficiency tests evaluate ratios of sample variances (Sokal and Rohlf 1981), with the premise that lower sample variances equate to more precise estimates. Precision was evaluated between samples obtained by: 1) Forest Service cover type and ELTPG, 2) Forest Service cover type (Fig. 3), and 3) ELTPG. Two aspects of the relative efficiency tests were used: 1) the magnitude of the variance ratio (" > 1 " indicated a gain in precision), and 2) the significance level. Significance tests were one-tailed ($P \leq 0.1$; $df = n-1, n(s-1)$; where n =replicates and s =samples within replicates) and tested the null hypothesis that samples obtained by Forest Service cover type exhibited precision equal to the precision of samples obtained by cover type and ELTPG



Replicate Units	Sample Variance	F-ratios	F-probabilities
Forest Service Cover Type: Jack Pine, Sapling, >70% Stocking			
Canopy Cover	$s^2(\text{cover type})$		
Canopy Height			
etc...			
Forest Service Cover Type & ELTPG: Jack Pine, Sapling, >70%, Outwash I			
Canopy Cover	$s^2(\text{cover type \& Outwash I})$	$\frac{s^2(\text{cover type})}{s^2(\text{cover type \& Outwash I})}$	F(prob)
Canopy Height			
etc...			
Jack Pine, Sapling, >70%, Outwash II			
Canopy Cover	$s^2(\text{cover type \& Outwash II})$	$\frac{s^2(\text{cover type})}{s^2(\text{cover type \& Outwash II})}$	F(prob)
Canopy Height			
etc...			

Figure 3. Example of resource planning map and associated relative efficiency analysis, where s^2 denotes sample variance.

combinations.

Relative efficiency tests were also performed between samples stratified by cover type or ELTPG. For this analysis, more replicates were typically obtained for ELTPG's as opposed to cover types (Fig. 4). To standardize the effects of sample size on estimates of variance, replicates were randomly selected from the ELTPG database (Fig. 4). The number of replicates within each ELTPG sub-set corresponded to the number of cover type replicates in the comparison (Fig. 4). This process was conducted 10 times for each vegetation variable to minimize the effects of outlier ELTPG replicates. Thus, within each vegetation variable, 10 relative efficiency tests per cover type were performed, resulting in 10 f-probabilities per cover type (Fig. 4). The f-probabilities were subjected to a statistical technique that combines numerous probabilities into one probability that is distributed as a chi-square statistic (see "Miscellaneous Methods" in Sokal and Rohlf 1981)(Fig. 4). The magnitude of the chi-square probability was used to determine significance, and gains in sampling precision could be determined by comparing the calculated probability to a chi-square probability derived from 10 F-ratios equal to 1 (i.e. if the F-ratio is ≤ 1 , sampling precision was not gained).

Vegetation parameters were grouped according to overstory, understory, and dead and downed. Overstory was defined as woody species ≥ 11.4 cm (4.5 in.) diameter breast height (DBH) and understory was woody species < 11.4 cm DBH. Dead and downed material included snags (standing, visibly dead overstory) and logs (≥ 17.8 cm (7 in.) DBH).

RESULTS

The number of replicates sampled ranged from 3-11 for cover type-ELTPG combinations, 6-17 for cover types, and from 3-49 for ELTPG's (Table 3). Univariate

Replicates

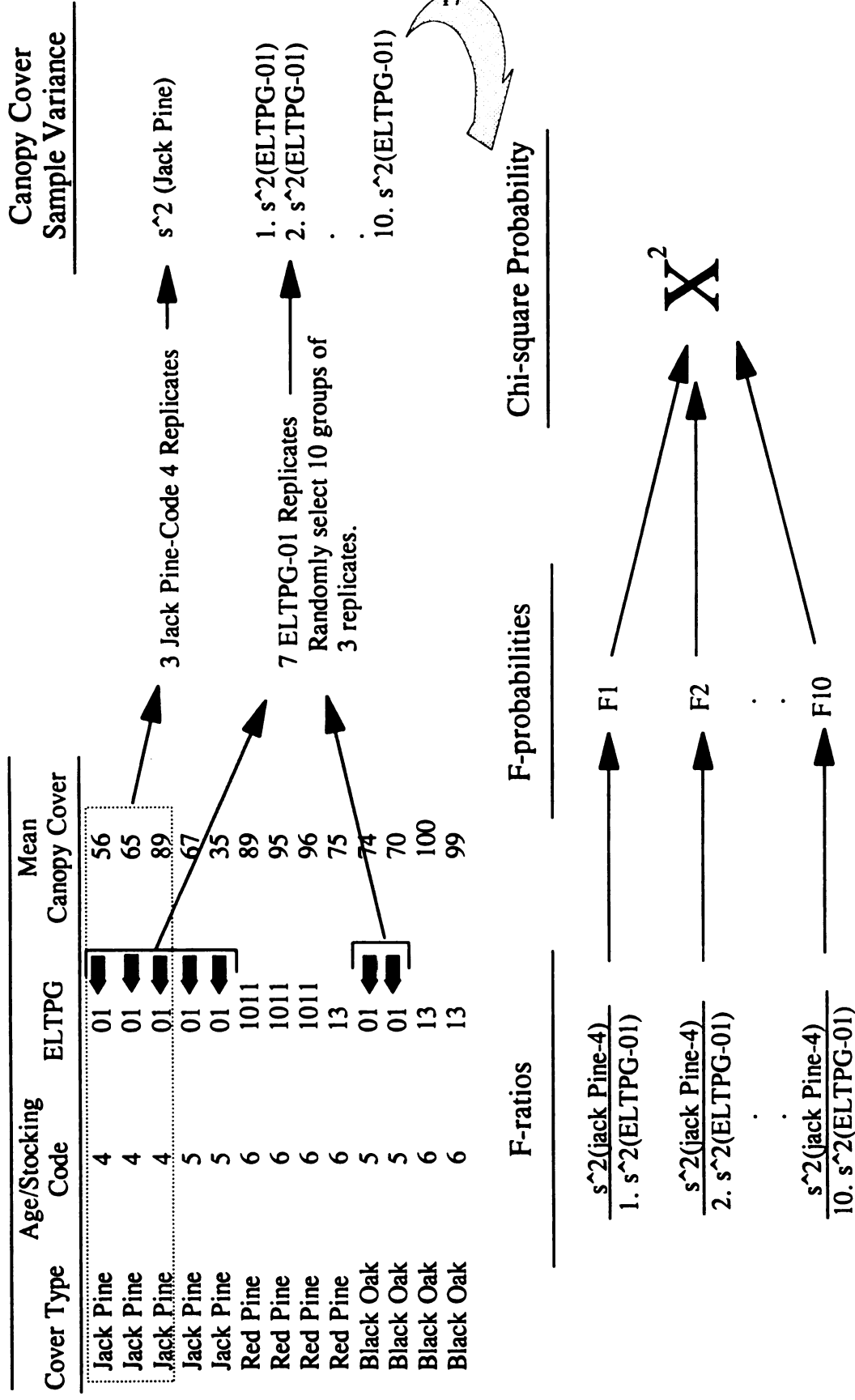


Figure 4. Relative efficiency tests between cover type and ecological land type phase grouping (ELTPG), where s^2 denotes sample variance.

Table 3. Number of replicates for cover type-ELTPG combinations, cover type, and ELTPG used for sample precision analyses.

Vegetation Category & Cover Type		Ecological Land Type Phase Grouping									
Cover Type	Code ¹	Cover Type Replicates	Outwash II	Outwash;Ice	Ice;Moraine I	Ice;Moraine II	Moraine I	Moraine II	Moraine III		
Overstory											
Jack Pine	4	6	3	3	.	.	.	.	.	.	.
Jack Pine	5	16	2	4	.	8	.	4	.	.	.
Jack Pine	6	17	3	3	.	11	.	.	.	.	.
Black Oak	5	9	.	.	.	6	.	3	.	.	.
N. Red Oak	6	13	.	.	3	.	5	5	.	.	.
Mixed Oak	6	10	.	.	.	5	.	5	.	.	.
N. Hardwoods	6	15	.	.	.	5	.	4	6	.	.
Trembling Aspen	3	14	.	3	.	6	.	5	.	.	.
Trembling Aspen	5	8	.	.	.	8	.	.	.	.	.
ELTPG Replicates⇒											
		6	13	3	3	49	5	26	6		
Understory											
Jack Pine	4	6	3	3	.	.	.	.	.	.	.
Jack Pine	5	16	.	4	.	8	.	4	.	.	.
Jack Pine	6	17	3	3	.	11	.	.	.	.	.
Black Oak	5	9	.	.	.	6	.	3	.	.	.
N. Red Oak	6	13	.	.	3	.	5	5	.	.	.
Mixed Oak	6	10	.	.	.	5	.	5	.	.	.
N. Hardwoods	6	15	.	.	.	5	.	4	6	.	.
Trembling Aspen	3	20	.	3	6	6	.	5	.	.	.
Trembling Aspen	5	8	.	.	.	8	.	.	.	.	.
ELTPG Replicates⇒											
		6	13	9	9	49	5	26	6		

Table 3. (Con't).

Vegetation Category & Cover Type		Ecological Land Type Phase Grouping							
Cover Type	Code ¹	Cover Type Replicates	Outwash II	Outwash;Ice	Ice;Moraine I	Ice;Moraine II	Moraine I	Moraine II	Moraine III
Dead & Downed									
Jack Pine	4	6	3	3	.	.	.	.	.
Jack Pine	5	16	.	4	.	8	.	4	.
Jack Pine	6	17	3	3	.	11	.	.	.
Black Oak	5	9	.	.	.	6	.	3	.
N. Red Oak	6	13	.	.	3	.	5	5	.
Mixed Oak	6	10	.	.	.	5	.	5	.
N. Hardwoods	6	9	.	.	.	5	.	4	6
Trembling Aspen	3	15	.	3	.	.	.	5	.
Trembling Aspen	5	8	.	.	.	8	.	.	.
ELTPG Replicates ²			6	13	3	43	5	26	6

¹Code=Corporate Database size/stocking codes where 3=sapling-sized, well-stocked; 4,5,6=pole-sized, poor-, medium-, and well-stocked, respectively (U.S. Forest Service 1988).

²No replicates sampled.

normality tests indicated most vegetation parameters required data transformations to satisfy the parametric assumptions of the relative efficiency tests (Table 2).

Relative Efficiency Tests - Cover Type and ELTPG versus Cover Type

Relative efficiency tests indicated that sampling precision of all vegetative variables could be potentially increased by incorporating ELTPG's (Tables 4-6). Typically, within each vegetation variable, greater than 50% of the sample variance comparisons suggested gains in sampling precision (i.e., the F-ratio was greater than 1.00; Table 7). For example, relative efficiency tests were conducted for basal area on 9 different cover types representing 7 ELTPG's (Table 4). Gains in sampling precision were implied by 13 (65%) of the relative efficiency tests, however, only 5 (25%) of the tests were significant ($P \leq 0.10$) (Table 7). For all vegetation variables, 43-74% (mean=59%) of the tests suggested a gain in sampling precision, with 5-39% (mean=24%) of the relative efficiency tests exhibiting significant gains (Table 7). At a lower probability ($\alpha=0.20$), the number of gains per vegetation variable typically increased, with several variables demonstrating significant gains for >40% of the tests (Table 7). Vegetation variables that exhibited the most significant gains in precision were understory cover ≥ 5 m tall and stem density, and snag diameter, whereas the fewest gains in precision were associated with overstory number of tree species, numbers of soft mast species, and snag density (Table 7).

Samples stratified by cover type-ELTPG were also evaluated for each cover type by vegetative variable grouping (i.e., overstory, understory, and dead and downed). Generally, gains in sampling precision were suggested for >50% of the relative efficiency tests, however, the number of significant gains was variable (Table 8). Pole-sized,

Table 4. Summary of relative efficiency tests between samples stratified by forest cover type-ecological land type phase groupings (ELTPG's) or forest cover type for overstory vegetation variables. Values are F-ratios (i.e. variance ratios).

Variable & Vegetation Type		Ecological Land Type Phase Grouping							
Cover Type	Code ¹	Outwash II	Outwash:Ice	Ice:Moraine I	Ice:Moraine II	Moraine I	Moraine II	Moraine III	
Basal Area									
Jack Pine	4	10.45*** ²	0.82	.	.	.	.	.	
Jack Pine	5	.	1.54	.	3.89***	.	1.03	.	
Jack Pine	6	1.75	6.15**	.	0.74	.	.	.	
Black Oak	5	.	.	.	8.81***	.	0.26	.	
N. Red Oak	6	.	.	1.27	.	1.12	3.12*	.	
Mixed Oak	6	.	.	.	1.01	.	1.15	.	
N. Hardwoods	6	.	.	.	0.73	.	2.32+	0.65	
Trembling Aspen	3	.	0.64	.	.	.	.	.	
Trembling Aspen	5	.	.	.	0.60	.	.	.	
Diameter									
Jack Pine	4	13.68***	3.35+	.	.	.	3.53*	.	
Jack Pine	5	.	14.75***	.	0.51	.	.	.	
Jack Pine	6	0.43	1.16	.	1.25	.	.	.	
Black Oak	5	.	.	.	1.54	.	0.52	.	
N. Red Oak	6	.	.	3.66*	.	0.80	0.50	.	
Mixed Oak	6	.	.	.	2.79*	.	0.49	.	
N. Hardwoods	6	.	.	.	1.99+	.	0.55	0.76	
Trembling Aspen	3	.	15.18***	.	.	.	.	.	
Trembling Aspen	5	.	.	.	1.19	.	.	.	
Height									
Jack Pine	4	3.05+	0.46	.	.	.	83.51***	.	
Jack Pine	5	.	3.01	.	1.58	.	.	.	
Jack Pine	6	1.12	35.75***	.	0.67	.	.	.	
Black Oak	5	.	.	.	3.85**	.	0.27	.	
N. Red Oak	6	.	.	10.92***	.	0.51	1.95+	.	
Mixed Oak	6	.	.	.	6.62***	.	0.48	.	
N. Hardwoods	6	.	.	.	0.48	.	2.48+	13.18***	
Trembling Aspen	3	.	1.85	.	.	.	.	.	
Trembling Aspen	5	.	.	.	0.92	.	.	.	

Table 4. (Con't).

Variable & Vegetation Type		Ecological Land Type Phase Grouping							
Cover Type	Code	Outwash II	Outwash, Ice	Ice, Moraine I	Ice, Moraine II	Moraine I	Moraine II	Moraine III	
Number Tree Species									
Jack Pine	4	0.48	1.02	.	.	.	.	.	.
Jack Pine	5	.	10.72***	.	1.17	.	0.73	.	.
Jack Pine	6	1.25	1.22	.	0.95	.	.	.	.
Black Oak	5	.	.	.	1.00	.	1.16	.	.
N. Red Oak	6	.	.	0.68	.	1.26	0.38	.	.
Mixed Oak	6	.	.	.	0.87	.	0.87	.	.
N. Hardwoods	6	.	.	.	0.81	.	0.98	0.74	.
Trembling Aspen	3	.	1.28	.	.	.	.	.	.
Trembling Aspen	5	.	.	.	1.28	.	.	.	.
Tree Density									
Jack Pine	4	0.39	1.91	.	.	.	.	.	.
Jack Pine	5	.	0.76	.	1.10	.	2.44+	.	.
Jack Pine	6	2.68+	6.29**	.	0.82	.	.	.	.
Black Oak	5	.	.	.	3.84**	.	0.45	.	.
N. Red Oak	6	.	.	18.05***	.	1.86+	2.14+	.	.
Mixed Oak	6	.	.	.	2.42+	.	0.49	.	.
N. Hardwoods	6	.	.	.	2.02+	.	0.37	4.84***	.
Trembling Aspen	3	.	1.97	.	2.96*	.	0.27	.	.
Trembling Aspen	5	.	.	.	0.95	.	.	.	.

¹Code=Corporate Database size/stocking codes where 3=sapling-sized, well-stocked; 4,5,6=pole-sized, poor-, medium-, and well-stocked, respectively (U.S. Forest Service 1988).

²Denotes level of significance for the test of H_0 : cover type precision=cover type-ELTPG precision; H_a : cover type precision < cover type-ELTPG precision where + ≤ 0.20 , * ≤ 0.10 , ** ≤ 0.05 , and *** ≤ 0.01 .

³Insufficient data.

Table 5. Summary of relative efficiency tests between samples stratified by forest cover type-ecological land type phase groupings (ELTPG's) or forest cover type for understory vegetation variables. Values are F-ratios (i.e. variance ratios).

Variable & Vegetation Type		Ecological Land Type Phase Grouping							
Cover Type	Code ¹	Outwash II	Outwash:Ice	Ice:Moraine I	Ice:Moraine II	Moraine I	Moraine II	Moraine III	
Cover $\geq$ 3m Tall									
Jack Pine	4	0.48	0.67	.	.	.	.	.	.
Jack Pine	5	?	1.37	.	.	.	.	.	.
Jack Pine	6	0.35	.	.	1.48	.	.	.	.
Black Oak	5	.	.	.	1.61+	.	.	.	.
N. Red Oak	6	.	.	7.52***	0.95	33.02***	41.97***	.	.
Mixed Oak	6	.	.	.	1.78	.	1.37	.	.
N. Hardwoods	6	.	.	.	.	.	1.11	.	.
Trembling Aspen	3	.	0.35	.	.	.	18.35***	122.17***	.
Trembling Aspen	5	.	.	.	0.74	.	1.31	.	.
Cover $\geq$ 5m Tall									
Jack Pine	4	0.31	0.61	.	.	.	.	.	.
Jack Pine	5	.	0.50	.	1.14	.	.	.	.
Jack Pine	6	0.31	.	.	1.17	.	.	.	.
Black Oak	5	.	.	.	0.81	.	546.1***	.	.
N. Red Oak	6	.	.	8.34**	.	219.3***	2.32+	.	.
Mixed Oak	6	.	.	.	2.74*	.	52.02***	.	.
N. Hardwoods	6	.	.	.	.	.	19.34***	50.87***	.
Trembling Aspen	3	.	1.95	.	.	.	0.58	.	.
Trembling Aspen	5	.	.	.	0.85	.	.	.	.
Deciduous Height									
Jack Pine	4	1.03	1.31	.	.	.	.	.	.
Jack Pine	5	.	0.66	.	1.33	.	.	.	.
Jack Pine	6	0.42	.	.	0.72	.	.	.	.
Black Oak	5	.	.	.	0.73	.	3.49*	.	.
N. Red Oak	6	.	.	3.32+	.	1.06	0.71	.	.
Mixed Oak	6	.	.	.	1.93+	.	0.55	.	.
N. Hardwoods	6	.	.	.	.	.	42.14***	14.75***	.
Trembling Aspen	3	.	0.44	.	.	.	1.75	.	.
Trembling Aspen	5	.	.	.	0.58	.	.	.	.

Table 5. (Con't).

Variable & Vegetation Type		Ecological Land Type Phase Grouping						
Cover Type	Code	Outwash II	Outwash;Ice	Ice;Moraine I	Ice;Moraine II	Moraine I	Moraine II	Moraine III
Size Density								
Jack Pine	4	1.45	0.65	.	.	.	.	.
Jack Pine	5	.	3.02*	.	0.91	.	1.61	.
Jack Pine	6	75.01***	5.59**	.	1.29	.	.	.
Black Oak	5	.	.	.	1.29	.	4.05*	.
N. Red Oak	6	.	.	3.25+	.	15.07***	0.38	.
Mixed Oak	6	.	.	.	8.07***	.	0.43	.
N. Hardwoods	6	.	.	.	2.19+	.	1.15	0.34
Trembling Aspen	3	.	17.52***	4.95***	1.64	.	1.49	.
Trembling Aspen	5	.	.	.	0.58	.	.	.
Soft Mast Cover								
Jack Pine	4	1.69	0.33	.	.	.	.	.
Jack Pine	5	.	1.04	.	0.78	.	1.24	.
Jack Pine	6	3.64*	12.50***	.	1.49	.	.	.
Black Oak	5	.	.	.	0.64	.	1.70	.
N. Red Oak	6	.	.	4.24*	.	0.82	0.72	.
Mixed Oak	6	.	.	.	5.10**	.	0.44	.
N. Hardwoods	6	.	.	.	0.77	.	0.83	0.37
Trembling Aspen	3	.	2.64+	0.88	1.09	.	0.70	.
Trembling Aspen	5	.	.	.	0.76	.	.	.
No. Soft Mast Species								
Jack Pine	4	0.85	1.00	.	.	.	.	.
Jack Pine	5	.	2.66+	.	1.75+	.	2.35+	.
Jack Pine	6	0.64	0.10	.	0.51	.	.	.
Black Oak	5	.	.	.	1.52	.	0.19	.
N. Red Oak	6	.	.	0.16	.	0.61	12.88***	.
Mixed Oak	6	.	.	.	0.15	.	2.78*	.
N. Hardwoods	6	.	.	.	0.87	.	11.16***	1.47
Trembling Aspen	3	.	0.47	1.68	1.50	.	2.45+	.
Trembling Aspen	5	.	.	.	2.02+	.	.	.

Table 5. (Con't).

¹Code=Corporate Database size/stocking codes where 3=sapling-sized, well-stocked; 4,5,6=pole-sized, poor-, medium-, and well-stocked, respectively (U.S. Forest Service 1988).

²Insufficient data.

³Denotes level of significance for the test of H_0 : cover type precision=cover type-ELTPG precision; H_a : cover type precision < cover type-ELTPG precision where $+ \leq 0.20$, $* \leq 0.10$, $** \leq 0.05$, and $*** \leq 0.01$.

Table 6. Summary of relative efficiency tests between samples stratified by forest cover type-ecological land type phase groupings (ELTPG's) or forest cover type for dead and downed vegetation variables. Values are F-ratios (i.e. variance ratios).

Variable & Vegetation Type		Ecological Land Type Phase Grouping							
Cover Type	Code ¹	Outwash II	Outwash;Ice	Ice;Moraine I	Ice;Moraine II	Moraine I	Moraine II	Moraine III	
S snag Density									
Jack Pine	4	1.25	2.72+	.	.	.	.	.	.
Jack Pine	5	2	13.19***	.	1.19	.	.	.	.
Jack Pine	6	0.45	.	.	1.08	.	.	.	.
Black Oak	5	.	.	.	0.74	.	0.49	.	.
N. Red Oak	6	.	.	2.21+	.	0.92	1.26	.	.
Mixed Oak	6	.	.	.	1.67	.	0.78	.	.
N. Hardwoods	6	.	.	.	.	.	0.92	31.00***	.
Trembling Aspen	3	.	.	.	.	.	.	.	.
Trembling Aspen	5	.	.	.	0.63	.	.	.	.
S snag Diameter									
Jack Pine	4	4.52* ³	6.82**	.	.	.	.	.	.
Jack Pine	5	.	162.5***	.	0.53	.	.	.	.
Jack Pine	6	0.90	.	.	1.16	.	.	.	.
Black Oak	5	.	.	.	0.63	.	4.15*	.	.
N. Red Oak	6	.	.	4.89**	.	0.54	0.43	.	.
Mixed Oak	6	.	.	.	0.85	.	1.56	.	.
N. Hardwoods	6	.	.	.	.	.	13.85***	0.42	.
Trembling Aspen	3	.	.	.	.	.	.	.	.
Trembling Aspen	5	.	.	.	1.04	.	.	.	.
Log Density									
Jack Pine	4	0.44	0.47	.	.	.	.	.	.
Jack Pine	5	.	1.12	.	1.25	.	.	.	.
Jack Pine	6	.	.	.	1.68	.	.	.	.
Black Oak	5	.	.	.	0.83	.	19.50***	.	.
N. Red Oak	6	.	.	0.70	.	1.61	0.56	.	.
Mixed Oak	6	.	.	.	4.49**	.	0.66	.	.
N. Hardwoods	6	.	.	.	.	.	8.60***	0.43	.
Trembling Aspen	3	.	0.37	.	.	.	819.61***	.	.
Trembling Aspen	5	.	.	.	1.21	.	.	.	.

Table 6. (Con't).

¹Code=Corporate Database size/stocking codes where 3=sapling-sized, well-stocked; 4,5,6=pole-sized, poor-, medium-, and well-stocked, respectively (U.S. Forest Service 1988).

²Insufficient data.

³Denotes level of significance for the test of H_0 : cover type precision=cover type-ELTPG precision; H_a : cover type precision < cover type-ELTPG precision where + ≤ 0.20 , * ≤ 0.10 , ** ≤ 0.05 , and *** ≤ 0.01 .

Table 7. Summary of precision gains for each vegetation variable stratified by forest cover type-ecological land type phase grouping (ELTPG) compared to samples stratified by forest cover type.

Vegetation Variable		Total Relative Efficiency Tests	No. Relative Efficiency Tests (% of Total)	
			Suggested Gains ¹	Significant Gains
				P ≤ 0.10
Overstory				
Basal Area	20	13 (65%)	5 (25%)	6 (30%)
Diameter	20	12 (60%)	6 (30%)	8 (40%)
Height	20	13 (65%)	6 (30%)	9 (45%)
Number Tree Species	20	9 (45%)	1 (5%)	1 (5%)
Tree Density	22	14 (64%)	5 (25%)	11 (50%)
Understory				
Cover ≥ 3m Tall	18	12 (67%)	5 (28%)	6 (33%)
Cover ≥ 5m Tall	18	11 (61%)	7 (39%)	8 (44%)
Deciduous Height	18	10 (56%)	3 (17%)	5 (28%)
Stem Density	23	17 (74%)	8 (35%)	10 (43%)
Soft Mast Cover	23	10 (43%)	4 (17%)	5 (22%)
No. Soft Mast Species	23	13 (57%)	3 (13%)	8 (35%)
Dead and Downed				
Snag Density	16	9 (56%)	2 (13%)	4 (25%)
Snag Diameter	16	9 (56%)	6 (38%)	6 (38%)
Log Density	17	9 (53%)	4 (24%)	4 (24%)

¹F-ratio greater than 1.00.

Table 8. Summary of precision gains for each forest cover type stratified by forest cover type-ecological land type phase grouping (ELTPG) compared to samples stratified by forest cover type.

Cover Type	Code ¹	Total Relative Efficiency Tests	No. Relative Efficiency Tests (% of Total)		
			Suggested Gains ²	Significant Gains	
				P ≤ 0.10	P ≤ 0.20

Overstory					
Jack Pine	4	10	6 (60%)	2 (20%)	4 (40%)
Jack Pine	5	15	12 (80%)	5 (33%)	6 (40%)
Jack Pine	6	15	10 (67%)	3 (20%)	4 (40%)
Black Oak	5	10	5 (50%)	3 (30%)	3 (30%)
N. Red Oak	6	15	10 (67%)	4 (27%)	7 (47%)
Mixed Oak	6	10	5 (50%)	2 (20%)	3 (30%)
N. Hardwoods	6	15	6 (40%)	2 (13%)	6 (40%)
Trembling Aspen	3	7	5 (71%)	2 (29%)	2 (29%)
Trembling Aspen	5	5	2 (40%)	0 (0%)	0 (0%)
Understory					
Jack Pine	4	12	4 (33%)	0 (0%)	0 (0%)
Jack Pine	5	15	11 (73%)	1 (7%)	4 (27%)
Jack Pine	6	15	8 (53%)	4 (27%)	5 (33%)
Black Oak	5	12	7 (58%)	4 (33%)	4 (33%)
N. Red Oak	6	18	12 (67%)	7 (39%)	10 (56%)
Mixed Oak	6	12	8 (67%)	5 (42%)	6 (50%)
N. Hardwoods	6	15	10 (67%)	7 (47%)	8 (53%)
Trembling Aspen	3	18	12 (67%)	2 (11%)	4 (22%)
Trembling Aspen	5	6	1 (17%)	0 (0%)	1 (17%)
Dead and Downed					
Jack Pine	4	6	4 (67%)	2 (33%)	3 (50%)
Jack Pine	5	6	5 (83%)	2 (33%)	2 (33%)
Jack Pine	6	5	3 (60%)	0 (0%)	0 (0%)
Black Oak	5	6	2 (33%)	2 (33%)	2 (33%)
N. Red Oak	6	9	4 (44%)	1 (11%)	2 (22%)
Mixed Oak	6	6	3 (50%)	1 (17%)	1 (17%)
N. Hardwoods	6	6	3 (50%)	3 (50%)	3 (50%)
Trembling Aspen	3	2	1 (50%)	1 (50%)	1 (50%)
Trembling Aspen	5	3	2 (67%)	0 (0%)	0 (0%)

¹Code=Corporate Database size/stocking codes where 3=sapling sized, well-stocked; 4,5,6=pole sized, poor-, medium-, and well-stocked, respectively (U.S. Forest Service 1988).

²F-ratio greater than 1.00.

medium-stocked black oak exhibited the highest significant sampling gains across all major vegetative groupings (30, 33, 33% for overstory, understory, and dead and downed respectively; Table 8). Other cover types that exhibited relatively high numbers of significant precision gains included pole-sized, well-stocked northern hardwoods (47%) and mixed oak (42%) for understory; pole-sized well-stocked northern hardwoods (50%) and sapling-sized, well-stocked trembling aspen (50%) for dead and downed variables (Table 8). Low numbers of precision gains were consistently demonstrated by pole-sized, medium-stocked trembling aspen (Table 8).

Generally, the cover type-ELTPG stratification exhibited higher precision for cover types not specifically defined by the Corporate Database classification. For example, "northern hardwoods" was an inclusive label typically referring to sugar maple-beech forests, however, these stands also included samples dominated by yellow birch (*Betula allenghensis*), paper birch (*B. papyrifera*), and oak. Similarly, the cover type denoted as "black oak" for this study included black oak, scarlet oak (*Q. coccinea*), and hickory (*Carya* spp.)(U.S. Forest Service 1988).

No ELTPG exhibited consistently higher or lower precision for samples stratified by cover type-ELTPG (Tables 4-6). Some ELTPG's were poorly represented in the sample (e.g. "Ice;Moraine I" and "Moraine III"), and these ELTPG groups typically exhibited the highest number of precision gains (Table 9).

Relative Efficiency Tests - Cover Type versus ELTPG

Relative efficiency tests comparing cover type and ELTPG stratification schemes indicated that for certain vegetation variables, an ELTPG stratification could provide more precise estimates of vegetative variables (Tables 10-12). Specifically, most vegetation variables sampled for pole-sized jack pine at all designated stocking densities exhibited

Table 9. Summary of precision gains for each ecological land type phase grouping (ELTPG) stratified by forest cover type-ELTPG compared to samples stratified by forest cover type.

Ecological Land Type Phase Grouping	Total Relative Efficiency Tests	No. Relative Efficiency Tests (% of Total)		
		Suggested Gains ²	Significant Gains	
			P ≤ 0.10	P ≤ 0.20
Overstory				
Outwash II	10	7 (70%)	2 (20%)	4 (40%)
Outwash;Ice	20	16 (80%)	6 (30%)	7 (35%)
Ice;Moraine I	5	4 (80%)	3 (60%)	3 (60%)
Ice;Moraine II	31	18 (58%)	7 (23%)	10 (32%)
Moraine I	5	3 (60%)	0 (0%)	1 (20%)
Moraine II	26	11 (42%)	3 (11%)	8 (31%)
Moraine III	5	2 (40%)	2 (40%)	2 (40%)
Understory				
Outwash II	12	5 (42%)	2 (17%)	2 (17%)
Outwash;Ice	21	10 (48%)	4 (19%)	6 (29%)
Ice;Moraine I	9	7 (78%)	4 (44%)	6 (67%)
Ice;Moraine II	36	20 (56%)	3 (8%)	8 (22%)
Moraine I	6	4 (67%)	3 (50%)	3 (50%)
Moraine II	33	23 (70%)	11 (33%)	14 (42%)
Moraine III	6	4 (67%)	3 (50%)	3 (50%)
Dead and Downed				
Outwash II	5	2 (40%)	1 (20%)	1 (20%)
Outwash;Ice	7	5 (71%)	3 (43%)	4 (57%)
Ice;Moraine I	3	2 (67%)	1 (33%)	2 (66%)
Ice;Moraine II	15	9 (60%)	1 (7%)	1 (7%)
Moraine I	3	1 (33%)	0 (0%)	0 (0%)
Moraine II	13	7 (54%)	5 (38%)	5 (38%)
Moraine III	3	1 (33%)	1 (33%)	1 (33%)

¹Code=Corporate Database size/stocking codes where 3=sapling-sized, well-stocked; 4,5,6=pole-sized, poor-, medium-, and well-stocked, respectively (U.S. Forest Service 1988).

²F-ratio greater than 1.00.

Table 10. Summary of relative efficiency tests between samples stratified by forest cover type or ecological land type phase grouping (ELTPG) for overstory vegetation variables. Values are chi square statistics derived from combining the results of 10 relative efficiency tests (see Fig. 4).

Variable & Vegetation Type		Ecological Land Type Phase Grouping									
Cover Type	Code ¹	Outwash I	Outwash II	Outwash, Ice	Ice; Moraine I	Ice; Moraine II	Moraine I	Moraine II	Moraine III		
Basal Area											
Jack Pine	5	31.76**	3.38	29.24*							
Jack Pine	6	100.51***		87.74***	75.63***	22.98					
Red Pine	6	51.52***	15.79		37.39***	18.71		17.67			
Black Oak	5					9.39		7.82			
Black Oak	6	12.35			7.61	3.30		2.99			
Red Oak	6			2.80	2.84	0.48		0.11		1.08	
Mixed Oak	6					0.93		0.72			
Mixed Upland Hardwoods	6				5.34	2.27		2.04		4.17	
Diameter											
Jack Pine	5	51.66***	8.32	22.18							
Jack Pine	6	36.17***		14.51	8.77	5.36					
Red Pine	6	5.29	0.27		1.06	0.31		0.62			
Black Oak	5					1.78		3.16			
Black Oak	6	6.97			1.45	0.43		0.89			
Red Oak	6			3.88	2.63	1.41		1.64		3.81	
Mixed Oak	6					22.20		27.48+			
Mixed Upland Hardwoods	6				7.34	2.86		4.85		7.57	
Height											
Jack Pine	5	29.02*	12.03	8.32							
Jack Pine	6	52.56***	40.17***	19.09	39.76***	9.31					
Red Pine	6	11.72	4.90		10.59	0.92		0.99			
Black Oak	5					7.39		8.53			
Black Oak	6	47.21***			39.06***	7.59		9.05			
Red Oak	6			3.32	14.50	1.99		2.32		35.61**	
Mixed Oak	6					4.59		5.44			
Mixed Upland Hardwoods	6				22.00	3.14		3.74		29.67*	

Table 10. (Con't).

Variable & Vegetation Type		Ecological Unit							
Cover Type	Code ¹	Outwash I	Outwash II	Outwash; Ice	Ice; Moraine I	Ice; Moraine II	Moraine I	Moraine II	Moraine III
Number Tree Species									
Jack Pine	5	16.56	30.52*	15.84	.	.	.	.	.
Jack Pine	6	6.87	.	5.34	0.98	2.32	.	.	.
Red Pine	6	10.12	20.05	.	2.36	3.54	.	4.13	.
Black Oak	5	.	.	.	.	19.02	.	21.65	.
Black Oak	6	42.76***	.	.	15.64	20.16	.	21.61	.
Red Oak	6	.	.	27.17+	9.72	13.00	.	10.21	10.34
Mixed Oak	6	.	.	.	.	0.78	.	1.20	.
Mixed Upland Hardwoods	6	.	.	.	1.18	1.85	.	2.22	2.10
Tree Density									
Jack Pine	5	1.55	.	5.97	.	.	.	.	.
Jack Pine	6	46.39***	.	78.16***	41.57***	38.79***	.	.	.
Red Pine	6	77.72***	108.66***	.	112.31***	72.20***	104.91***	.	33
Black Oak	5	.	.	.	.	5.21	.	10.74	.
Black Oak	6	1.22	.	.	3.16	1.32	.	3.03	.
Red Oak	6	.	.	7.77	2.64	1.74	.	5.23	1.78
Mixed Oak	6	.	.	.	.	8.29	.	16.05	.
Mixed Upland Hardwoods	6	.	.	.	0.07	0.04	.	0.07	0.02

¹Code=Corporate Database size/stocking codes where 5,6=pole-sized, medium-, and well-stocked, respectively (U.S. Forest Service 1988).

²Denotes level of significance for the test H_0 : cover type precision=ELTPG precision; H_1 : cover type precision < ELTPG precision where + ≤ 0.20 , * ≤ 0.10 , ** ≤ 0.05 , and *** ≤ 0.01 . **Bold-face** indicates that the Chi-square statistic suggested (i.e. F-ratios were greater than 1) a higher sample precision for ELTPG stratification.

³Insufficient data.

Table 11. Summary of relative efficiency tests between samples stratified by forest cover type or ecological land type phase grouping (ELTPG) for understory vegetation variables. Values are chi square statistics derived from combining the results of 10 relative efficiency tests (see Fig. 4).

Variable & Vegetation Type		Ecological Land Type Phase Grouping									
Cover Type	Code ¹	Outwash I	Outwash II	Outwash:Ice	Ice:Moraine I	Ice:Moraine II	Moraine I	Moraine II	Moraine III		
Cover ≥ 3m Tall											
Jack Pine	4	27.49+ ²	10.59	30.87*	.	.	.	.	.	.	.
Jack Pine	5	12.10	3.86	15.04	.	6.42	.	2.45	.	.	.
Jack Pine	6	27.50+	10.59	30.87*	15.58	16.67	.	.	.	.	.
Red Pine	6	2.03	0.48	.	0.84	0.82	.	0.28	.	.	.
Black Oak	6	3.39	.	.	1.13	1.94	.	0.46	.	.	.
Red Oak	6	.	.	32.74**	12.40	19.27	52.72***	6.59	4.78	.	.
Mixed Oak	6	.	.	.	.	9.12	.	3.63	.	.	.
Cover ≥ 5m Tall											
Jack Pine	5	19.82	.	5.63	.	10.14	.	5.07	.	34	.
Jack Pine	6	39.90***	.	13.64	41.43***	23.26	.	.	.	.	.
Red Pine	6	7.83	.	.	11.00	3.43	.	1.54	.	.	.
Black Oak	6	5.99	.	.	2.21	2.48	.	1.00	.	.	.
Red Oak	6	.	.	13.51	18.74	20.43	.	11.54	11.73	.	.
Mixed Oak	6	.	.	.	.	12.20	.	6.21	.	.	.
Deciduous Height											
Jack Pine	4	8.72	4.37	3.88	.	.	.	.	.	.	.
Jack Pine	5	28.83*	16.97	14.63	.	23.06	.	45.28***	.	.	.
Jack Pine	6	16.28	8.83	7.68	17.60	13.72	.	.	.	.	.
Red Pine	6	1.57	0.67	.	2.64	1.69	.	6.32	.	.	.
Black Oak	6	64.41***	.	.	66.85***	48.99***	.	71.89***	.	.	.
Red Oak	6	.	.	7.58	30.69*	15.08	8.45	27.00+	8.26	.	.
Mixed Oak	6	.	.	.	.	13.49	.	37.47***	.	.	.

Table 11. (Con't).

Variable & Vegetation Type		Ecological Land Type Phase Grouping									
Cover Type	Code ¹	Outwash I	Outwash II	Outwash, Ice	Ice; Moraine I	Ice; Moraine II	Moraine I	Moraine II	Moraine III		
Stem Density											
Jack Pine	4	1.40	1.15	41.23***	.	.	.	.	.	.	.
Jack Pine	5	10.07	11.90	262.49***	.	4.13	.	6.89	.	.	.
Jack Pine	6	13.80	14.16	100.00***	1.29	6.34	.	.	.	.	.
Red Pine	6	0.69	0.54	.	0.08	1.39	.	0.64	.	.	.
Black Oak	5	.	.	.	.	6.51	.	4.27	.	.	.
Black Oak	6	6.97	.	.	1.07	4.35	.	7.80	.	.	.
Red Oak	6	.	.	138.15***	3.21	11.98	2.18	17.59	1.46	.	.
Mixed Oak	6	.	.	.	.	38.23***	.	37.14***	.	.	.
Mixed Upland Hardwoods	6	.	.	.	0.59	2.62	.	4.94	0.58	.	.
Trembling Aspen	3	.	.	.	1.33	6.49	.	8.06	.	.	.
Big-toothed Aspen	3	.	.	.	0.40	.	.	2.74	0.39	.	.
Openings	0	.	.	.	14.17	33.41**	9.29	46.99***	.	.	.
Soft Mass Cover											
Jack Pine	4	18.45	7.15	15.65	.	.	.	.	.	.	.
Jack Pine	5	5.59	2.42	8.52	.	9.30	.	2.47	.	.	.
Jack Pine	6	34.82**	19.23	36.92***	22.43	44.69***	.	.	.	.	.
Red Pine	6	27.64+	8.77	.	12.43	21.14	.	27.92+	.	.	.
Black Oak	5	.	.	.	.	51.76***	.	56.07***	.	.	.
Black Oak	6	50.71***	.	.	26.78+	43.24***	.	43.96***	.	.	.
Red Oak	6	.	.	23.28	15.17	24.75	1.31	7.95	11.67	.	.
Mixed Oak	6	.	.	.	.	7.95	.	33.99**	.	.	.
Mixed Upland Hardwoods	6	.	.	.	4.81	8.57	.	14.35	3.53	.	.
Trembling Aspen	3	.	.	.	9.50	20.91	.	7.20	.	.	.
Big-toothed Aspen	3	.	.	.	4.31	.	.	14.99	3.08	.	.
Openings	0	.	.	.	36.88***	58.00***	11.84	54.52***	.	.	.

Table 11. (Con't).

Variable & Vegetation Type		Ecological Land Type Phase Grouping							
Cover Type	Code ¹	Outwash I	Outwash II	Outwash, Ice	Ice, Moraine I	Ice, Moraine II	Moraine I	Moraine II	Moraine III
No. Soft Mast Species									
Jack Pine	4	52.30***	20.26	33.30**	.	.	.	.	.
Jack Pine	5	55.59***	33.04**	39.46***	.	.	.	13.42	.
Jack Pine	6	26.43+	11.47	11.40	14.25	11.39	.	.	.
Red Pine	6	16.64	6.35	.	7.63	8.24	.	3.57	.
Black Oak	5	.	.	.	.	3.67	.	3.75	.
Black Oak	6	80.38***	.	.	52.74***	51.22***	.	33.24**	.
Red Oak	6	.	.	24.70	19.34	18.72	5.45	6.86	3.18
Mixed Oak	6	.	.	.	.	29.57*	.	28.70*	.
Mixed Upland Hardwoods	6	.	.	.	8.23	8.41	.	3.90	2.72
Trembling Aspen	3	.	.	.	8.93	7.41	.	1.24	.
Big-toothed Aspen	3	.	.	.	65.16***	.	.	39.12***	21.58
Openings	0	.	.	.	1.49	1.65	0.40	0.60	.

¹Code = Corporate Database size/stocking codes where 0 = openings (no stocking); 3 = sapling-sized, well-stocked; 4, 5, 6 = pole-sized, poor-, medium-, and well-stocked, respectively (U.S. Forest Service 1988).

²Denotes level of significance for the test H_0 : cover type precision = ELTPG precision; H_a : cover type precision $\leq$ ELTPG precision where + ≤ 0.20 , * ≤ 0.10 , ** ≤ 0.05 , and *** ≤ 0.01 . **Bold-face** indicates that the Chi-square statistic suggested (i.e. F-ratios were greater than 1) a higher sample precision for ELTPG stratification.

³Insufficient data.

Table 12. Summary of relative efficiency tests between samples stratified by forest cover type or ecological land type phase grouping (FLTPG) for dead and downed trees.

Table 12. Summary of relative efficiency tests between samples stratified by forest cover type or ecological land type phase grouping (ELTPG) for dead and downed vegetation variables. Values are chi square statistics derived from combining the results of 10 relative efficiency tests (see Fig. 4).

Variable & Vegetation Type		Ecological Land Type Phase Grouping									
Cover Type	Code ¹	Outwash I	Outwash II	Outwash;Ice	Ice;Moraine I	Ice;Moraine II	Moraine I	Moraine II	Moraine III		
S snag Density											
Jack Pine	4	24.33	9.22	52.52***	.	.	.	.	.	.	.
Jack Pine	5	40.04*** ²	17.01	75.58***	.	44.06***	.	68.50***	.	.	.
Jack Pine	6	11.07	3.61	27.98+	32.42**	13.55	.	.	.	.	.
Red Pine	6	16.64	5.80	.	42.98***	19.58	.	34.66**	.	.	.
Black Oak	6	7.43	.	.	13.41	7.27	.	11.07	.	.	.
Red Oak	6	.	.	40.92***	29.81*	17.90	.	25.54+	.	.	.
Mixed Oak	6	.	.	.	.	11.50	.	21.95	.	.	.
S snag Diameter											
Jack Pine	4	54.41***	12.54	62.82***	.	.	.	.	.	.	37
Jack Pine	5	4.79	0.54	6.35	.	1.49	.	3.99	.	.	.
Jack Pine	6	38.34***	8.23	47.53***	51.07***	17.68	.	.	.	.	.
Red Pine	6	43.90***	9.99	.	57.38***	20.93	.	40.36***	.	.	.
Black Oak	6	45.62***	.	.	59.32***	17.90	.	36.70***	.	.	.
Red Oak	6	.	.	6.80	10.87	1.28	.	4.05	.	1.64	.
Mixed Oak	6	.	.	.	.	3.85	.	9.59	.	.	.
Log Density											
Jack Pine	4	21.50	8.73	26.41+	.	.	.	.	.	.	.
Jack Pine	5	27.23+	11.60	32.66**	.	57.33***	.	19.02	.	.	.
Red Pine	6	21.12	8.54	.	18.24	17.84	.	47.84***	.	.	.
Black Oak	6	0.06	.	.	0.06	0.40	.	0.10	.	.	.
Red Oak	6	.	.	24.30	24.77	66.82***	25.58+	27.27+	.	17.85	.
Mixed Oak	6	.	.	.	.	31.96**	.	8.53	.	.	.

Table 12. (Con't).

¹Code=Corporate Database size/stocking codes where 4,5,6=pole-size, poor-, medium-, and well-stocked, respectively (U.S. Forest Service 1988).

²Denotes level of significance for the test H_0 : cover type precision = ELTPG precision; H_a : cover type precision $\leq$ ELTPG precision where $+ \leq 0.20$, $* \leq 0.10$, $** \leq 0.05$, and $*** \leq 0.01$. **Bold-face** indicates that the Chi-square statistic suggested (i.e. F-ratios were greater than 1) a higher sample precision for ELTPG stratification.

³Insufficient data.

higher precision when stratified by ELTPG's (Tables 10-12). Many of the differences in sampling precision for jack pine were significant ($P \leq 0.10$). Also, dead and downed vegetation variables for pole-sized, well-stocked red pine and pole-sized, medium-stocked black oak were more precisely sampled by ELTPG (Table 12). Some overstory and understory vegetation variables for these cover types also exhibited higher precision when sampled by ELTPG's (Tables 10-11). Vegetation variables for most other cover types did not exhibit gains in precision when sampled by ELTPG (Tables 10-12).

Most of the cover types found on ELTPG's with greater moisture availability (e.g., red oak, mixed oaks, and mixed upland hardwoods on "Ice; Moraines" and Moraines"; Fig. 2) exhibited fewer gains in precision relative to cover types found on the "Outwash" ELTPG's (Tables 10-12). Sampling conducted on "Outwash I" and "Outwash II" proved to be the most efficient for overstory variables relative to other ELTPG's (Table 10). Conversely, gains in sampling efficiency for understory and dead and downed vegetation variables exhibited no consistent trends across ELTPG's (Tables 11-12).

DISCUSSION

The objective of stratification (in a statistical sense) is to minimize within block variation such that between block variation is detectable. When choosing a vegetation sampling protocol from more than one stratification scheme, one tries to select the stratification that provides accurate estimates of sample parameters in the most efficient manner. A primary concern regarding the compilation of vegetative databases for resource planning is the sampling effort required. Resource agencies do not have the personnel or finances to collect data inefficiently, and with the need for ecosystem-based data inventories increasing (Davis 1980), the need for efficient data collection is necessary. Vegetation databases for the U.S. Forest Service have historically been stratified

by dominant commercial cover types (U.S. Forest Service 1988). Further stratification, for example, by cover type and ELTPG, should provide more precise estimates of vegetative variables, consistent with stratification theory. Often, the primary question is whether the stratification scheme(s) are legitimate. An inappropriately designed or applied stratification scheme may in fact increase within block variability.

The cover type-ELTPG stratification scheme generally followed stratification theory. The probability of increasing sampling precision when obtaining a vegetation sample stratified by cover type-ELTPG was almost always $>50\%$ (as suggested by relative efficiency tests), however, these gains were only significant ($P \leq 0.10$) for about 25% of the comparisons. Insignificance may have been due to low sample sizes, and indeed, the required sample sizes (as indicated by Frees's (1978) sample size formula) (Table 13) for most of the vegetation variables were above the number collected.

The relatively consistent precision gains for cover type-ELTPG stratification regardless of vegetative structure or ELTPG was encouraging. Increasingly complex stratification is best suited (both in terms of sampling practicality and logistics) for operational level forest planning, however, planning must also occur at the strategic and tactical levels (Roloff and Haufler 1993). At these higher broader planning levels, systems demonstrating consistency across landscapes are extremely valuable. The cover type-ELTPG stratification appeared to provide reasonable gains in sampling precision throughout the study areas. However, caution should be exercised when interpreting these results because 1) the assumption is made that a representative sample of "ecological units" were obtained (Driscoll et al. 1984), 2) there may be large-scale variability in cover type-ELTPG units (Bailey et al. 1978) not detected by the stratification scheme, and 3) the personnel and financial constraints on this type of data collection are of concern.

Table 13. Means and standard errors (parenthesis) for number of replicates required ($t_{0.10}$; accuracy 0.20 of the mean) according to Freese (1978) for vegetation variables sampled by Forest Service cover type and ELTPG.

Vegetation Variable	Forest Service Cover Type- ELTPG Stratification	
	N ¹	Replicates
Overstory		
Basal Area	19	17(4)
Diameter	19	3(1)
Height	19	3(1)
Number Tree Species	19	26(6)
Tree Density	19	26(8)
Understory		
Cover $\geq$ 3m Tall	18	35(10)
Cover $\geq$ 5m Tall	18	45(13)
Deciduous Height	19	13 (3)
Stem Density	23	35 (8)
Soft Mast Cover	23	40 (7)
No. Soft Mast Species	23	15 (6)
Dead and Downed		
Snag Density	16	48(11)
Snag Diameter	16	3 (1)
Log Density	17	81(18)

¹N=number of replicates sampled to compute the minimum number of replicates required.

For rapid, landscape assessments of some vegetative variables, the ELTPG stratification may provide a "quick and dirty" alternative. For most overstory variables, ELTPG was a more efficient stratifier than the jack pine cover type classification, particularly on the "Outwash" ELTPG's. Jack and red pine dominate these ELTPG types across the study areas, and thus, samples from "Outwash" ELTPG's were relatively homogenous with respect to cover type. In contrast, jack pine and red pine were sampled on numerous ELTPG's, including morainal systems. Morainal ELTPG's had greater structural diversity potentials than "Outwash" ELTPG's, and thus, this diversity is included in the cover type samples.

Understory variables exhibited no consistent trends in precision gain when sampled only by ELTPG. Some ELTPG's yielded more precise estimates of dead and downed vegetative variables, particularly "Outwash:Ice" and "Ice:Moraine I" for snag density and snag diameter (Table 8).

All stratification schemes used in this study were complex. The cover type-ELTPG stratification had 6 components, whereas the cover type and ELTPG schemes each had 3 major components. The cover type scheme was based on dominant, commercial tree species; stocking density, and age. The ELTPG's were delineated based on landform, soils, and potential vegetation (climate was viewed as equally applicable to both stratification schemes). The cover type-ELTPG scheme was a combination of cover type and ELTPG. The major difference (besides cover type being commercially driven as opposed to the ecosystem-based ELTPG's) was the spatial extent of each scheme. Cover type-ELTPG sample units were delineated at small spatial scales, cover types were delineated at intermediate scales, and ELTPG's were delineated at large scales. For example, 148 cover type-ELTPG units, 70 different cover types, and 14 ELTPG's were

identified on the study sites. To measure overstory basal area with 90% confidence and 20% allowable error according to Freese's (1978) sample size formula, 8 replicates from each cover type-ELTPG, 15 replicates from each cover type, and 15 replicates per ELTPG would need to be sampled. Total number of replicates required for each stratification scheme would be 1,184 for cover type-ELTPG, 1,050 for cover type, and 210 for ELTPG. At first glance, one would recommend sampling basal area by ELTPG, however, one must also consider the stratification scheme that provides the "best" estimate of the variable. If less variation around the mean equates to a "better" estimate, then the ELTPG stratification is inferior to the cover type-ELTPG strata (as suggested by the relative efficiency tests). The issue now becomes management oriented. If gross, landscape descriptions of basal area are acceptable, then ELTPG should be considered as a stratification scheme. In contrast, for situations requiring precise, defensible estimates (e.g., legal mitigation), the cover type-ELTPG stratification is recommended.

Relative efficiency tests were used to evaluate trends in sample precision as they related to cover type, ELTPG, and vegetative variables. Spatial scale appeared to be an important consideration in delineating these relationships. For example, at smaller scales (cover type-ELTPG units), sampling precision among ELTPG's was generally consistent, with all ELTPG's demonstrating some reductions in sample variance. At larger spatial scales, (e.g. ELTPG's), vegetation variables for cover types naturally associated with extremes on the ELTPG scale (e.g., jack pine naturally occurs on Outwash 1; Fig. 2) were more efficiently sampled when stratified by ELTPG as opposed to cover type. Conversely, vegetation variables associated with cover types that grow on more fertile ELTPG's (e.g., morainal systems) exhibited higher sampling precision when stratified by cover type.

These observations may be related to the inherent variability of the ECS units.

At small spatial scales, the sample is stratified such that within block variation is minimized. At larger scales, within-stand heterogeneity causes increases in sampling error. More fertile ELTPG's have a greater potential for complexity (Spies and Barnes 1985). Whereas "Outwash" systems could only support species adapted to well-drained soils, "Moraines" could support most of the "Outwash" species in addition to species requiring more moisture. The morainal systems sampled for this study exhibited more diverse vegetative communities than outwash systems (Fig. 2).

The relatively poor ability of "Morainal" ELTPG's to increase sampling precision at larger spatial scales may also be attributed to performing the vegetative analysis at too coarse of scale on the ECS hierarchy. Since morainal systems are more vegetatively heterogeneous, grouping similar ecological land type phases (Cleland et al. 1993) may have obscured the discriminatory capabilities of the system. However, when one examines ELTPG's based on only a single ecological land type phase (e.g., "Outwash I", "Ice; Moraine I", "Moraine II"), no consistent gains in sampling precision were evident.

Integrating an ecological classification system into a vegetative sampling strategy is appealing in that the delineated boundaries have greater ecological significance (as opposed to cover type maps). An ecological classification system permits managers to more accurately predict successional pathways and the effects of their management activities on vegetation structure. Also, a knowledge of the ecological relationships provides insight to natural disturbance regimes at a large spatial scale.

An explanation for the variability in sampling precision from the cover type-ELTPG stratification scheme may be that silviculturally induced disruptions of the natural ecological processes have created a vegetative structure not conducive to ecological quantification. The ages of vegetative cover types varied, but most stands on the study

sites recently (< 70 years) experienced some sort of silvicultural prescription or disturbance. The variable results from the relative efficiency analyses may be representative of: 1) some past silvicultural prescriptions being more ecologically based than others, 2) the plasticity of terrestrial plants; relying on several individual and population-level characteristics that buffer them from environmental unpredictability (Bazzaz and Sipe 1987), or 3) different cover type-ELTPG units exhibiting temporal variation in returning to a pre-disturbance state (O'Neill et al. 1986, Swanson et al. 1988). This would partially explain gains in sampling precision for some understory vegetative variables (e.g., the soft mast attributes), especially at larger spatial scales (i.e. the ELTPG scheme). In theory, understory and ground components should exhibit greater resiliency to disturbance than overstory species because of shorter turn-around times (Cleland et al. 1993), however, inconsistent gains in sampling precision for most understory variables may suggest that micro-site variation in physiography and soils not detected by the ECS further influences forest composition and understory recruitment (Host et al. 1987, McNab 1991). Also, certain understory units may be associated with specific overstory units (McNab 1991) rather than ELTPG's.

Vegetative attributes exhibited varying degrees of sample precision dependent on stratification scheme. Jones (1991) found that vegetation patterns within pine types in Georgia were not a function of environmental conditions (i.e., an ECS) but rather conditions of stand establishment or subsequent anthropogenic influences. These factors are also important in northern Michigan and undoubtedly explain some of the vegetative sampling variability exhibited in this study.

SUMMARY AND CONCLUSIONS

The relationships between the ECS and vegetation (and associated processes) is almost always through spatial patterns and structures (Bailey et al. 1978). Although processes are controlled by structure, processes emerge only at the integration of composition, structures, and associated interactions (Bailey et al. 1978). Each level of the ECS is defined by an integration of the dominant ecological factors affecting biological systems at that scale, thus, spatial concerns and large-scale interactions are incorporated into the ECS (Cleland et al. 1993). Also, the ECS fulfills legal and policy requirements for coordinating and integrating resource inventories on National Forest lands (Cleland et al. 1993). The utility of incorporating ECS into vegetative sampling is warranted, however, as demonstrated, there is seldom an individual classification scheme that addresses all of the needs and concerns of resource managers. For overstory vegetative descriptions, the Forest Service cover type is an effective tool, however, it is important to remember that the Forest Service classification provides measures of existing overstory conditions, whereas the ECS identifies potential natural vegetation (Russell and Jordan 1991, Cleland et al. 1993). Also, contrary to some cover types, ECS's are expected to respond to management alternatives similarly (Driscoll et al. 1984).

Chapter 2: Wildlife Habitat Model Validation

Habitat has been theoretically defined as the location supporting wildlife including space, food, cover, and other animals (Giles 1978); often characterized by vegetation, landform, and hydrology (Odum 1971). Resource management influences the physical features of habitat, the effects of which are often paralleled by differences in animal abundance (Farmer et al. 1982). Resource managers are responsible for wildlife habitat and must justify and account for their management activities, however, they seldom know what kinds, amounts, and distributions of habitats are necessary to maintain population viability for all species (as required by law; Hurley et al. 1982). With top-level management, legislators, and fiscal officers wanting plans, budgets, and project proposals in quantitative terms (as opposed to documents created by intuition or expertise) (Davis 1980, Jarvinen 1985, Maurer 1986, Flather et al. 1989), the need for rapid and precise estimates of habitat quality has escalated. In response, resource management agencies have turned to modeling wildlife-habitat relationships.

Numerous studies have identified specific vegetation characteristics important for determining wildlife abundances and distributions. Avian studies dominate the literature, with most showing generalized relationships such as avian species richness, avian community structure, or bird species abundance to habitat heterogeneity, vegetation abundance, foliage height diversity, and/or habitat complexity (MacArthur and MacArthur 1961, Brewer 1963, Karr 1968, Karr and Roth 1971, Willson 1974, Pearson 1975,

Franzreb and Ohmart 1978, Lines and Perry 1978, Moss 1978, Asherin et al. 1979, Noon et al. 1980, Ambuel and Temple 1982, Dobkin and Wilcox 1986, Freemark and Merriam 1986, Freemark 1988, Johnson and Brown 1990, O'Brien 1990). Others have documented relationships between bird species and specific vegetation structures (Jackson 1970, James 1971, Holmes and Robbins 1981, James and Wamer 1982, Balda et al. 1983, Grue et al. 1983, Morrison 1983, Bock and Webb 1984, Swift et al. 1984, Rotenberry 1985, Maurer 1986, Askins and Philbrick 1987, Askins et al. 1987). Research on other wildlife species is more limited. Studies have correlated (with varying degrees of specificity) vegetative structures and small mammal abundances (Rosenzweig and Winakur 1969, Brown 1973, Miller and Getz 1977, August 1983). Pianka (1967) and Marcot et al. (1983) demonstrated similar relationships for lizards.

Since the beginning of the 20th century, theories regarding the relationships between wildlife populations and their habitats have abounded (see review in Morrison et al. 1992; Chapter 1), however, the first effective efforts to organize and integrate available information into resource planning did not occur until the middle 1970's (Thomas 1986). Based on numerous studies and theories supporting the concept of wildlife-habitat relationships, quantitative models that describe these phenomenon were developed. An appealing characteristic of habitat models is the relative ease of inventorying habitats as opposed to wildlife populations. Also, habitat models establish a link between wildlife populations and the land altered by management practices (Flather et al. 1989). Habitat models are also more applicable (as opposed to population-based models) to regional and national analyses (Cooperrider 1986).

Numerous habitat models exist (see Verner et al. 1986, review in Morrison et al. 1992; Chapter 6), all with several inherent assumptions; that 1) animal distribution and

abundance can be explained and predicted by environmental attributes (Marcot et al. 1983), 2) similar patterns or configurations of habitat will reflect similar patterns of animal abundance (Flather and Hoekstra 1985), and 3) animal abundance is positively associated with habitat quality (Van Horne 1983, Best and Stauffer 1986). These assumptions make the task of model validation difficult, and indeed, validation and robustness studies have seldom been conducted (Lancia et al. 1982).

In wildlife-habitat relationships modeling, the assumption that species density is a direct measure of habitat quality has seldom been questioned (Van Horne 1983). Van Horne (1983) summarized the potential problems with this assumption. The most accurate estimates of wildlife density and habitat quality relationships are dependent on an understanding of population demographics and the factors influencing survival and reproduction (Maguire 1973, Van Horne 1983). However, until databases link population dynamics to specific habitat attributes and configurations across landscapes, the most feasible means of estimating wildlife potentials across a landscape is through habitat measurements.

The Habitat Evaluation Procedures were developed by the U.S. Fish and Wildlife Service (1981) to provide a habitat-based approach to assessing environmental impacts of resource development projects (Schamberger and Krohn 1982, Cole and Smith 1983). Habitat suitability indices (HSI's) are integral components of the Habitat Evaluation Procedures and are best viewed as hypotheses of species-habitat relationships (Schamberger and Krohn 1982). The value of HSI's lies in documenting a repeatable assessment procedure and providing an index to particular environmental characteristics that can be compared with alternative management plans (Schamberger and Krohn 1982, Morrison et al. 1992). HSI's are based on the premise that habitat suitability can be linked

to habitat attributes with linear relationships (Morrison et al. 1992). Subsequently, these attributes are mathematically combined into an index to habitat quality (Blake and Karr 1984, Laymon and Barrett 1986).

The development and implementation of HSI models has been flawed due to the lack of field validation (Cole and Smith 1983). Critics of HSI models argue that management decisions should not be based on untested models (Lancia et al. 1982, Cole and Smith 1983, Marcot et al. 1983, Laymon and Barrett 1986). Validation is the act of increasing, to an acceptable level, the confidence that an inference about a simulated process is correct for the actual process (Van Horn 1971). Using a variety of validation techniques, some researchers have found positive correlations between model predictions and measures of wildlife distributions and abundance (Lancia et al. 1982, Cole and Smith 1983, Cook and Irwin 1985, Dedon et al. 1986, Hammill and Moran 1986, Latka and Yanhke 1986, Laymon and Barrett 1986, Laymon and Reid 1986, Raphael and Marcot 1986, Stauffer and Best 1986), whereas others have found negative or no correlations (Seitz et al. 1982, Clark and Lewis 1983, Bart et al. 1984, Johnson and Temple 1986, Lancia et al. 1986, Larson and Bock 1986, Seng 1991, Robel et al. 1993). A variety of explanations for these discrepancies have been offered including inadequate or unrepresentative sampling (Cole and Smith 1983), model equations not representative of the actual wildlife-habitat relationships (Farmer et al. 1982, Cole and Smith 1983, Cooperrider and Behrend 1983, Warwick and Cade 1988, Van Horne and Wiens 1991), misinterpretation of results (Capen et al. 1986, Brower and Zar 1984), or application of the model to inappropriate spatial scales (Wiens 1986).

Successful habitat models should be valid, general enough that a single model can apply to a wide range of situations without major modifications, and usable by land

managers (Van Horne and Wiens 1991). In terms of U.S. Forest Service assessment goals, a synthesis of ecological theory into a modeling framework is necessary to fulfill legislative mandates. The goal of this chapter was to validate several HSI models for Forest Service management indicator species and to evaluate their usefulness in land use planning. Specific objectives are:

1. to determine if habitat quality as predicted from HSI models is correlated with measures of animal abundance, and
2. to evaluate the appropriate spatial scale of model applicability.

METHODS

Twelve wildlife management indicator species were selected to represent a broad range of habitat requirements. Most of these species were included in the Land Resources Management Plan for the Huron-Manistee National Forest and have been designated as species indicative of the health, status, and quality of habitat management on the forest. The management indicator species included gray squirrel (*Sciurus carolinensis*), eastern meadowlark (*Sturnella magna*), pileated woodpecker (*Dryocopus pileatus*), wood duck (*Aix sponsa*), black-throated green warbler (*Dendroica virens*), ruffed grouse (*Bonasa umbellus*), eastern bluebird (*Sialia sialis*), upland sandpiper (*Bartramia longicauda*), white-tailed deer (*Odocoileus virginianus*), black bear (*Ursus americana*), American redstart (*Setophagus ruticilla*), and ovenbird (*Seiurus aurocapillus*).

Habitat suitability models existed for all of the aforementioned species. Models developed by researchers affiliated with the U.S. Fish and Wildlife Service included those for the gray squirrel (Allen 1987), eastern meadowlark (Schroeder and Sousa 1982), wood duck (Sousa and Farmer 1983), pileated woodpecker (Schroeder 1983),

and black bear (Rogers and Allen 1987). The model for ruffed grouse was developed specifically for Michigan by Hammill and Moran (1986). A model for ovenbird was derived from Capen et al. (1986) (Appendix B). The remaining models were developed at Michigan State University through literature reviews and consultation with experts and followed the protocol of the U.S. Fish and Wildlife Service (1981). These models included black-throated green warbler (Bender et al. 1989; Appendix C), white-tailed deer (Bender and Haufler 1990; Appendix D), eastern bluebird (Moses et al. 1990; Appendix E), upland sandpiper (Roloff and Haufler 1990; Appendix F), and American redstart (Minnis and Haufler 1991; Appendix G).

Vegetation data used to run HSI models were collected from a stratified random sampling design using a variety of plot, line intercept, and ocular estimation techniques (Table 14). Sampling was stratified according to Corporate Database cover type (U.S. Forest Service 1988) and underlying ecological land type (Cleland et al. 1993). Replicates were the cover type-ecological land type stands, and each replicate contained a minimum of 3 samples. The wildlife habitat database¹ consisted of 56 measures of vegetation structure and composition derived from overstory, understory, dead and downed, ground, and wetlands categories. Arithmetic means of vegetation attributes for each cover type-ecological land type (Appendix A) combination were used for habitat model input.

To facilitate the computation of HSI values, Corporate Database cover type and ecological land type maps were digitized and habitat models were programmed into an

¹Detailed documentation of the wildlife habitat database is provided in Roloff and Haufler (1994), U.S. Forest Service Huron-Manistee National Forest, Corporate Database and Ecological Classification System: Vegetation Databases and Documentation, Unpublished Report. Cadillac, MI.

Table 14. Vegetation variables and associated sampling techniques. Methodology for each technique is described by Hays et al. 1981.

Vegetation Variable	Sampling Technique
Overstory¹	
Basal Area (total and by species)	10 X 25 m (0.06 ac) Plot
No. Tree Species	10 X 25 m (0.06 ac) Plot
No. Trees	10 X 25 m (0.06 ac) Plot
Height	Haga Altimeter
No. Cavities	10 X 25 m (0.06 ac) Plot
Coverage	25 m (82 ft) Line Intercept
Diameter	Diameter Tape
Understory	
Height Lowest Live Conifer Branch	Meter Stick
Coverage > 5 m Tall	10 X 25 m (0.06 ac) Plot
Coverage > 3 m Tall	10 X 25 m (0.06 ac) Plot
Coverage < 3 m Tall	10 X 25 m (0.06 ac) Plot
No. Stems	2 X 25 m (0.01 ac) Plot
Coverage Soft Mast	25 m (82 ft) Line Intercept
No. Soft Mast Species	25 m (82 ft) Line Intercept
Dead and Downed	
Snag Density (Total and by species)	10 X 100 m (0.25 ac) Plot
Snag DBH	Diameter tape
No. Cavities	10 X 100 m (0.25 ac) Plot
No. Logs	10 X 100 m (0.25 ac) Plot
No. Stumps	10 X 100 m (0.25 ac) Plot
Ground	
Distance to Passerine Perch Site	Pacing
Coverage	10 m (33 ft) Line Intercept
Coverage of Grass	10 m (33 ft) Line Intercept
Height of Spring Growth	Meter Stick
Wetlands	
Coverage of Water by Emergents/Debris	Ocular Estimate
Snag Availability	Ocular Estimate

¹ **Overstory**=All trees ≥ 11.4 cm (4.5") diameter breast height (DBH); **Understory**=All woody stems < 11.4 cm (4.5") DBH; **Dead and Downed**=Snags ≥ 11.4 cm (4.5") DBH, Logs ≥ 18 cm (7.1") DBH, Stumps ≥ 30 cm (11.8") tall and 11.4 cm (4.5") DBH; **Ground**=All species associated with the forest floor (typically ≤ 1 m tall).

Arc/Info geographic information system (GIS)¹. Both sets of maps were obtained from U.S. Forest Service field offices. Base map coverages (e.g., roads, lakes, rivers) were obtained from Michigan State University, Center for Remote Sensing. Cover type and ecological land type maps were verified at each sample point, and discrepancies were updated. Habitat models were programmed using the Arc/Info macro language.

Wildlife population indices were collected in 1991 and 1992. Songbird surveys (Robbins 1981a,b, Skirvin 1981, Askins and Philbrick 1987, O'Brien 1990), ruffed grouse drumming counts (Petraborg et al. 1953, Davis and Winstead 1980, Robbins 1981a,b, Rodgers 1981, O'Brien 1990), pileated woodpecker surveys, deer pellet counts (Neff 1968, Collins 1981, Rowland et al. 1984), and squirrel time-area counts (Bouffard and Hein 1978) were conducted using the point-count method (Ryder 1986; p. 304). All point surveys were standardized as to weather, time, and observer to minimize sampling bias (Ralph and Scott 1981, pp. 251-325).

North-south transects (1.6 km long) were delineated across township sections. Each township section contained 3 transects spaced a minimum of 0.4 km apart. Eight points were systematically located ≥ 150 m apart along the transects (Freemark and Merriam 1986, O'Brien 1990). Sample points from 1991 and 1992 were independently located.

Two different approaches were used to validate HSI's, depending on the home range of the wildlife species (Table 15). For species having home ranges ≤ 10 ha, a home-range-sized circle was centered over each sample point and an HSI was computed.

¹Detailed documentation of the modeling process is provided in Roloff and Haufler. (1994) U.S. Forest Service Huron-Manistee National Forest, Wildlife Habitat Management Procedure: Documentation, Unpublished Report. Cadillac, MI.

Table 15. Wildlife species, home range estimates, and supporting research.

Wildlife Species	Home Range Estimate (ha)	Citation
American Redstart	1.5	Sturm 1945, Ficken 1962, Samson and Lewis 1979, Sherry 1979
Ovenbird	1.5	
Eastern Bluebird	1.5	Pinkowski (1976)
Black-throated Green Warbler	1.5	Kendeigh (1945), Morse (1976), Sabo (1980)
Upland Sandpiper	10 ¹	Bowen (1976), Ailes (1980)
Gray Squirrel	0.4	Nowak (1991)
Pileated Woodpecker	70	Renken (1988)
White-tailed Deer	173	Larson et al. (1978)
Ruffed Grouse	16	Cade and Sousa (1985)
Black Bear	10,000	Norton (1981), Manville (1983), Garshelis (1986)

¹Includes only the area required for nesting. The model calculations accounted for the amount of contiguous habitat surrounding the nest site.

The HSI score was subsequently compared to the number of individuals censused at the point. For species requiring > 10 ha, circles spatially corresponding to the home range were distributed across study sites (regardless of sample point location) and HSI's were computed for each circle. The mean number of individuals censused per point located within each circle was compared to the corresponding HSI score. HSI values were computed using the area-weighting procedures as described by the U.S. Fish and Wildlife Service (1981).

Home-range circles were excluded from the analysis if vegetation data were unavailable for < 75% of the area. Although most cover type-ecological land type combinations were inventoried, several combinations were not found in sufficient numbers or area to warrant sampling. Also, the GIS overlay procedure develops "sliver polygons" that may not truly exist. It was determined that excluding areas without data rather than counting them as unsuitable habitat provided a better representation of habitat quality.

Numerous researchers have cautioned against using singing bird surveys to index bird populations (Willson 1974, Bart and Schoultz 1984, Sieg 1991), but the technique remains the most practical for landscape assessments. Songbird surveys were conducted 3 times annually at all sample points from sunrise to 1000 (Skirvin 1981, Askins and Philbrick 1987), 15 May to 15 June. Only bird species selected for this analysis were recorded at each point to minimize the "saturation effect" (Bart and Schoultz 1984). Individual observers would remain at each point for 5 minutes tallying all unique individuals seen or heard.

Point surveys for ruffed grouse and pileated woodpeckers were conducted from sunrise to 1000, 25 April to 5 May, annually. Grouse and woodpecker surveys were conducted at alternating points along the transects to minimize duplicate counts (Rodgers 1981). Observers remained at each point 5 minutes, recording all unique grouse and pileated woodpeckers seen or heard. Random sightings of grouse were also recorded as to location on

field maps.

Deer pellet group counts were annually conducted prior to leaf-out in a circular 40 m² plot at each wildlife sampling point. Pellet groups were defined as consisting of ≥ 5 pellets (Rowland et al. 1984), and pellet groups lying on the edge of a plot were systematically included or excluded in the tally.

Population indices of gray squirrels proved difficult to obtain. Several survey techniques were used including a modified time-area count method (Bouffard and Hein 1978) in 1991. The original method involved estimating distances to observed squirrels, however, the modified method only documented the number of squirrels seen. Squirrels were censused once at alternating sample points for 20 minutes from sunrise up to 2 hours later, 7 June to 7 July. The time area count yielded poor results, thus, random sightings and live-trapping were used to index squirrel populations in 1992. Live traps were systematically placed in specific cover type-ecological land type combinations and monitored daily for 6 trap nights. All captured gray squirrels were marked with a temporary dye and released. Random sightings of gray squirrels were recorded as to location on field maps. The total number of squirrels from all census techniques per cover type-ecological land type combination was the population index.

Wood duck population indices were obtained through brood counts (Lokemoen 1982) on 29 wetlands throughout the "Hoist Lakes" area. Counts were conducted from 0530 to 0900 hours, 10 to 17 July and 8 to 15 August, corresponding to the periods during which wood duck broods are most visible (Bellrose 1976). Counts occurred from predetermined points (maximizing wetland visibility) along wetland edges for 10-minute periods. The sum of July and August counts was the population index for each wetland.

Association between HSI values and population indices were tested using

Spearman's Rank Correlation Coefficient (Siegel 1956, O'Brien 1990). The Spearman's test provides the statistic *Rho* as a measure of association between 2 variables and a probability of significance can also be computed (Siegel 1956). Correlations were performed for individual years and for years pooled. To help discern chance significance, Spearman's tests were performed on randomly generated data for 4 different sample sizes; 20, 100, 300, and 700. Randomly generated HSI values were between 0.00 and 1.00, and random population indices ranged from 0-5 (typical of any one sample point). Random numbers were drawn from a uniform data distribution.

To determine the proper spatial scale for habitat model applicability, HSI values were calculated at several spatial scales (ranging from 0.1 to 182.4 ha) for wildlife models. Spatial scales deemed biologically inappropriate because of their small size (e.g. deer habitat use typically would not be assessed at small (0.1-7.3 ha) spatial scales) were excluded from the analyses (Laymon and Reid 1986). The results were plotted on histograms with "percentage of total area evaluated" on the ordinate, "HSI values" (grouped by 0.10's) on the abscissa, and "spatial scale" on the z-axis. The spatial scale at which the most habitat quality categories were delineated (i.e. the discriminatory capability of the model was maximized) was deemed the proper spatial scale for model applicability. This analysis applied only to final model output and assumed that vegetative structures (i.e. model components) were consistently perceived and responded to by wildlife species regardless of spatial scale.

RESULTS

Population indices were obtained for 9 of the 12 management indicator species. No eastern meadowlarks and too few eastern bluebirds were censused to warrant statistical analysis. Population indices for black bears were not estimated due to the lack of practical field techniques. Only one year of suitable data was obtained for some species including pileated woodpecker, black-throated green warbler, and wood duck (Table 16). Population indices per census point ranged from 0-6 for songbirds, 0-3 for pileated woodpeckers, 0-15 for deer, and 0-4 for ruffed grouse. Two wood duck population indices were obtained, ranging from 0-2 for number of broods and from 0-9 for ducklings per wetland. The number of squirrels censused in cover type-ecological land type combinations ranged from 0-11.

HSI values ranged from 0.00-1.00 for the American redstart, ovenbird, deer, squirrel, and wood duck models. HSI's ranged from 0.00-0.75 for upland sandpiper and ruffed grouse and between 0.00-0.60 for pileated woodpecker and black-throated green warbler. HSI values were calculated at spatial scales corresponding to the home range requirements of each species (Table 17).

Spearman correlation tests for 1991 and 1992 produced variable results. The most consistent (across years) performing model was the American redstart (Table 16). All other songbird models exhibited either non-significant correlations or inverse correlation coefficients (Table 16). Correlation coefficients for combined years were all significant ($P \leq 0.10$) (Table 16).

Although the deer models appeared to provide significant positive correlations with population indices (Table 16), some of the significance tests may have been due to

Table 16. Spearman rank correlation coefficients, associated probability, and sample size for tests between habitat suitability indices and wildlife population indices. Probability was one-tailed and tested if the correlation coefficient equaled 0 (Siegel 1956).

Wildlife Species	Year					
	1991			1992		
	ρ^1	P	n	ρ^1	P	n
American redstart	0.180	0.005	299	0.313	0.0005	394
Ovenbird	0.049	n.s. ²	298	0.253	0.0005	444
Black-throated Green Warbler	³	.	.	-0.008	n.s.	124 ⁴
Upland Sandpiper	.	.	.	-0.223	n.s.	23 ⁴
Pileated Woodpecker	.	.	.	0.175	0.10	64 ⁴
Food/Cover Sub-model	.	.	.	0.130	n.s.	64 ⁴
Reproduction Sub-model	0.164	0.10	96	0.090	n.s.	130
Ruffed Grouse	.	.	.	0.131	0.025	226
Gray Squirrel	.	.	.	.	.	.
Winter Food Sub-model	.	.	.	0.254	0.005	122
Cover/Reproduction Sub-model	.	.	.	0.307	0.0005	122
White-tailed Deer	.	.	.	.	.	.
Fall-winter Food Sub-model	-0.427	0.10	11	0.514	0.025	20
Security Cover Sub-model	-0.060	n.s.	11	0.368	0.10	20
Spring Food Sub-model	0.644	0.025	11	0.065	n.s.	20
Wood Duck	.	.	.	.	.	.
HSI Value to Broods	.	.	.	0.200	n.s.	29
HSI Value to Ducklings	.	.	.	0.144	n.s.	29

¹ ρ = Spearman rank correlation coefficient, P = one-tailed probability, n = sample size.

²Insufficient census data collected.

³Not significant.

⁴Relatively few individuals were censused.

Table 17. Spatial scale(s) of wildlife model applicability. Estimates are based on subjective evaluations of histograms depicting the range of HSI values and their evenness across the landscape for different spatial scales.

Wildlife Model	Scale of Applicability (ha)	Home Range (ha)
American redstart	1.5	1.5
Ovenbird	1.5-7.3	1.5
Black-throated Green Warbler	1.5	1.5
Upland Sandpiper	. ¹	10
Pileated Woodpecker		
Food/Cover Sub-model	16.4	70
Reproduction Sub-model	16.4	70
Ruffed Grouse	16.4-29.2	16
Gray Squirrel		
Winter Food Sub-model	0.1-29.2	0.4
Cover/Reproduction Sub-model	0.1-29.2	0.4
White-tailed Deer		
Fall-winter Food Sub-model	16.4	173
Security Cover Sub-model	16.4	173
Spring Food Sub-model	.	173

¹Scale of applicability could not be determined from histogram.

chance (Table 18). However, all of the significant ($P \leq 0.05$) deer correlation coefficients were positive (Table 16), unlike the random analysis (Table 18). Also, all of the deer models combined across years provided significant ($P \leq 0.10$), positive correlations.

The gray squirrel and ruffed grouse models for combined years exhibited positive, higher probabilities of significance ($P \leq 0.025$) (Table 16) than random analyses on similar sample sizes (Table 18). The correlation coefficients for pileated woodpecker and wood duck models were also positive, however, the association could not be discerned from chance (Table 16). The black-throated green warbler and upland sandpiper models yielded negative, non-significant correlation coefficients (Table 16).

The appropriate spatial scale for most habitat models corresponded to the species home range (Table 17). For example, the ovenbird model discriminated habitats equally (as indicated by the number of habitat classes) across a broad range of spatial scales (0.3-29.2 ha) (Fig. 5). The area of model applicability, however, was deemed as 1.5-7.3 ha (Table 17) because of the uneven distribution of area in each HSI class for 0.3, 16.4, and 29.2 spatial scales (Fig. 5). The effects of area weighting on the ovenbird HSI model are evident at scales larger than 70.0 ha as depicted by the convergence of HSI scores (Fig. 5).

Generally, the ability to discriminate habitats of different quality decreased as spatial scale increased (Figs. 5-14). These models typically converged into 2 or 3 HSI classes at large spatial scales (Figs. 5-14), a property related to the area-weighting averaging procedure used in HSI calculations (U.S. Fish and Wildlife Service 1981). Habitat models for wildlife species for which habitat assessments were based on the presence or absence of a specific cover type (e.g. openings) exhibited nominal output (i.e. the model only discriminated "good" and "bad" habitats). Models for upland sandpiper

Table 18. Spearman rank correlation coefficients, associated probability, and sample size for randomly generated habitat suitability indices (HSI's) and wildlife population indices. Probability was one-tailed and tested if the correlation coefficient differed from 0 by chance (Siegel 1956).

Simulation 1 (n=20)		Simulation 2 (n=100)		Simulation 3 (n=300)		Simulation 4 (n=700)	
<i>rho</i> ¹	<i>P</i>	<i>rho</i>	<i>P</i>	<i>rho</i>	<i>P</i>	<i>rho</i>	<i>P</i>
0.134	n.s. ²	-0.054	n.s.	0.011	n.s.	0.023	n.s.
0.347	0.05	0.042	n.s.	0.010	n.s.	-0.096	n.s.
0.087	n.s.	0.013	n.s.	0.003	n.s.	-0.020	n.s.
-0.029	n.s.	0.111	n.s.	-0.018	n.s.	-0.019	n.s.
-0.383	0.05	0.023	n.s.	0.041	n.s.	-0.035	n.s.
-0.268	n.s.	0.085	n.s.	0.008	n.s.	-0.005	n.s.
0.035	n.s.	-0.129	0.10	-0.014	n.s.	0.046	n.s.
-0.412	0.05	-0.088	n.s.	0.104	n.s.	0.008	n.s.
0.237	n.s.	-0.044	n.s.	-0.004	n.s.	0.012	n.s.

¹*rho* = Spearman rank correlation coefficient, *P* = one-tailed probability.

²Not significant.

Ovenbird

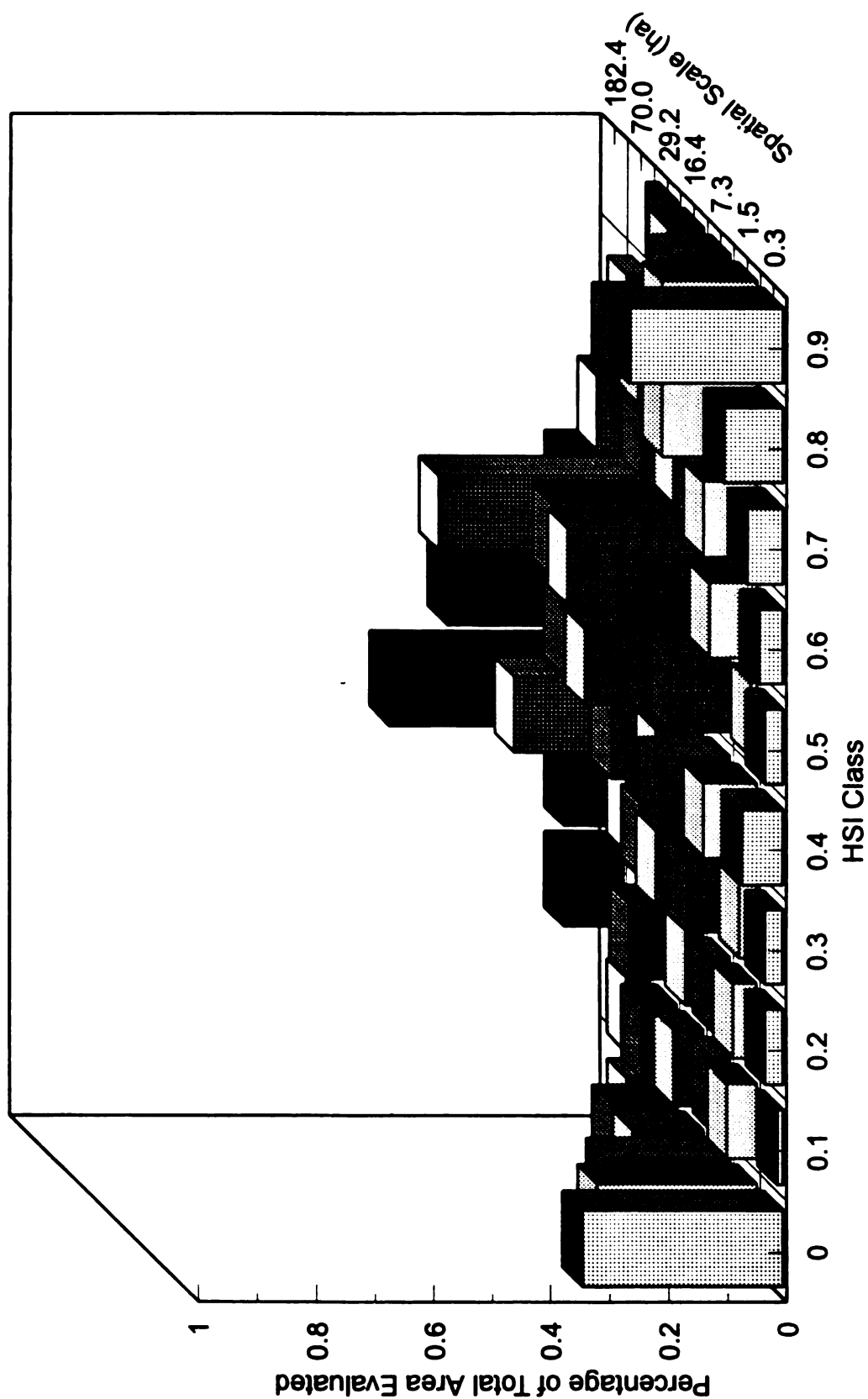


Figure 5. The influence of spatial scale on HSI model output for ovenbird.

Gray Squirrel - Cover/Reproduction

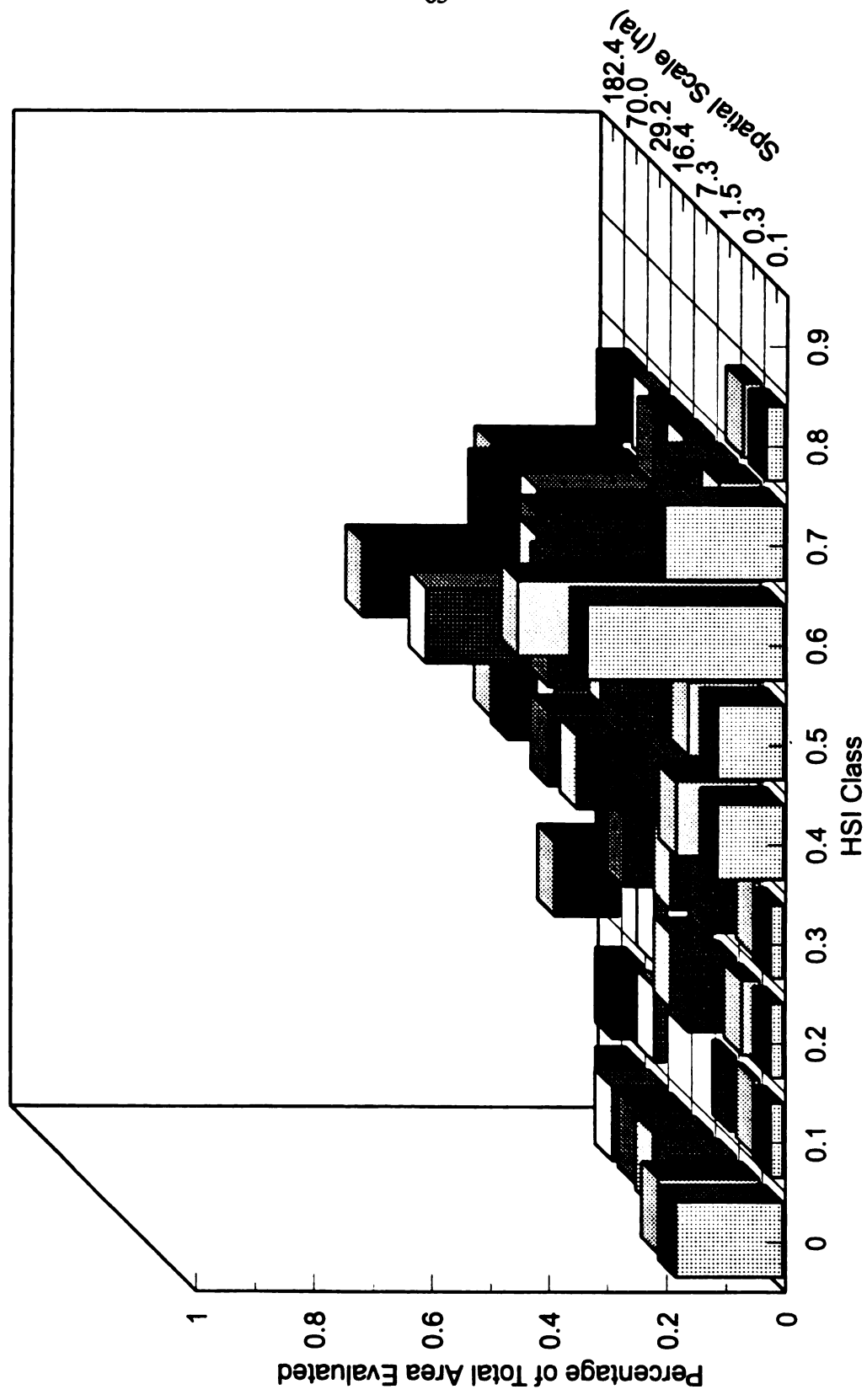


Figure 6. The influence of spatial scale on HSI model output for gray squirrel.

White-tailed Deer - Fall Winter Foods

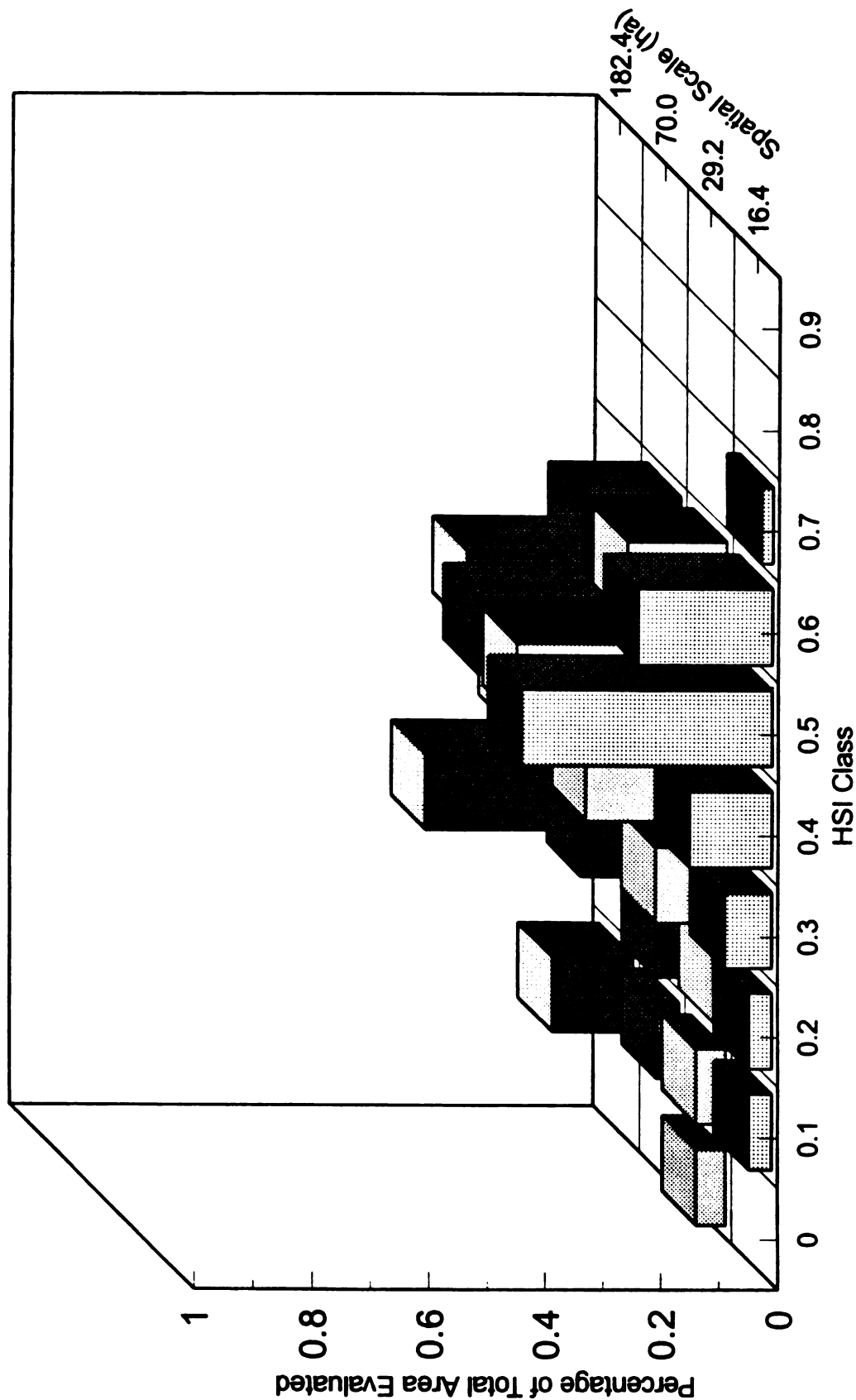


Figure 7. The influence of spatial scale on HSI model output for white-tailed deer fall-winter foods.

White-tailed Deer Security Cover

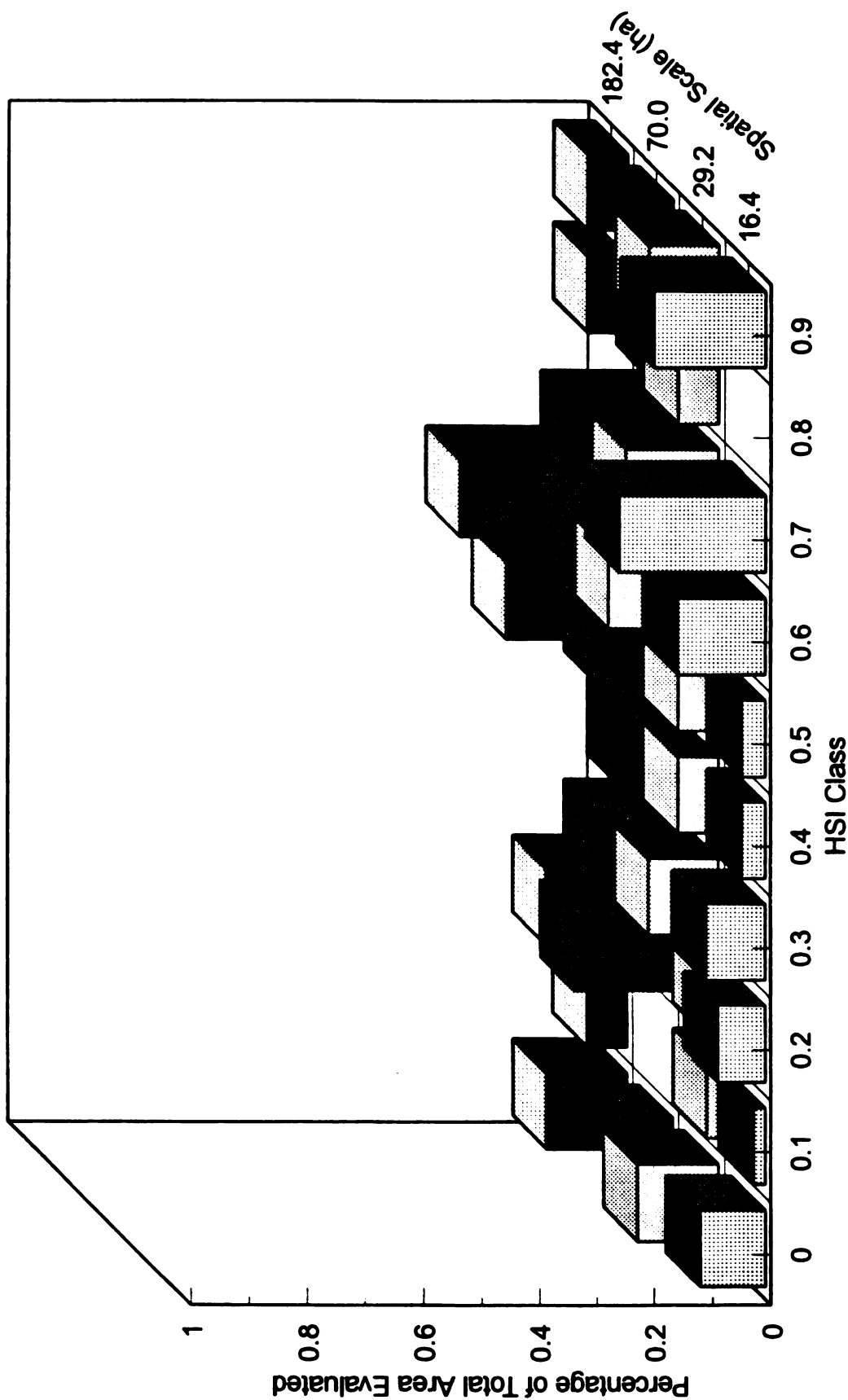


Figure 8. The influence of spatial scale on HSI model output for white-tailed deer security cover.

American Redstart



Figure 9. The influence of spatial scale on HSI model output for American redstart.

Black-throated Green Warbler

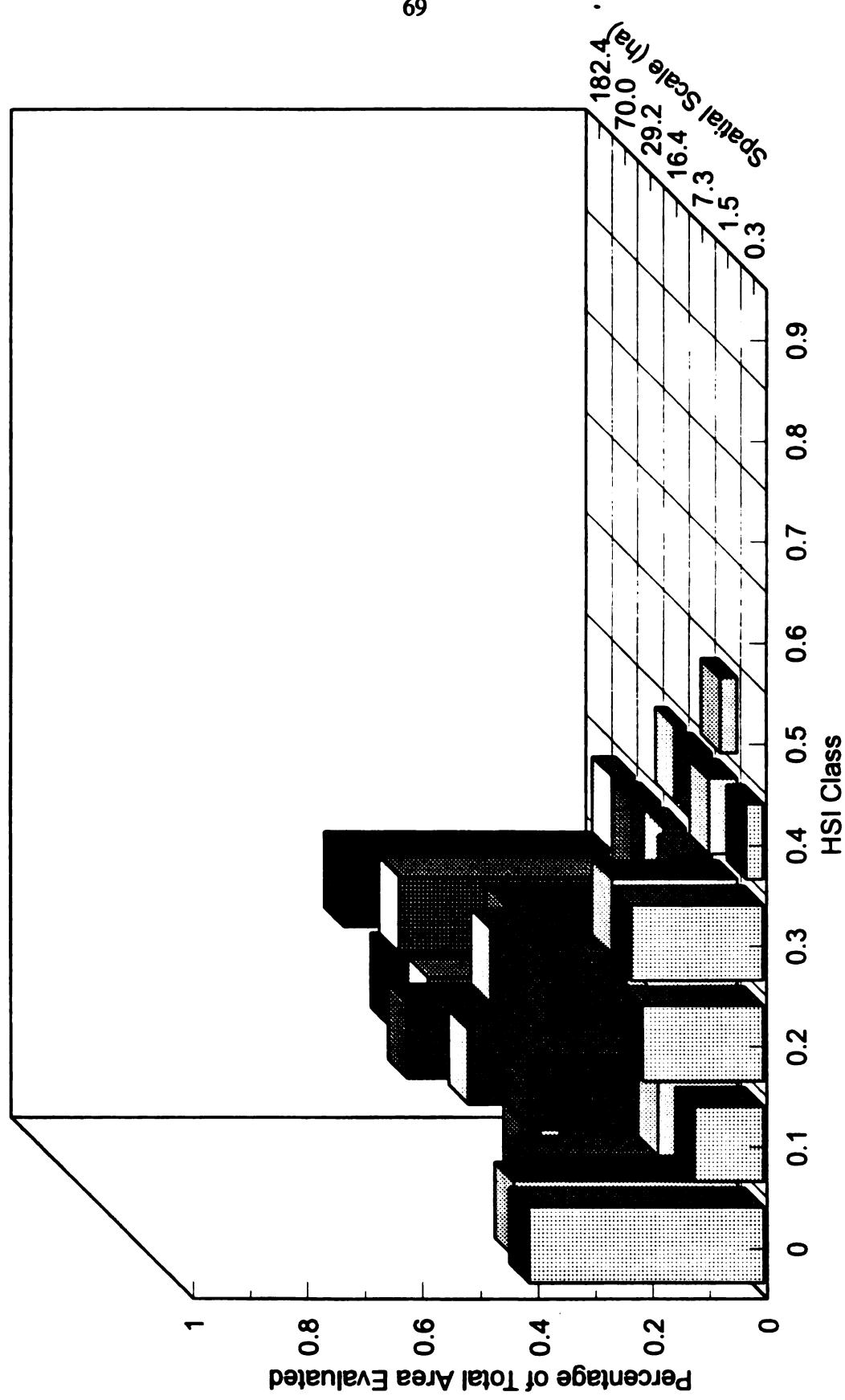


Figure 10. The influence of spatial scale on HSI model output for black-throated green warbler.

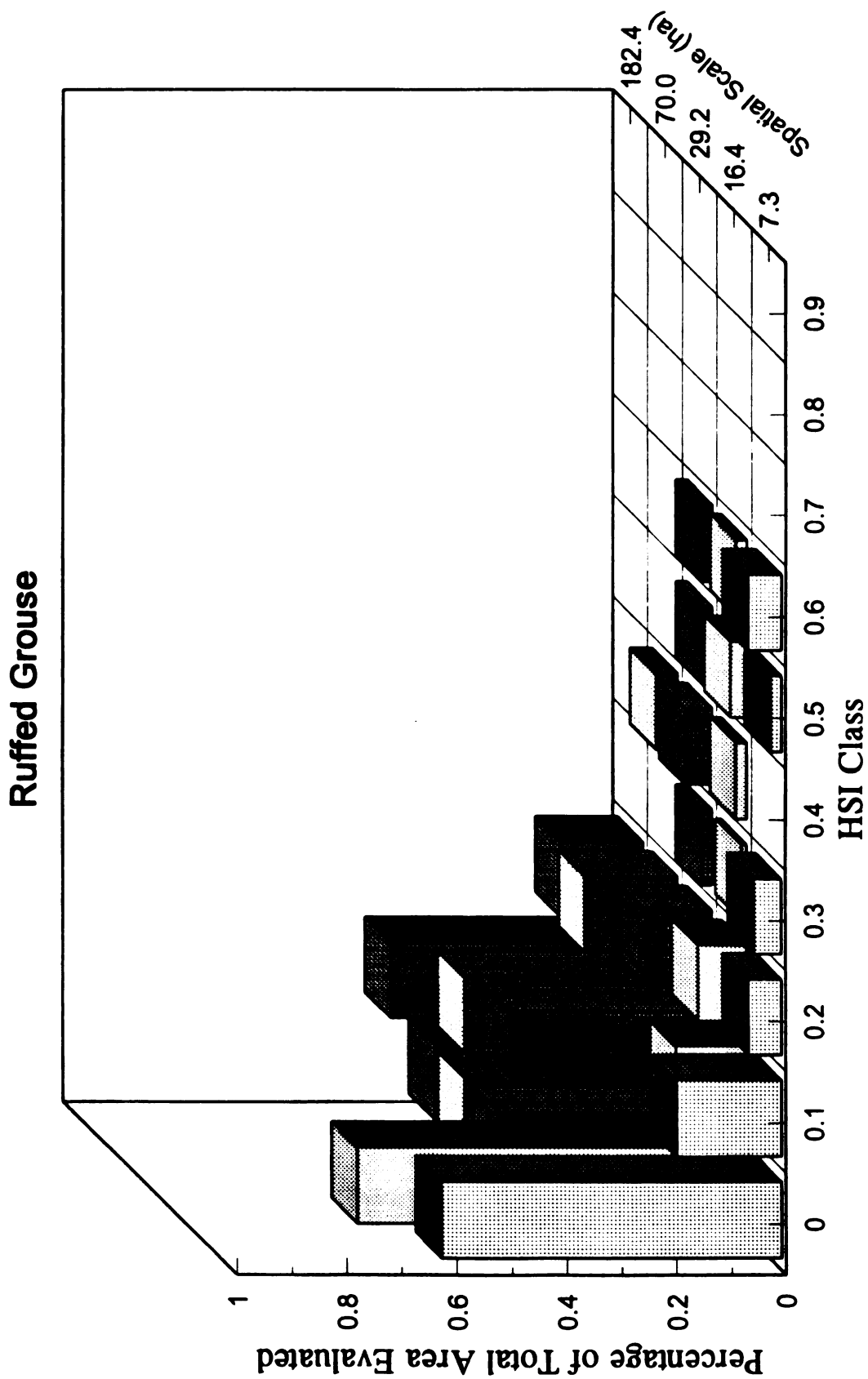


Figure 11. The influence of spatial scale on HSI model output for ruffed grouse.

Pileated Woodpecker - Cover/Reproduction

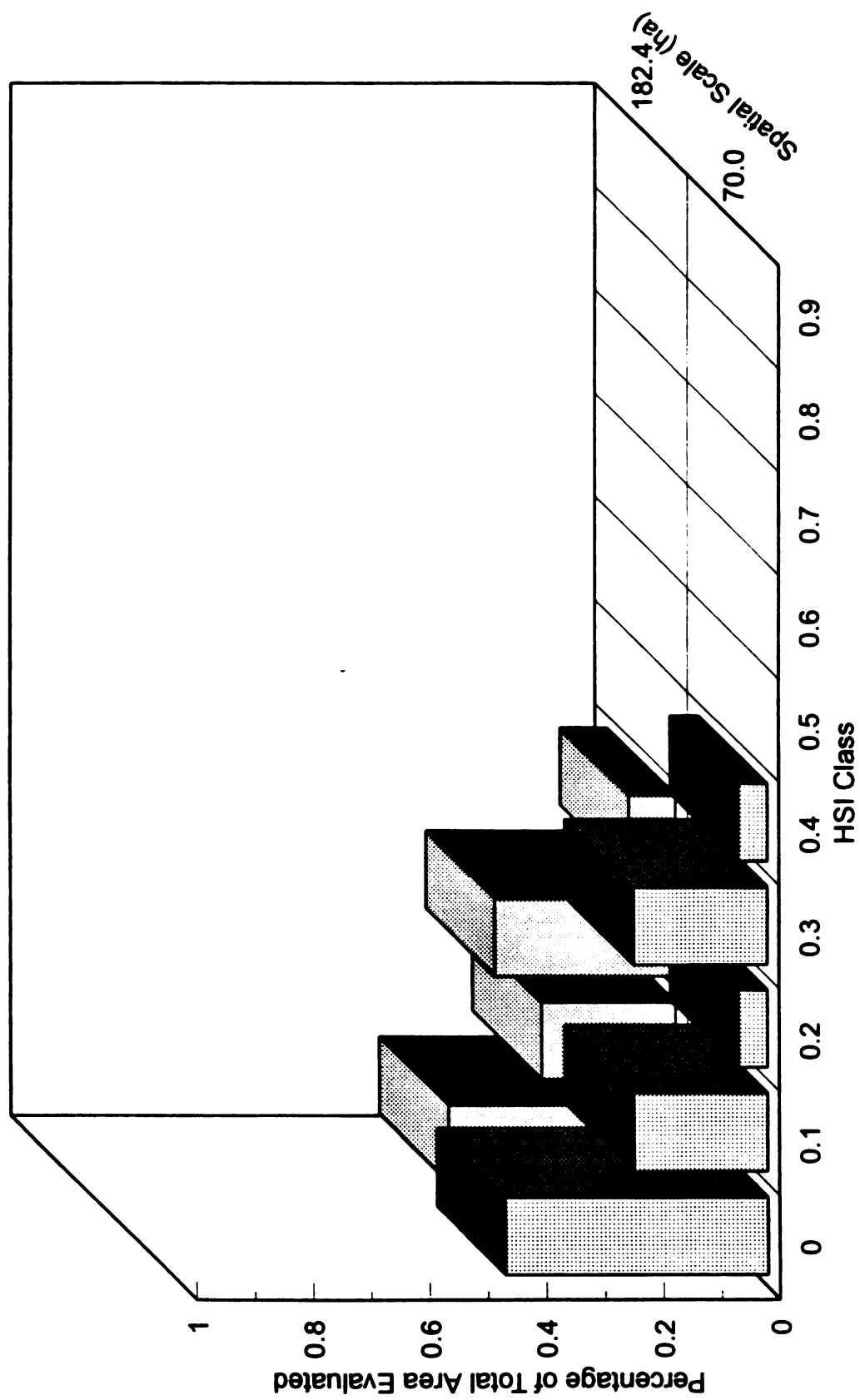


Figure 12. The influence of spatial scale on HSI model output for pileated woodpecker cover and reproduction.

Pileated Woodpecker - Food

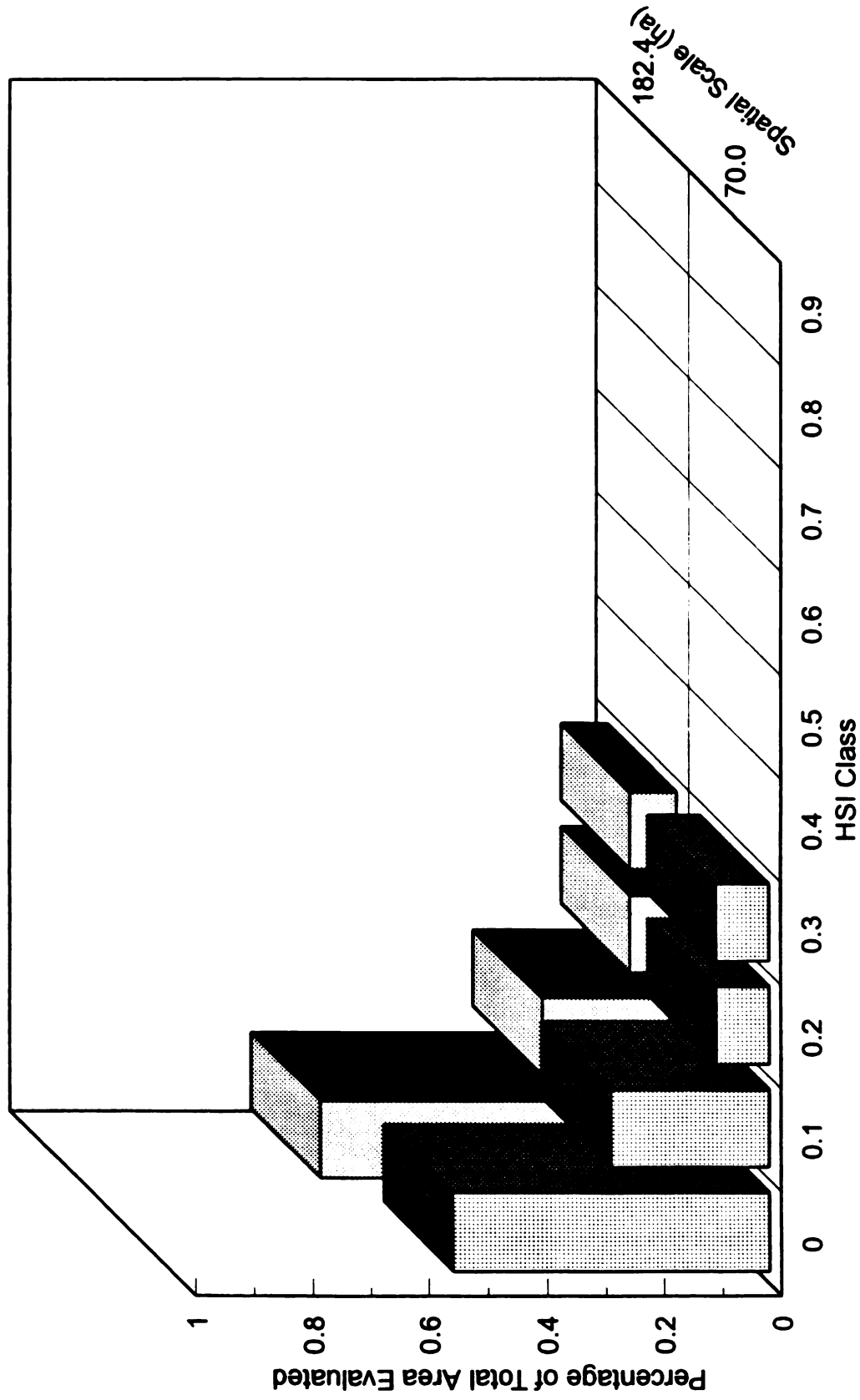


Figure 13. The influence of spatial scale on HSI model output for pileated woodpecker food.

Gray Squirrel - Winter Food

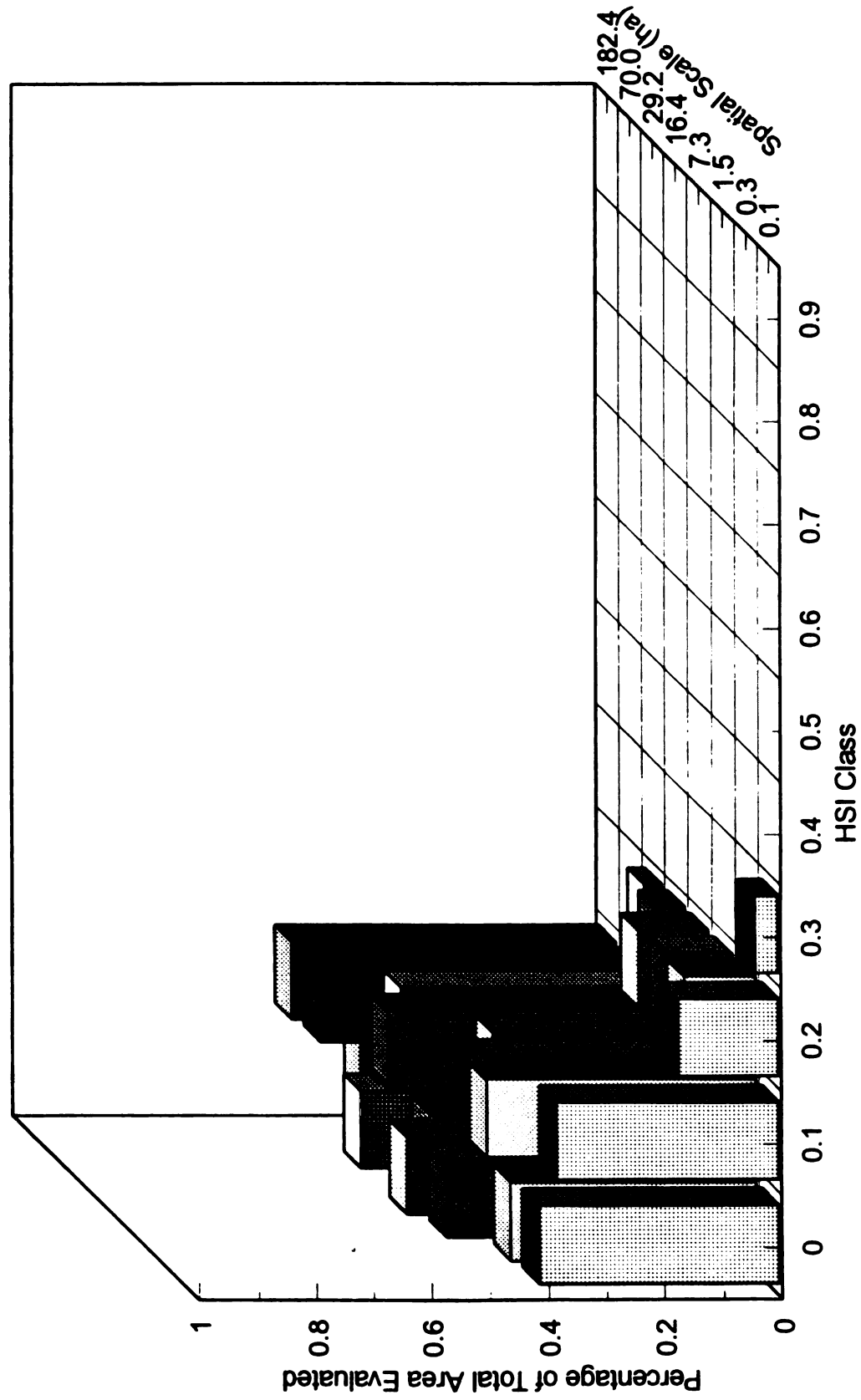


Figure 14. The influence of spatial scale on HSI model output for gray squirrel winter food.

(Fig. 15) and white-tailed deer spring food sub-model (Fig. 16) provided nominal output.

The histograms also proved to be useful for evaluating the quantity of different habitat categories. At the appropriate spatial scale the ovenbird model produced a relatively even distribution of HSI categories (Fig. 5). Most of the other models were skewed either high or low. Species for which more high quality habitat existed were gray squirrel cover/reproduction sub-model (Fig. 6) and white-tailed deer fall/winter foods and security cover sub-models (Figs. 7-8). Species for which low quality habitat predominated the landscape included American redstart (Fig. 9), black-throated green warbler (Fig. 10), ruffed grouse (Fig. 11), pileated woodpecker cover/reproduction and food sub-models (Figs. 12-13), gray squirrel food sub-model (Fig. 14), and white-tailed deer spring food sub-model (Fig. 16).

DISCUSSION

Habitat Model Validation

Wildlife habitat modeling is an evolving process. As hypotheses about nature, models cannot be proven correct, but merely corroborated by experiences and tentatively accepted until proven false (Naylor and Finger 1967, Caswell 1976). Validation is a critical step in estimating the reliability of models for forest planning. Habitat model validation is difficult because there are no consistent standards relating to habitat quality and because the models are founded on concepts (e.g., limiting factors, carrying capacity) often viewed with ambiguity (Schamberger and O'Neil 1986). The approach of HSI models is generally valid in that habitat quality is likely to exhibit thresholds below which habitat becomes unsuitable and above which further changes make little difference in quality (Van Horne and Wiens 1991, Williamson and Lawton 1991).

Upland Sandpiper - Openings

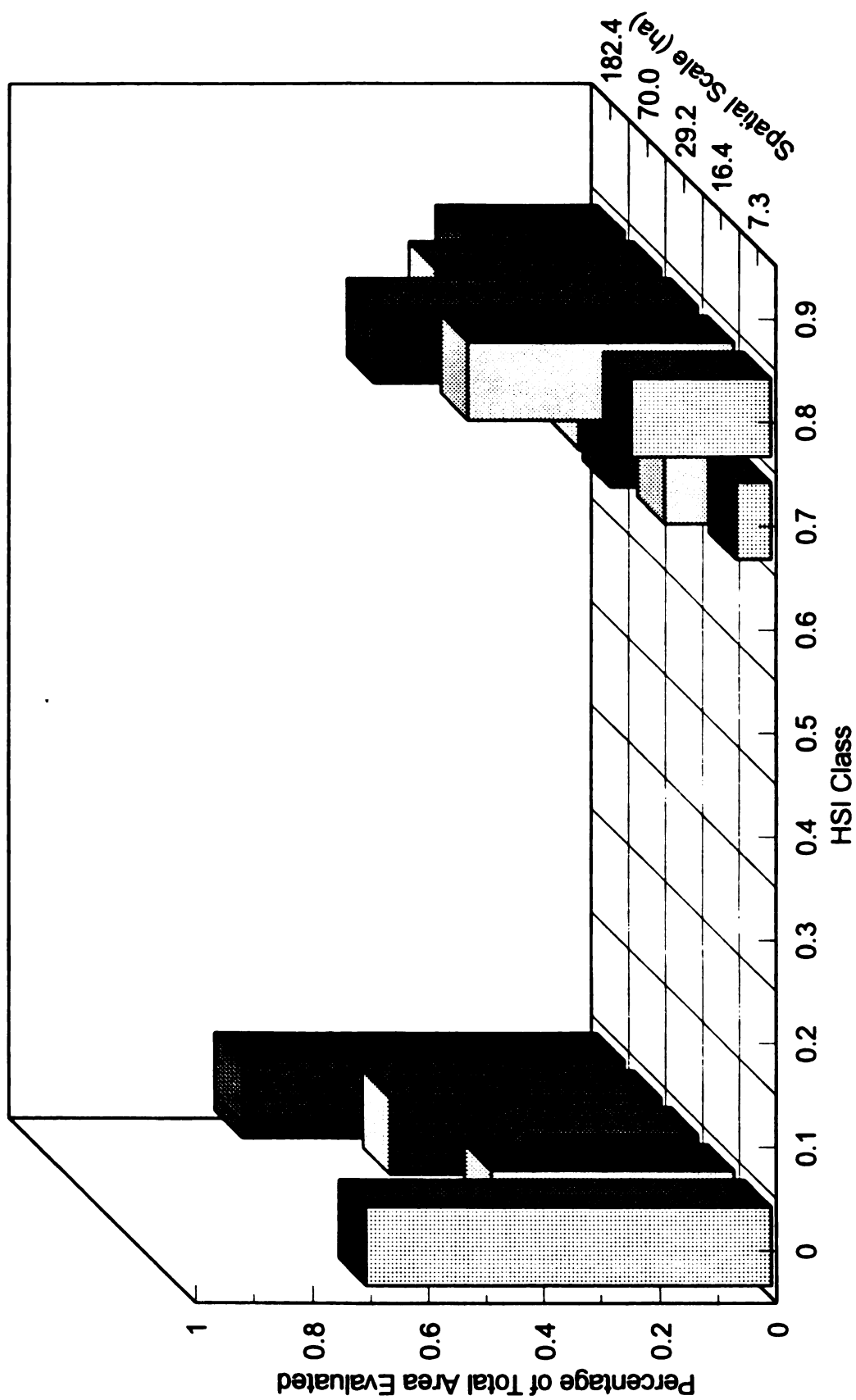


Figure 15. The influence of spatial scale on HSI model output for upland sandpiper openings.

White-tailed Deer Spring Food

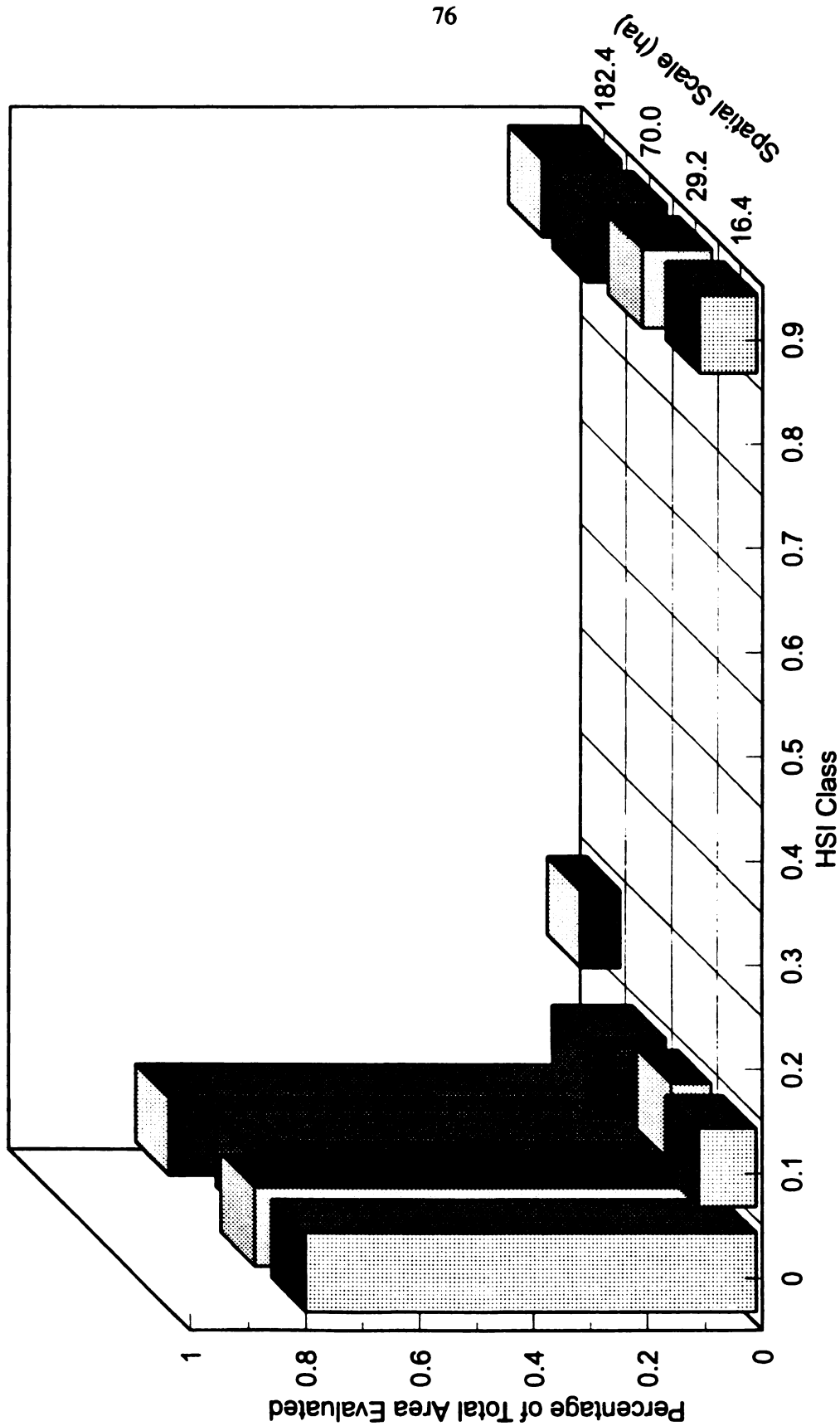


Figure 16. The influence of spatial scale on HSI model output for white-tailed deer spring foods.

The HSI models evaluated for this study generally exhibited positive associations with wildlife population indices. Most models that performed poorly were for wildlife species censused in low numbers (e.g., black-throated green warbler, upland sandpiper, pileated woodpecker)(Table 16). The upland sandpiper and pileated woodpecker use relatively large areas and can be detected over long distances, and Verner et al. (1986) demonstrated the difficulty in detecting changes in population numbers and habitat use for these types of species. The HSI approach does appear to be a valid and useful tool, particularly for comparisons of large sample sizes and for landscape habitat quantification (see "Chapter 3: Applying an Ecological Classification System and HSI Models to Landscape Planning").

Correlation coefficients of low magnitude may be attributed to measurement errors that accumulated throughout the validation process. For example, each model component is based on measures of vegetative structure or habitat spatial configurations. There is error inherent in sampling vegetation structure (both sampling error and error caused by the inherent variability of the vegetation) and in GIS maps. The conversion of vegetative structures into indices of habitat suitability is also a source of error in that the relationships depicted in most models are untested. When all of these components are mathematically combined into a final HSI score, the error is compounded. As a result, confidence limits around HSI values are probably quite large. Also, error inherent in sampling wildlife populations is common. Although human-induced sampling biases can be minimized, it is difficult (if not impossible) to account for natural variation in population detectability.

Cumulatively, the amount of error in HSI validation is large. The consistent performance of HSI models in this study is especially encouraging in considering the

potential error involved. The significance of the correlations indicated HSI models have good potential as comparative tools for forest planning or habitat mitigation. As sources of error in modeling and population estimation are reduced, higher correlation coefficients would be expected.

Pooling data across years tended to increase the consistency of population index and HSI association, often masking a poor within-year association (Table 16).

Comparisons to correlations on random numbers suggest that this phenomena is not simply a function of larger sample sizes, but rather, as sample size increases the affects of outliers have less influence on the correlation analysis.

Several characteristics of HSI modeling have been identified as sources of potential error: 1) the linear relationships between wildlife density and habitat parameters that do not reflect the inherent variability in these relationships over geographic regions (Bart et al. 1984, Maurer 1986, Johnson et al. 1991), 2) adequacy of observed density as an indicator of habitat quality (Van Horne 1983, Noon 1992), 3) inter- and intra-specific population interactions (Lancia et al. 1982, Flather and Hoekstra 1985, Schamberger and O'Neil 1986, Smith and Shugart 1987), 4) restricted model breadth (Farmer et al. 1982, Rotenberry 1985), 5) the method used to calculate final HSI values from model components (Bart et al. 1984, Van Horne and Wiens 1991), and 6) using means to describe vegetative attributes (Van Horne and Wiens 1991). Also, differences in individual animal behavior as to habitat preferences may introduce error (Cody and Walter 1976, Collins et al. 1982, Verner 1984). Undoubtedly valid, these concerns require attention in future modeling efforts, however, there is an immediate need for habitat evaluation tools in forest planning. Managers must understand that HSI models will likely only capture a portion (typically half or less) of the variation in population estimates (Morrison et al. 1992). Most managers are

willing to accept a limited degree of error, and past experiences suggest that the error inherent in HSI validity and landuse conversion functions are usually tolerable for strategic decision making (Lancia et al. 1986).

Although individual animals exhibit specific habitat selection behaviors (Maurer 1986), the basis for using animal abundance to assess habitat quality is that all members of a population have the same limiting habitat components, and although individual variation in habitat selection occurs, the average habitat selection pattern will be consistent with their limiting needs. Thus, for large samples, the individual behavior is averaged out.

Nonetheless, caution should be exercised when using animal abundance to assess habitat quality. A general consensus exists among researchers that better estimates of habitat quality are obtained through direct measures of population fitness, such as fecundity and survival (Van Horne 1983, Van Horne and Wiens 1991, Martin and Nur 1992, Noon 1992). Data on population demographics is extremely difficult and expensive to obtain, and currently, an impractical way to assess habitat conditions. Until more efficient population assessment techniques are developed, animal abundance will continue to be used as an indicator of habitat quality, not by choice, but by necessity.

Spatial Scale of HSI Model Applicability

The scale of space and time on which ecological systems are viewed influences the patterns and/or processes that are detected (Wiens 1986). Spatial and temporal characteristics of landscape structure potentially important to wildlife species include forest stand size, variability in habitat conditions among stands (heterogeneity), temporal dynamics in habitat use, and the regional context (extent and configuration) of forest stands in a landscape (Logan et al. 1983, Freemark 1988). To date, few model validation studies have evaluated habitats across landscapes, even though the effects of scale on model output is of

concern (Logan et al. 1983, Maurer 1986, O'Neill et al. 1986, Wiens 1986). Explicit consideration of multiple scales is essential for addressing the complexity of habitat selection, and our inability to deal with the complexity of the spatio-temporal framework may be one of the underlying causes of model failures (O'Neill et al. 1986). Model validations have been applied to spatial scales often associated with the organism's life requisites (e.g., home range, muskrat marshes) (Table 19). The assumption is made that the habitat model operates most appropriately at the chosen spatial scale, however, no quantitative support for this assumption is provided. HSI models are extremely sensitive to spatial scale (Van Horne and Wiens 1991), especially in heterogenous environments. For example, figure 17 demonstrates the response of area-weighted HSI values for 4 wildlife species at 10 sample points as size of the analysis area changes. The influence of scale on model output is related to the habitat configuration surrounding each point, explaining why some points (for individual models) exhibit 10-fold changes in HSI value whereas others remain relatively constant (Fig. 17).

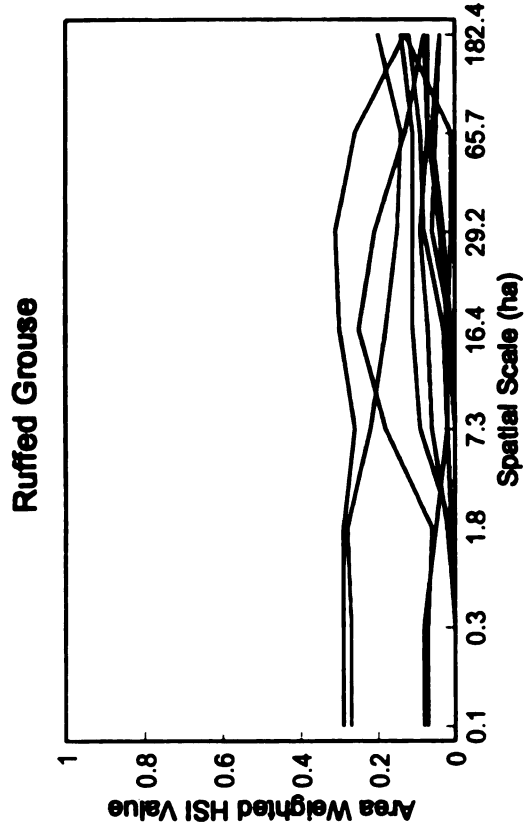
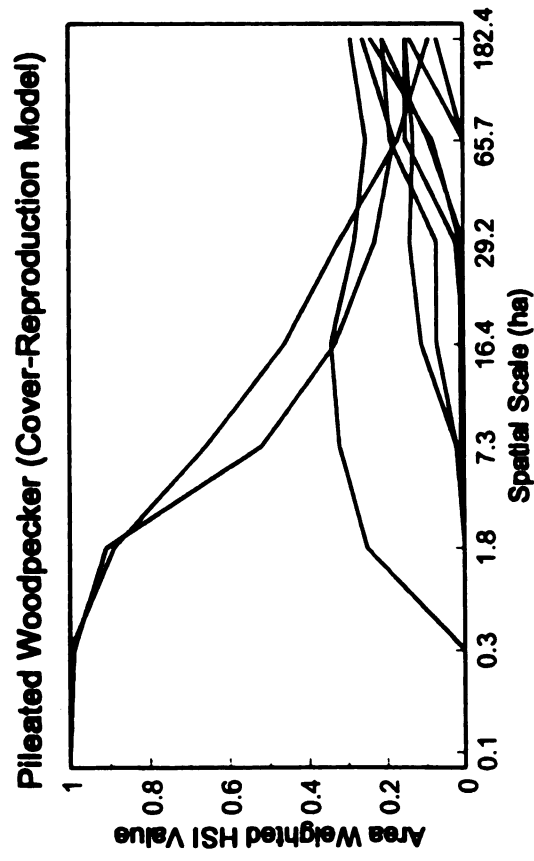
Vegetative heterogeneity provides a potentially complex basis for resource partitioning within and among coexisting species (Denslow 1985, Kaufmann and Landsberg 1990). In a vegetatively heterogenous analysis area, as spatial scale increases, the number of polygons included in the analysis increases. Scaling up output from a small-scale model is not just a matter of altering parameter values to account for the range of habitat conditions; there may also be additional feedbacks and interactions at higher organizational levels (Kaufmann and Landsberg 1990). The convergence of HSI values at large spatial scales follows standard statistical theory in that as sample size increases (i.e. the number of polygons in the analysis), the variance around the mean will decrease and the mean HSI becomes the "best" estimator of habitat quality for the landscape. At this scale,

Table 19. Studies validating habitat suitability index (HSI) models and the spatial scale at which the validation was performed.

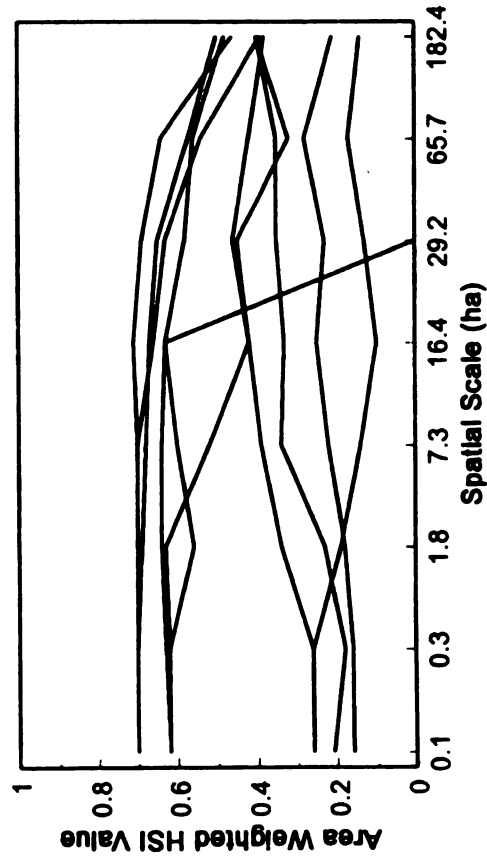
Wildlife Species	Area of Validation (ha)	Citation
Pronghorn Antelope (<i>Antilocapra americana</i>)	6,500-12,000	Cook and Irwin (1985)
Ruffed Grouse	66-151	Hammill and Moran (1986)
Clapper Rail (<i>Rallus longirostris</i>)	40	Clark and Lewis (1983)
Muskrat (<i>Ondatra zibethicus</i>)	80-280	Bart et al. (1984)
Gray Squirrel Fox Squirrel (<i>Sciurus niger</i>)	49	Seng (1991)
Beaver (<i>Castor canadensis</i>)	2.4 km ¹	Robel et al. (1993)
Lynx (<i>Lynx canadensis</i>)	1.25 ²	Lancia et al. (1982)
Eastern Phoebe (<i>Sayornis phoebe</i>) Red-eyed Vireo (<i>Vireo olivaceus</i>) Prairie Warbler (<i>Dendroica discolor</i>) Field Sparrow (<i>Spizella pusilla</i>) Meadow Vole (<i>Microtus pennsylvanicus</i>) White-footed Mouse (<i>Peromyscus leucopus</i>) Cottontail Rabbit (<i>Sylvilagus floridanus</i>)	2-11	Cole and Smith (1983)
Spotted Owl (<i>Strix occidentalis</i>)	16 ²	Laymon and Barrett (1986)
Douglas' Squirrel (<i>Tamiasciurus douglasii</i>)	4 ²	
Spotted Owl	4-16 ²	Laymon and Reid (1986)

¹HSI's computed per stream length.

²HSI's computed per map grid cell.



White-tailed Deer - Fall-Winter Food



American Redstart

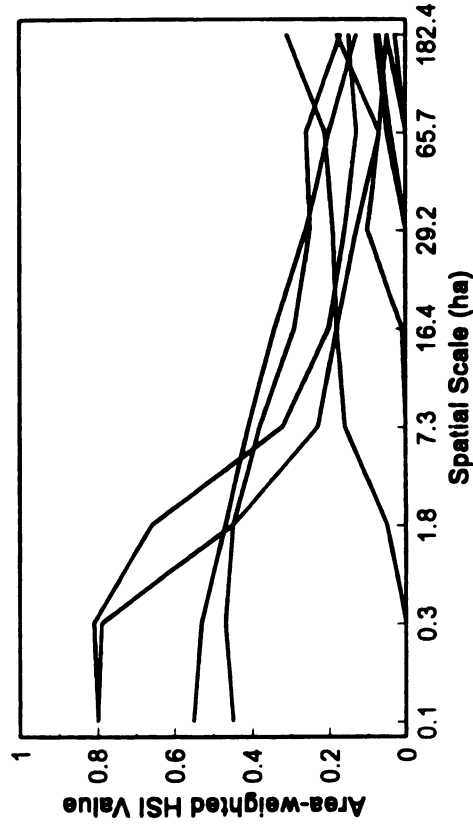


Figure 17. Area weighted HSI values for selected wildlife species at 10 randomly selected inventory points. HSI's were calculated for different spatial scales at each point.

heterogeneity in model output declines and the landscape is viewed as relatively homogenous (Welsh and Healey 1993). This approach, however, ignores the biological limits of habitat perception for the organism. Individuals do not respond to average HSI conditions for entire landscapes, but rather, they respond to habitat patches at a scale compatible with obtaining their life requisites (Wiens 1974, Rice et al. 1984, Mackie and Dusek 1992). Of additional importance is identifying areas providing "source" or "sink" habitats (Morrison et al. 1992). Average HSI scores across landscapes will disguise these important habitat considerations. If habitat models are developed to discriminate habitat quality best at biologically meaningful scales, then the scale at which between patch variation is maximized provides the "best" estimate of habitat quality.

The effects of spatial scale on model output may also be confounded by the mathematics within each model. For example, the mathematics of the grouse model make it difficult to obtain high quality ratings (Fig. 17), a characteristic often associated with models based on geometric means. Also, several models (e.g., ruffed grouse, white-tailed deer) contained spatial components (e.g. proximity of winter food to cover for grouse) that could potentially distort model output. For example, the HSI value for excellent ($HSI = 1.00$) grouse habitat within 16 ha may equal 0.00 if no winter food occurs within the 16 ha. To avoid these biases, spatial analyses should be conducted independently from model computations, i.e., the proximity analysis of winter food to cover within the 16 ha grouse area should be conducted prior to delineating the 16 ha analysis unit.

Prior to implementing habitat models, the proper spatial scale of model applicability should be determined. The geographic scale at which models are applied should reflect the size of the animal's home range, the degree of habitat specialization by the animal, the heterogeneity of the habitat, and the intended use of the model (Flather and Hoekstra 1985, Laymon and

Barrett 1986). The use of histograms to evaluate the behavior and consistency of models was demonstrated by Van Horne and Wiens (1991), however, their analyses did not incorporate varying spatial scales. Wildlife species have been documented to exhibit different habitat selection patterns over small geographic areas (Stauffer and Best 1986) and to detect differences in habitat quality, HSI models must be implemented at appropriate spatial scales.

SUMMARY AND CONCLUSIONS

The current regulatory environment effecting resource management decisions requires wildlife habitat assessments that are repeatable, scientifically credible, and legally defensible. The HSI approach conforms to these requirements, and the persistence of habitat evaluation procedures attests to their usefulness. There has been a general divergence of opinion regarding the reliability of HSI models; with many "researchers" demonstrating and questioning "poor" model performance (usually as defined by some pre-determined level of significance), in contrast to many managers who are excited about a tool capable of predicting animal distributions with 60% reliability. The results of this study support the contention that HSI models are a legitimate tool for increasing the reliability of habitat assessments. Recognizing that HSI models are not static and should evolve with the acquisition of new data (e.g. validation studies), the performance of HSI models should continue to improve.

An appealing characteristic of using HSI's integrated with GIS is the ability to incorporate the spatial requirements of an organism. Most animals require certain configurations of their life requisites to successfully reproduce and survive (e.g. the interspersions of food and cover for ruffed grouse), and these relationships can be quantified and displayed. Caution must be exercised when using HSI models in that the size of the evaluation area will influence the results. This study demonstrated that HSI models should be used at biologically meaningful scales (e.g. home range), the size of which will be influenced by an organism's needs and

heterogeneity of the habitats. Habitat assessments conducted on a stand-by-stand basis are no longer acceptable for many species, and the HSI-GIS tools can provide insights as to the effects of land management activities on these spatial constraints.

Chapter 3: Applying an Ecological Classification System and HSI Models to Landscape Planning

Ecosystem management has recently been adopted as the approach to land management on federal lands. Encompassing multiple spatial, temporal, and trophic levels, ecosystem management is a response to address social, economic, and biological issues including concerns about biodiversity, habitat fragmentation, population viability, and genetic heterozygosity. These issues cannot be addressed using stand-based, species by species assessments, but rather, a combination of coarse and fine filter approaches to resource management must be used.

Coarse filter approaches involve assessing resources by broad biological groupings such as "ecosystems" or "communities" (Hunter 1990, Morrison et al. 1992). Coarse filter approaches for resource management assume that a representative sample of biological groupings appropriately spaced across a landscape will insure ecological integrity, where *ecological integrity* is the interaction of the physical, chemical, and biological elements of a system in a manner that ensures long-term health and sustainability of the system (Environmental Protection Agency 1994, memo referencing the "Edgewater Consensus"). The problem with coarse filter approaches is that some ecosystem components are certain to slip through the analysis (Hunter 1990). The fine filter approach is designed to catch individual components known to be missed or inadequately represented by the coarse filter. For example, threatened and endangered plant and animal species are appropriate fine filter

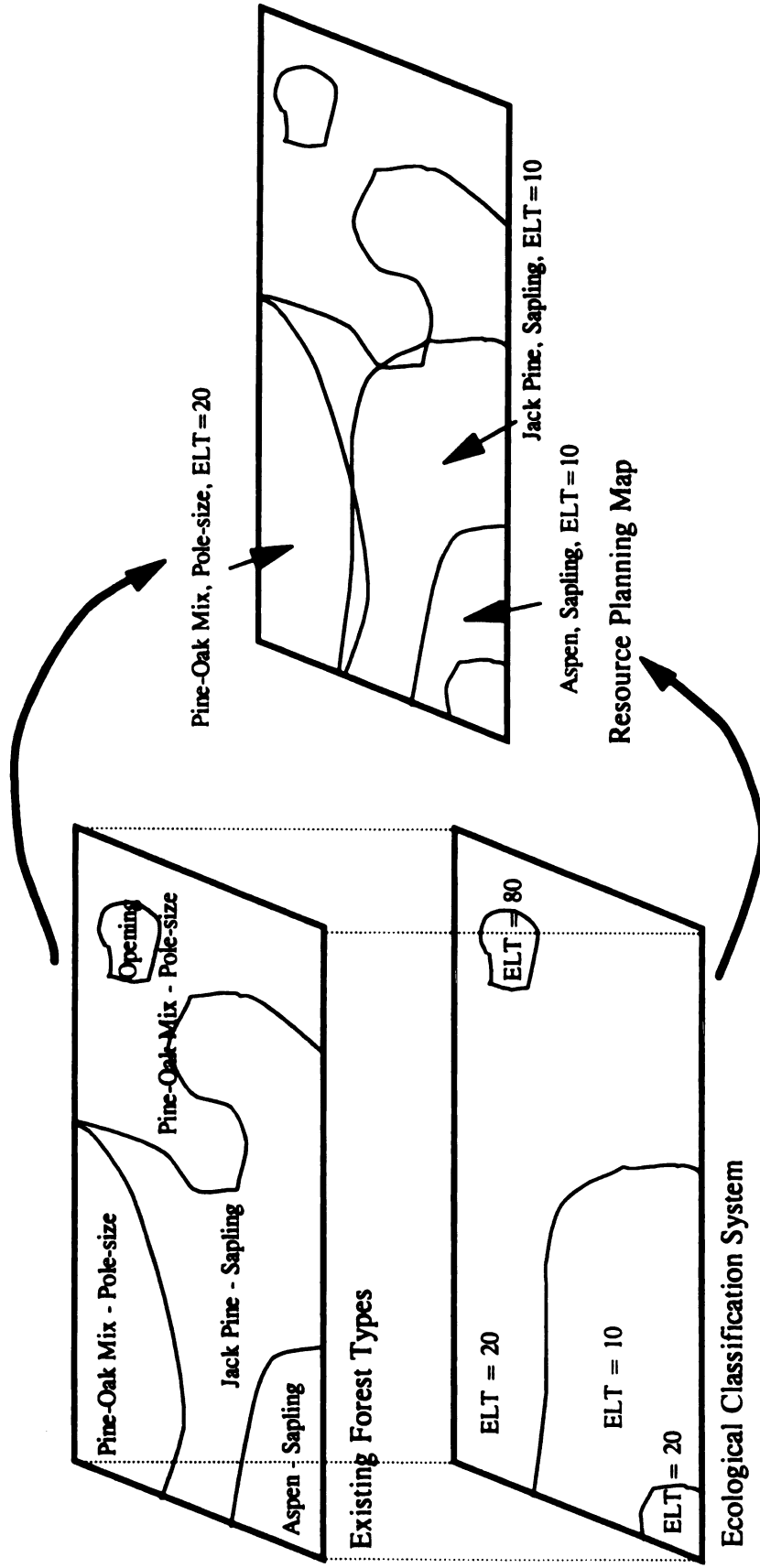
candidates in that land managers must meet their legal requirements in resource plans.

To implement a coarse filter-fine filter approach, managers need large amounts of data to account for wildlife objectives, and thus, the complexity of forest planning has increased. Databases created from an ecological classification system, used in conjunction with existing vegetative cover types, provide a coarse filter resource planning tool capable of integrating ecological components into a manageable entity (Fig. 18). Used in conjunction with HSI models (a fine filter tool) and GIS, the resource planning tool allows managers to: 1) quantify existing vegetation, 2) evaluate the correspondence of existing vegetation to ecological units (as identified by ECS), 3) develop management alternatives consistent with land capabilities (both plant and animal), 4) assess current wildlife conditions, 5) simulate the impacts of management alternatives, and 6) estimate short- and long-term dynamics of wildlife habitat across landscapes (Roloff and Haufler 1993).

The goal of this chapter is to review the ideas and concepts for landscape assessments of wildlife habitat that have resulted from this research project. This is not an exhaustive review of all landscape assessment tools, but rather, a summary of a methodology that incorporates the systems and tools evaluated for this project.

The Ecosystem Diversity Matrix

The foundation of an ecological, landscape-based approach to wildlife habitat management involves maintaining adequate representation of the diverse ecosystems inherent in our landscapes (Haufler 1994). Haufler (1994) proposed the use of an ecosystem diversity matrix for landscape assessments. The matrix is a coarse filter approach that allows for quantification of cover types and seral stage by ecological unit and provides a basis for identifying appropriate management options (Haufler 1994).



Resource Planning Map Capabilities:

1. Describe existing vegetation, overstory and understory.
2. Predict successional change.
3. Identify natural disturbance factors.

Figure 18. Overlay of existing vegetation and underlying ecological classification system to create a resource planning map.

Specifically, general vegetative types by ecological classification, age class (from which existing vegetative structure and composition can be described), and quantity can be identified from the matrix (Fig. 19). Integral components of the ecosystem diversity matrix are the successional linkages within ecological land type (Haufler 1994), e.g., aspen on ELT's in the 20's and 30's succeeds into oak and pine if undisturbed, and oak on the high 30's ELT succeeds into northern hardwoods (Fig. 19). The successional linkages depicted in figure 19 (i.e. down an ELT column) are crude by nature of the classification schemes. For example, oak on ELT 30 has multiple successional pathways (into an oak-pine, white pine, or northern hardwoods community) depending on ELTPG. Matrix cells containing vegetative cover types and age classes not typically supported by underlying ELT can also be identified (Fig. 19).

Vegetation structure can be estimated from cover type, age class, and underlying ELT; and more exact estimates can be obtained by incorporating existing vegetative inventories (e.g. the cover type-ELT database discussed in Chapter 2). Also, knowledge of disturbance regimes and successional pathways can be used to identify management options (Haufler 1994). For example, ELT's that historically supported mature cover types (e.g. northern hardwoods on ELT's in the 40's) can be identified and managed to support older forests. Similarly, ecological land types that historically exhibited frequent disturbances (e.g. jack pine on ELT's in the 0's and 10's) should not be managed on long timber rotations (Haufler 1994).

The ecosystem diversity matrix and associated cover type graphs for "Hoist Lakes", "Reed Lake", and the "Sandpiper Fields" are presented in figures 19-21. Eight major vegetative groupings were delineated (Fig. 19). Forest cover type categories were sub-divided into 3 age classes; sapling-, pole-, (11.4-25.4 cm), and saw-sized (> 25.4 cm)

Ecosystem Diversity Matrix

Existing Forest Cover Types	Age Class	Ecological Land Type										Total
		0	10	20	30	40	50	60	70	80		
Pine	Sapling	1288	92	40	6	0	0	3	0	0	1429	
Jack, Red, White	Pole	277	269	375	75	0	0	2	0	0	998	
Jack Pine-Black Oak	Saw	0	0	14	16	0	0	4	0	0	33	
Aspen	Sapling	4	25	601	635	2	9	9	0	60	1344	
Trembling, Big-tooth	Pole	8	27	413	571	3	6	7	0	63	1098	
	Saw	0	0	14	15	0	0	0	0	3	33	
Oak	Sapling	0	0	83	50	0	0	0	0	0	133	
Black, Red, White	Pole	49	42	595	1296	2	36	1	2	38	2061	
Red, White Pine-Oak	Saw	0	0	8	24	0	0	0	0	0	32	
Northern Hardwoods	Sapling	0	0	0	8	2	0	0	0	0	10	
Beech, Sugar Maple,	Pole	0	0	43	326	80	4	10	0	6	469	
Birch	Saw	0	0	0	0	0	0	0	0	0	0	
Lowland Deciduous	Sapling	0	0	0	0	0	0	0	0	0	0	
Black-ash	Pole	0	0	2	13	0	0	5	0	15	35	
	Saw	0	0	0	0	0	0	0	0	0	0	
Lowland Conifers	Sapling	0	0	0	0	0	0	0	0	0	0	
Cedar, Spruce	Pole	0	3	2	24	0	0	8	0	228	265	
	Saw	0	0	0	0	0	0	0	0	0	0	
Openings		0	6	55	88	0	6	2	3	88	247	
Total		1625	464	2243	3147	89	61	51	5	501	8185	



Denotes cells typically not supported by corresponding ELT (i.e. gaps in successional linkage).



Denotes the cover type where successional processes slow or are reset.

Figure 19. The ecosystem diversity matrix for Hoist Lakes and vicinity, Reed Lake, and Sandpiper Fields, Alcona County, MI. Cell contents are hectares.

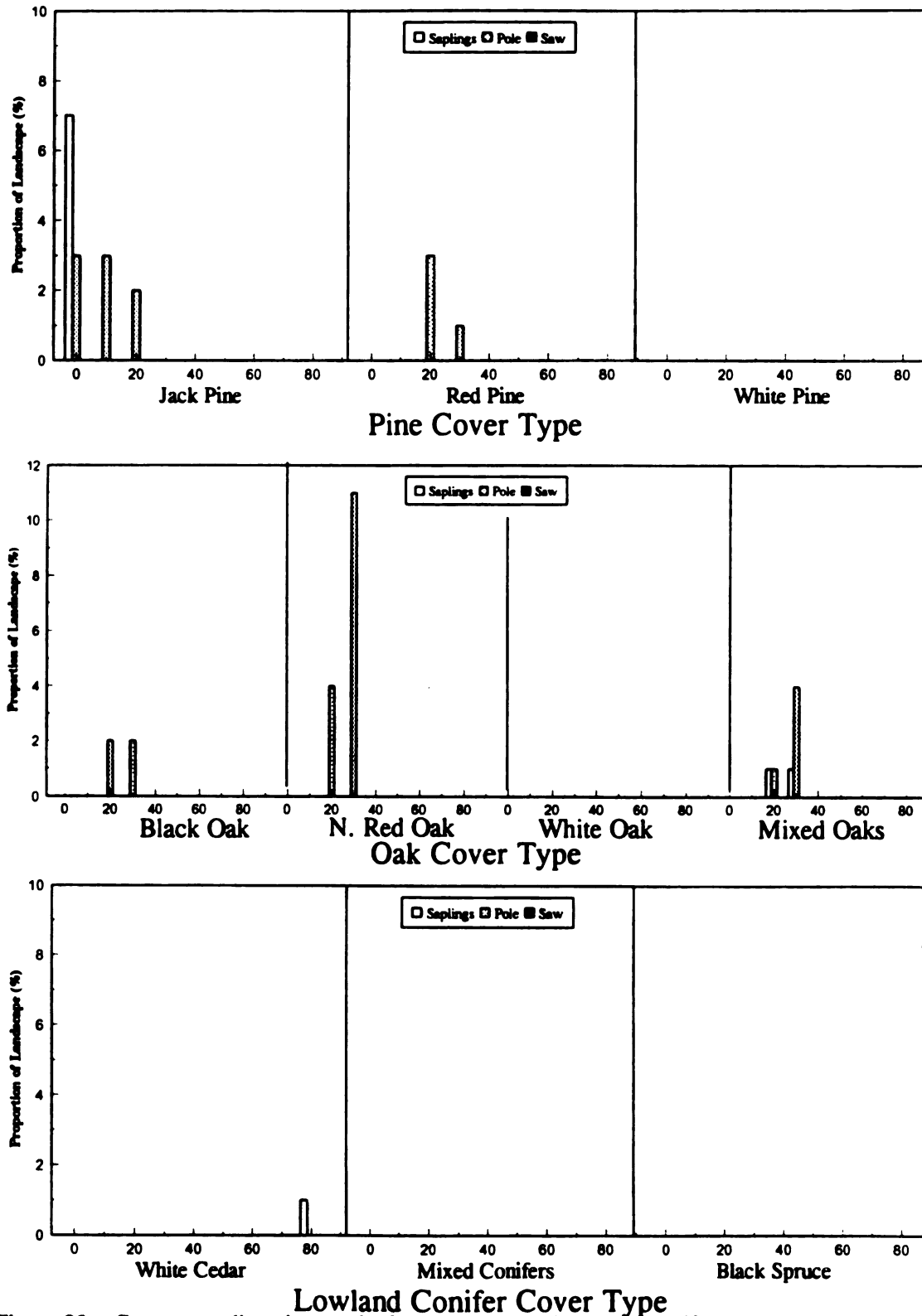


Figure 20. Cover type diversity graphs for pine, oak, and lowland conifer cover types. The x-axis is ecological land type (ELT).

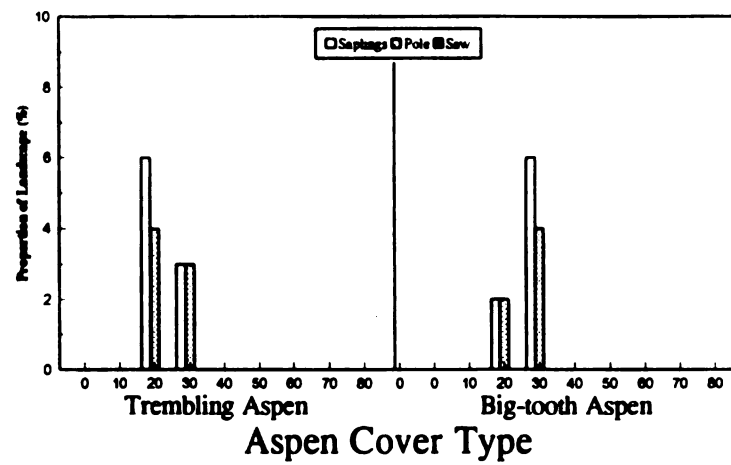
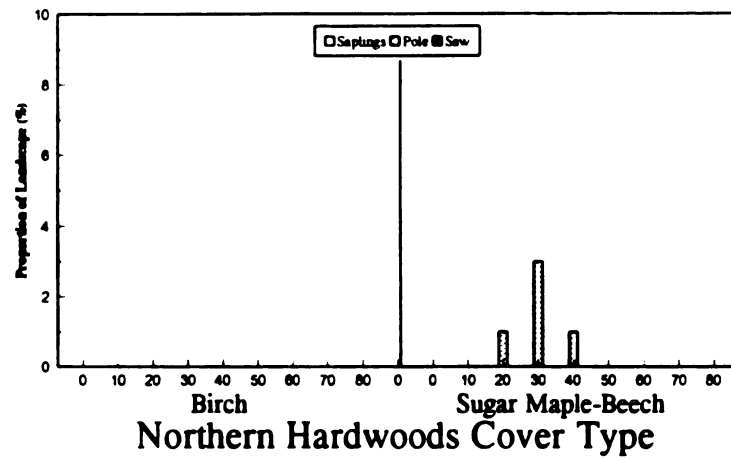
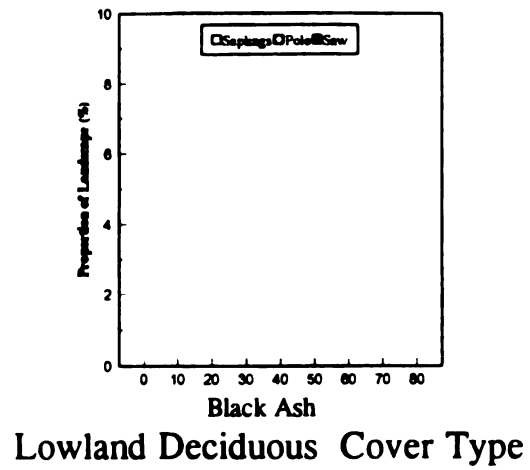


Figure 21. Cover type diversity graphs for lowland deciduous, northern hardwoods, and aspen cover types. The x-axis is ecological land type (ELT).

(Fig. 19); consistent with U.S. Forest Service Corporate Database classification (U.S. Forest Service 1988). Matrix cells contain cumulative hectares of vegetative type and corresponding age class per ELT (Fig. 19). Forest cover types representing the seral stage at which the successional process typically slows or is reset according to underlying ELT are denoted by shaded cells (Fig. 19).

The ecosystem diversity matrix was quickly constructed with the aid of GIS. The overlay capabilities of GIS were used to create the resource planning map, and subsequent polygon attribute database queries were conducted to determine hectares in each matrix category.

The ecosystem diversity matrix is a coarse filter approach to landscape assessments. The matrix is designed to delineate generalized, inclusive units; a necessary generalization to transform a continuum of ecological units and successional stages across landscapes into manageable entities. Data trends in the ecosystem diversity matrix may be masked by numerical complexity, thus a graphical tool was developed (Figs. 20-21). Cover type diversity graphs are derived from and linked to the matrix, providing a tabular and graphical representation of the landscape. The graphs allow for a species-specific evaluation of cover type-ecological unit representations in the landscape, however, numeric details and successional linkages are sacrificed (Figs. 20-21). To depict the same species-specific information in the ecosystem diversity matrix, the matrix would have increased in complexity by an additional 24 rows.

The most prevalent cover types on the study areas were aspen, oak, and pine, in decreasing order of abundance (Fig. 19). Lowland conifers and lowland deciduous cover types were least prevalent. The aspen cover types were equally distributed between trembling and big-tooth, consisting of sapling- (17% of the landscape) and pole-sized (13%)

on ELT's 20 and 30 (Fig. 21). The oak cover type was predominantly pole-sized on ELT's 20 and 30, dominated by northern red oak (15% of the landscape; Fig. 20). The pine cover type accounted for 19% of the landscape and was dominated by sapling- and pole-sized jack pine (Fig. 20). Jack pine inhabited the lower, less fertile ELT's, while red pine was found on ELT's 20 and 30.

White pine was historically a major forest component on the sandy soils typical of the Hoist Lakes area, however, white pine is poorly represented in the ecosystem diversity matrix and cover type diversity graphs (Figs. 19-21). White pine grows best on the sandy loams of ELT 30 (accounting for 45% of the landscape), and indeed, substantial white pine regeneration was noted, often under an oak canopy. In contrast, oak stands on ELT 20 were more likely to contain red pine in the understory. Both 20 and 30 ELT's have the potential to produce saw-sized stands of pine-oak mixes (Fig. 19), however, the pines require release from the existing overstory.

The majority (4,937 ha) of the landscape was in pole-sized timber, followed by sapling size (2,917) and saw-sized (98 ha). The abundance of pole-sized timber reflects: 1) second-growth timber from past, broad-scale silvicultural applications, and 2) the disruption of natural disturbance regimes. Most of the saw-sized timber consists of residual pine and oak, although some aspen also exists (Fig. 19).

The most prevalent ELT's were the 20's and 30's, consisting of mesic ice-contact sand hills and herb-poor moraines, respectively (Cleland et al. 1993). All of the prevalent vegetative types (pine, oak, northern hardwoods, and aspen) occur in the potential successional pathways for these ELT's (Cleland et al. 1993). Also prevalent were ELT's in the 0's and 10's, consisting of outwash plains (Cleland et al. 1993). These ELT's historically exhibited high fire frequency and thus, seldom attained saw-sized timber.

Outwash plains support pine and oak (Cleland et al. 1993).

The remaining ELT's, the 40's, 50's, 60's, 70's and 80's, accounted for 7% of the landscape and included mesic sites (ELT 40) characteristically supporting northern hardwoods and aspen-birch associations, dry ELT complexes (50's and 60's)(Cleland et al. 1993) supporting aspen and oak, and poorly drained sites (ELT's 70 and 80) that support lowland species (Cleland et al. 1993).

Generally, forest types appear to be appropriately distributed in regards to ELT's. Small amounts of aberrant acreages (e.g. 3 ha of lowland conifers on ELT 10) are probably due to GIS mapping error. The 43 ha of northern hardwoods on ELT 20 (Fig. 19) consists of a forest stand that may have been more appropriately classified as mixed oak (as suggested by ground reconnaissance).

Of concern among forest type age classes is the disproportionate amount of sapling:pole-sized:saw-sized timber. This analysis, however, has the potential to be deceiving. For example, acreages of mature oak that were selective harvested on high 30's ELT's frequently had an abundance of northern hardwoods saplings in the understory. Most of these stands retained their "oak" classification at a lower stocking density when in reality both the oak and sapling-sized northern hardwood classification could have been appropriate. Similarly, a flaw in grouping tree species (in the ecosystem diversity matrix) is evident in that saw-sized jack pine is naturally infrequent, however, the potential for saw-sized red and white pines is much higher (warranting the use of cover type diversity graphs). The key to successful use of the matrix is to track matrix changes over time (perhaps on 10-year cycles) and develop the proper successional linkages. For example, aspen on 30's ELT will succeed into northern red oak-red maple cover types (Cleland et al. 1993). Having a knowledge of the silvicultural applications during the 10-year period, and assuming that the vegetative GIS

layer is current, one can track these natural shifts between vegetation types. The ultimate goal of a coarse filter analysis using the ecosystem diversity matrix is to achieve adequate ecological representation across the landscape (Haufler 1994).

Although designed to be evaluated over larger spatial scales, the question for the Hoist Lakes area becomes "what is adequate ecological representation?" The ecosystem diversity matrix indicates that virtually none of the "late-successional" ecological units are represented (Fig. 19). The greatest potential (as indicated by ELT potentials and areas) to meet this deficiency lies in creating saw-sized oak or oak-red pine mixes on ELT 20; saw-sized oak, oak-white pine mixes, or white pine on ELTPG's in the low 30's; or saw-sized northern hardwoods on ELTPG's in the high 30's. Similarly, the absence of "late-successional" lowland conifers on ELT 80 may warrant consideration. On the other extreme, adequate ecological representation or openings requires an understanding of historic disturbance patterns. For example, large openings on ELT's 0 and 10 historically existed as a result of fires inherent to these sites. Presently, no "openings" exist on ELT 0, although large areas of jack pine seedlings do occur. Species associated with vast openings (e.g. upland sandpiper) need large openings to successfully reproduce, and adequate ecological representation may not be achieved without sandpiper populations. These types of issues need to be addressed on a larger and more diverse landscape than the Hoist Lakes area.

Integrating HSI Models: The Fine Filter

With adequate ecological representation of cover type-ELT combinations, one assumes that all wildlife species inherent to the landscape will be accounted for. However, to check the effectiveness of the coarse filter, fine filter approaches for selected species are warranted (Haufler 1994). Also, management and legal requirements may mandate species-specific assessments.

The forest potentials as indicated by ELT's help delineate "adequate ecological representation" across the landscape (Haufler 1994). For example, late successional stages for ELT's 30 and 40 (e.g. northern hardwoods) are under-represented, with the majority of the ELT's in secondary successional species (aspen and oak; Fig. 19). Thus, one would assume that wildlife species associated with this ecological unit are also under-represented.

Where appropriate, HSI models can be used as the fine filter analysis tool. Assume pileated woodpeckers were identified as indicators of "mature forests" or "late successional stages" (as identified by shaded cells in Fig. 19). The ecosystem diversity matrix suggests that the study areas provide inadequate ecological representation of this vegetation type (Figs. 19-21), and the woodpecker model can be used to verify this prediction in addition to geo-referencing the habitat linked to "mature forest" potentials (Fig. 22). If the management objectives were to increase "mature forest" representation across the landscape, one would expect pileated woodpecker habitat quantity and quality to increase over time. Thus, HSI maps can be used to document the progress towards meeting management objectives. Similarly, if management objectives are to provide for a diversity of ecological representation and wildlife species, several wildlife species can be identified and HSI maps created to evaluate management effectiveness. HSI maps for several species can be overlaid, and where high quality habitat overlaps for several species, these areas can be identified and managed accordingly. This type of approach also allows for consideration of patch and metapopulation dynamics.

Stand by stand assessments of most wildlife habitats are inadequate (see Chapter 2), and therefore, models must account for relationships between multiple stands across landscapes. Areas must be identified that provide both structural and spatial habitat. Recognizing that each point in the landscape may provide a unique habitat configuration, HSI

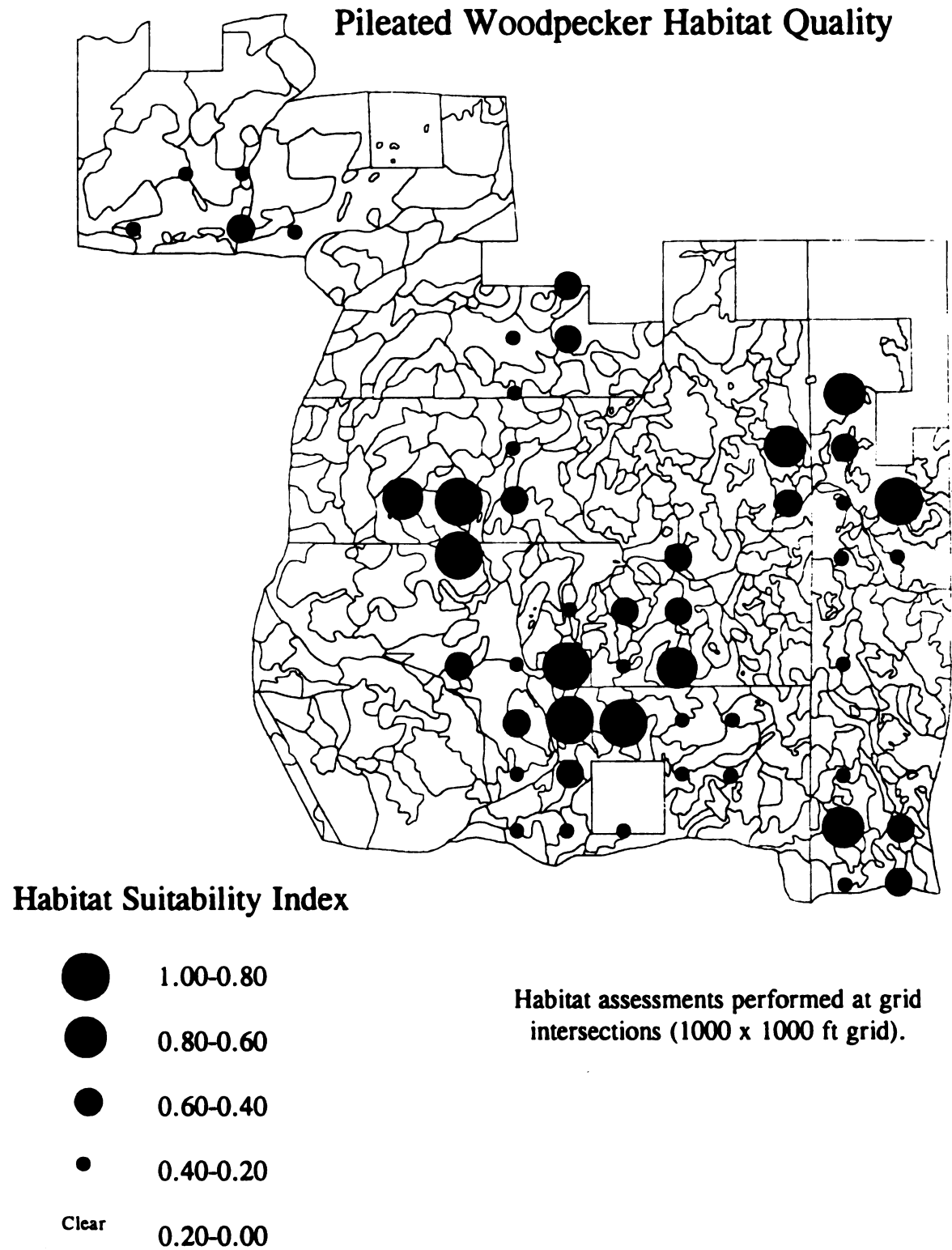


Figure 22. Hypothetical habitat suitability assessment for pileated woodpeckers across the "Hoist Lakes" landscape.

models should assess habitats from a representative sample of these points. One such approach is to overlay the landscape with a grid (Fig. 22), the dimensions of which would be dictated by the habitat requisites of the species. For example, the grid size for pileated woodpeckers was 1000x1000m (Fig. 22), roughly corresponding to the scale of model applicability (see Chapter 2). A home-range-sized circular plot was subsequently overlaid on each grid intersection, and HSI values were computed (Fig. 22). Subsequently, maps delineating regions of similar HSI values can be created and integrated into management decisions.

The question still remains as to the fulfillment of wildlife management goals across the landscape. The HSI maps are useful for evaluating the spatial distribution and quantity of habitats, but there is still a need to efficiently summarize habitat quality in terms of management goals. Assume HSI values were calculated at 175 points across a landscape (as was done to create the pileated woodpecker HSI map (Fig. 22)). A histogram depicting HSI value (abscissa) and the number of points (ordinate) can be used to evaluate landscape potentials (Fig. 23). Minimum levels of acceptable habitat quantity and quality can be identified and related to existing habitat conditions (Fig. 23). Acceptable levels of habitat may be driven by minimum viable population requirements (e.g. Fig. 23), metapopulation theory (i.e. providing large amounts of low quality corridors between isolated, high quality patches), and/or arbitrary management objectives (e.g. an even representation of all HSI classes across the landscape (Fig. 23)). For example, the habitat evaluation depicted in figure 23 indicates that both minimum viable population requirements and management objective are not being met, however, there is an abundance (above the management objective) of low quality habitat that can be enhanced. Although the histogram is a useful display tool, it cannot be separated from the spatial HSI map and both should be used

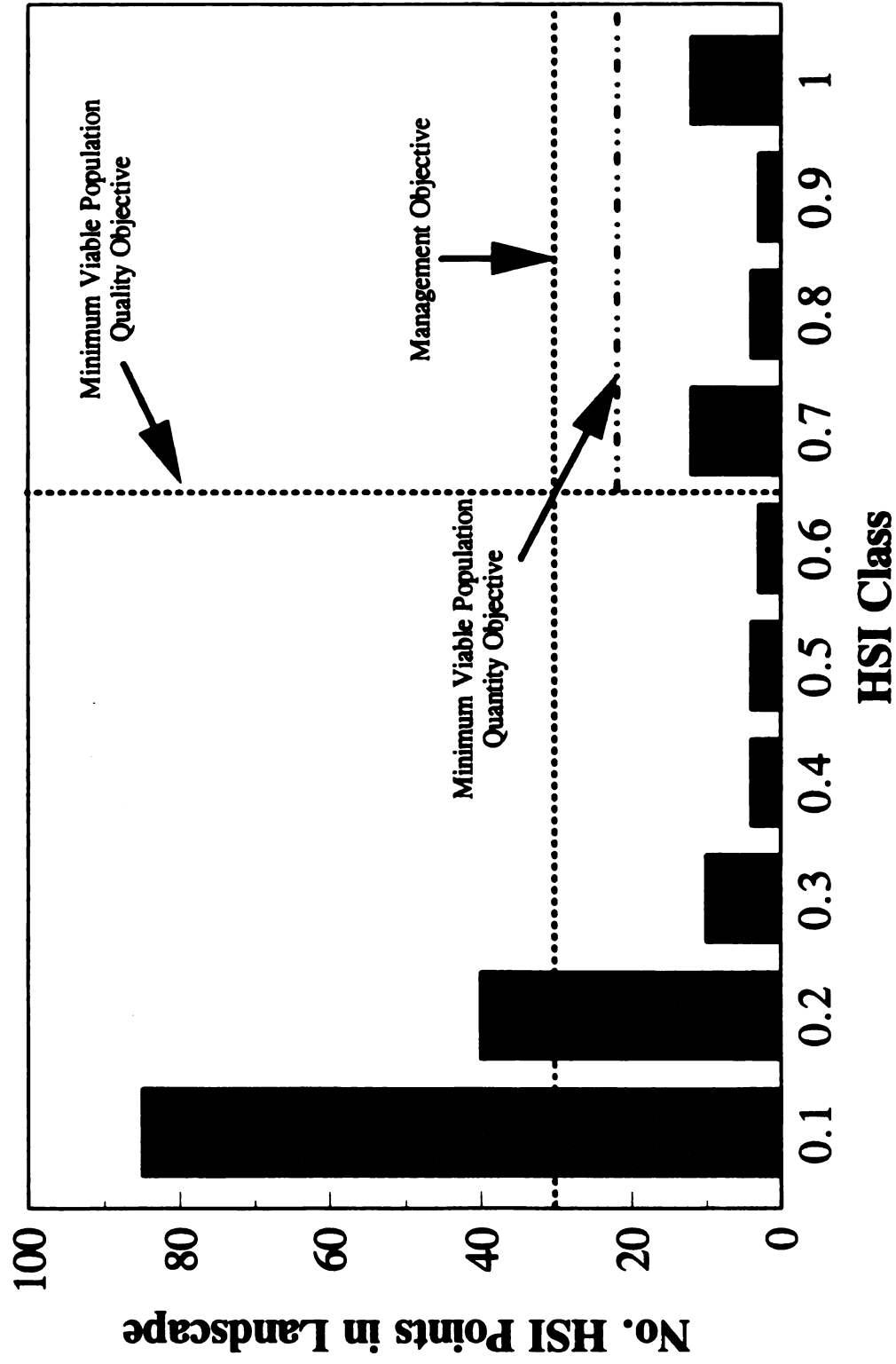


Figure 23. Habitat suitability index (HSI) histogram depicting management objectives for a hypothetical landscape.

together. Considerably more information is needed to link habitat planning processes with the needs for minimum viable populations, and in turn, adequate ecological representation for the coarse filter approach.

SUMMARY AND CONCLUSIONS

The ecosystem diversity matrix and graphs provide static views of landscapes, however, the dynamic characteristics inherent to all lands is also a critical component of successful ecosystem management. Preserving biodiversity in temperate regions requires the maintenance of all successional stages (Franklin 1988). With biodiversity at the forefront of contemporary management issues, the dynamic nature of the landscape must be accounted for. Although succession is a continuous process rather than a discrete event (Flather and Hoekstra 1985), techniques to incorporate successional change into forest planning are required. The ecosystem diversity matrix, as a coarse filter, provides a basis for incorporating the natural successional pathways of land into resource management plans. With the addition of successional models, both ecosystem diversity matrices and HSI models can provide quantitative pictures of landscape dynamics over time.

APPENDICES

APPENDIX A

Description of Ecological Classification System Components

The Ecological Classification System (ECS) of the Huron-Manistee National Forests is a hierarchical classification (Fig. 24) developed in northern lower Michigan (Cleland et al. 1993). Appendix A briefly summarizes the ecological land types (ELT's), ecological land type phase groupings (ELTPG's), and ecological land type phases (ELTP's) as described by Cleland et al. (1993). Only ecological units used for analyses in Chapter 1 are described.

ELT: Outwash Plains

Characterized as black oak-white oak-*Vaccinium* plant association on excessively well drained sands of outwash plains. Coarse textures and near siliceous minerology of these outwash deposits cause few soil morphological differences to exist within the ELT.

ELTPG: Outwash I

Characterized as pin oak (*Quercus palustris*)-white oak-*Deschampsia* plant association on excessively well drained sands of glacial outwash plains. The driest and most nutrient impoverished ELTPG. Differs from other outwash ELTPG's by having a more open canopy, no red maple in the understory and low ground layer coverage.

ELTP: 01 - Same as ELTPG Outwash I.

ELTPG: Outwash II

Characterized as black oak-white oak-*Vaccinium* plant association on excessively well drained sands of outwash plains. May include thin coarse

Ecological Classification System Hierarchy

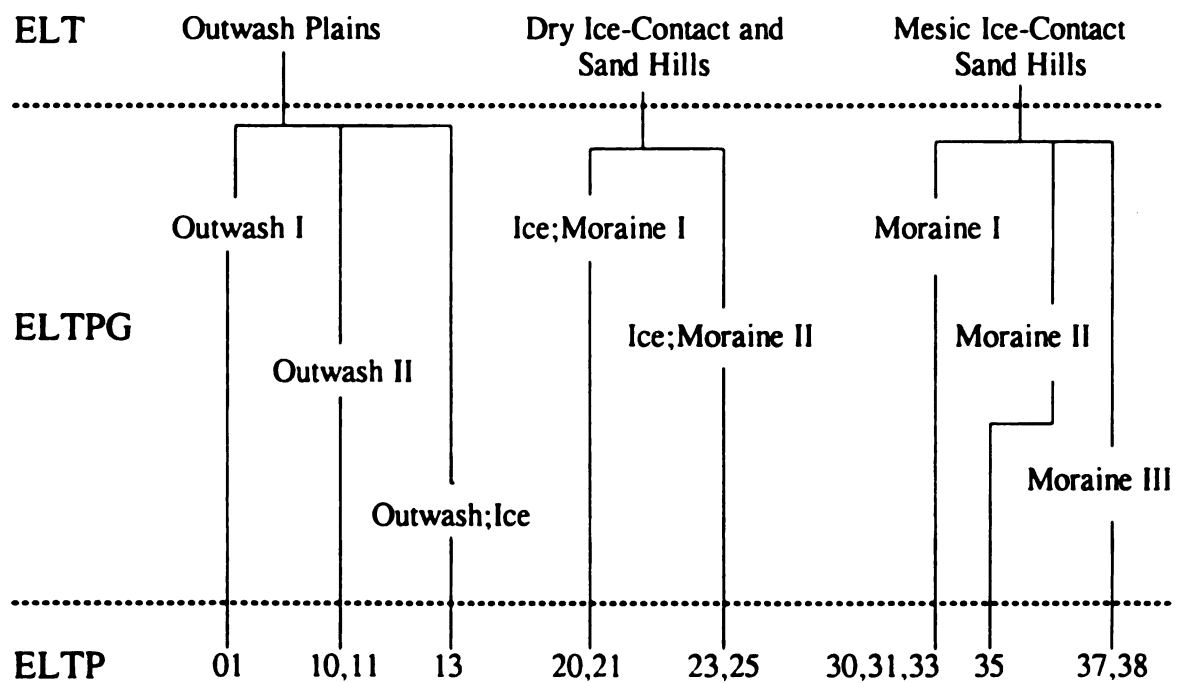


Figure 24. Ecological classification system hierarchy. Based on the works of Cleland et al. (1993).

loamy bands throughout the soil strata. Typically a moderately closed canopy with bracken fern (*Pteridium aquilinum*) and red maple in the understory.

ELTP: 10 - Excessively well drained sands of outwash plains;sandy throughout the soil profile.

ELTP: 11 - Excessively well drained sands with thin coarse loamy bands beneath sandy sola.

ELTPG: Outwash; Ice

Characterized as black oak-white oak-*Vaccinium* plant association on outwash and ice contact sands and gravels. Gravel and cobble common throughout soil profile; finer textured soils found below 150 cm.

ELTP: 13 - Same as ELTPG Outwash; Ice.

ELT: Dry Ice-Contact and Sand Hills

Mixed oak-red maple-*Trientalis* plant association on well to excessively well drained sands on overwashed moraines, kame terraces, and glacial spillways. Includes red maple in the overstory with a potential for aspen and paper birch (*Betula papyrifera*) seres.

ELTPG: Ice;Moraine I

Characterized as mixed oak-red maple-*Trientalis* plant association on well to excessively well drained sands or banded sands on ice-contact and overwashed topography. Typically found in rolling topography with red maple and a higher predominance of red oak in the overstory; also contains a spodic soil horizon.

ELTP: 20 - Lack of sandy loam textural bands in the substratum.

ELTP: 21 - Sandy loam textural bands in the substratum.

ELTPG: Ice;Moraine II (Note: Description based on limited data).

Characterized as mixed oak-red maple-*Trientalis* plant association on well to excessively well drained sands on overwashed moraines, kame terraces, and glacial spillways. Medium textured sands generally occur down to 50 cm; and soils in the lower stratum have coarse loamy textural discontinuities.

ELTP: 23 - Unsorted sands and gravels. Some banding in the solum.

ELTP: 25 - Unsorted sands and gravels. Presence of a fine textured substrata.

ELT: Mesic Ice-Contact Sand Hills

Characterized as red oak-red maple-*Viburnum* plant association on moisture-enriched, well drained sands on moraines and ice-contact topography. Subsections in this ELT are distinguished primarily by differences in geologic parent material, soil morphology and classification, and temperature gradient effects on forest succession. Soils are predominantly well to moderately well drained sands with higher moisture holding and cation exchange capabilities than the outwash ELT's.

ELTPG: Moraine I (Note: Description based on limited data).

Red oak-red maple-*Viburnum* plant association on well-drained sandy moraines and ice contact topography. Typically occurs within glacial lakebeds and adjacent to existing bodies of water. Underlain by perennial water tables with sandy soil textures to depths generally greater than 1.5 m.

ELTP: 30 - Medium sands occur in substratum; coarse sand and gravels are generally absent from substratum; E horizon 2-7 cm thick are common.

ELTP: 31 - Generally has banding below 150 cm; gravelly banded and highly stratified sand substratums of ELTP 230 are included in this unit.

ELTP: 33 - The subirrigated unit of the Moraine I system with perennial water tables lying 2-4.5 m deep.

ELTPG: Moraine II

Characterized as red oak-red maple-*Viburnum* plant association on well-drained sands with fine loamy substrata on moraines and ice-contact topography.

Sandy soils to depths generally > 1.5 m with a substrata of sandy clay loam or finer textures within depths of 1.5-4.5 m; usually more than 15 cm thick. This ELTPG may be more nutrient rich than other ELTPG's in this ELT because of higher bases available in loamy substrata, and possibly higher available phosphorous due to higher soil pH.

ELTP: 35 - Same as ELTPG Moraine II description.

ELTPG: Moraine III

Characterized as red oak-red maple-*Desmodium* plant association on well to moderately well drained sandy loams over loamy substrata on ground moraines and fine-textured glacial lakebeds. High water availability; physiography is level to gently undulating. This ELTPG occurs within or adjacent to

landforms characterized by frequent presettlement fires. May be viewed as a fire climax unit, which will succeed to northern hardwoods without fire.

ELTP: 37 - Some pedons have loamy sand caps.

ELTP: 38 - Sandy or loamy cap is generally absent.

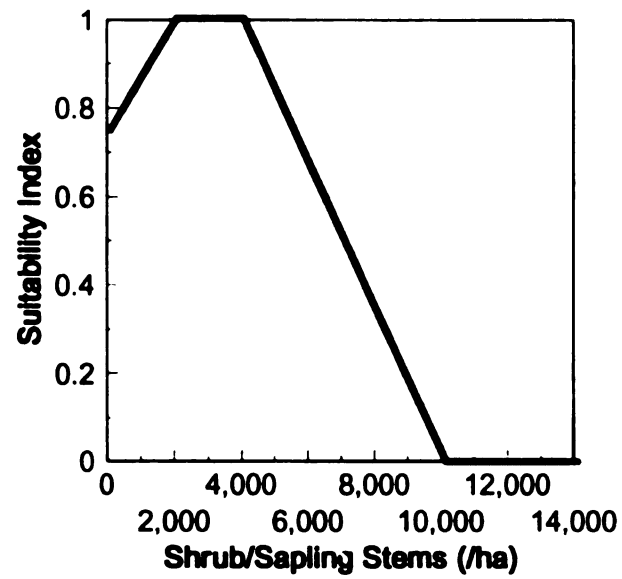
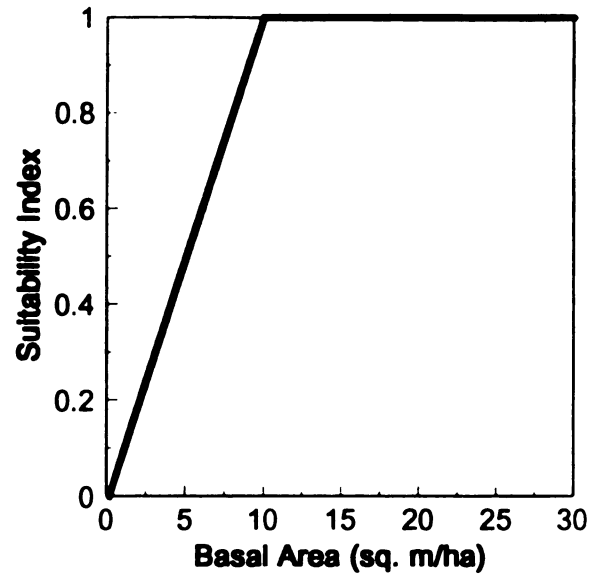
APPENDIX B

Habitat Suitability Index Model for the Ovenbird

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The habitat suitability model for the ovenbird was based on the research conducted by Capen et al. (1986). They described suitable ovenbird habitat as a mature deciduous forest with ample regeneration in the understory.

Using univariate comparisons between means, Capen et al. (1986) identified 5 habitat variables (out of 20) that differed between plots used and unused by ovenbirds. Discriminant function analysis (DFA) was used to identify 2 habitat variables that were estimated to be important in determining ovenbird habitat suitability (Fig. 25). The suitability graphs are not empirically based, but rather, the graphs are designed to provide a high suitability rating to stands that exhibit the structural and compositional characteristics of mature deciduous forest with medium to dense regeneration in the understory. The overall habitat suitability index is the geometric mean of the suitability indices.



$$HSI = (\text{Basal Area Suitability Index} * \text{Shrub/Sapling Suitability Index})^{0.5}$$

Figure 25. Habitat suitability index graphs for the ovenbird.

APPENDIX C

A Habitat Suitability Index (HSI) Model for the Black-throated Green Warbler in the Huron-Manistee National Forest, Michigan.

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The black-throated green warbler (*Dendroica virens*)(BTGW) is a common summer resident of northern coniferous and mixed coniferous-deciduous forests of eastern North America.

Although occurring in a variety of vegetation types, the BTGW is most closely associated with mature spruce (*Picea* spp.), hemlock (*Tsuga canadensis*), and pine (*Pinus* spp.) stands (Pitelka 1940, Bent 1963, DeGraaf and Rudis 1983). During the breeding season the BTGW is one of the more frequent wood warblers in the conifer regions of northern lower Michigan, although it is not common (Pitelka 1940). Despite being used as an indicator of mature conifer forests (U.S. Forest Service 1989), the BTGW is less specific in habitat preferences than such characteristic coniferous species as the pine warbler (*D. pinus*), and will occur in areas having only a few scattered conifers for nesting (Pitelka 1940).

Habitat Use Information

Food

The BTGW is primarily insectivorous, although the species will eat a variety of plant matter (DeGraaf and Rudis 1983). Feeding habits consist primarily of gleaning insects from the limbs and foliage of conifers, usually between 3 and 15 m (10-50 ft) above the ground. BTGW's will also forage in the upper canopies of tall shrubs or midstory trees (MacArthur 1958, Morse 1967, Rabenold 1978, Sabo 1980, Mauer and Whitmore 1981). Feeding

behavior consists of hopping, peering, or hovering followed by gleaning; hawking is much less common (DeGraaf and Rudis 1983).

Major foods of BTGW's include leaf eating caterpillars, beetles, flies, gnats, plant lice, spiders, moths, and ants (Reading and Hayes 1933, DeGraaf and Rudis 1983). Seeds and fruits are taken to a lesser extent ; berries of poison ivy (*Toxicodendron radicans*) and juniper (*Juniperus* spp.) are particularly preferred (Barrows 1912, DeGraaf and Rudis 1983).

BTGW's forage in a variety of vegetation types, including swamps, orchards, deciduous trees, and shrubs (Bent 1963). Morse (1973) felt BTGW's were opportunistic foragers and not food limited. BTGW's can be selective in their foraging strategies, however, Morse (1976a) noted preferences for foraging red spruce (*P. rubens*) in Maine, although food was more abundant in white spruce (*P. glauca*).

Water

No information relating to water requirements was found in the literature. Due to the nature of the BTGW's insectivorous diet, it is unlikely that the species requires free water.

Cover

BTGW's utilize a variety of vegetation types; most commonly, however, some type of conifer presence is included in habitats used (Pitelka 1940, Bent 1963, DeGraaf and Rudis 1983). Collins (1983) identified 5 plant communities commonly used by BTGW's across their geographic range. These communities were 1) pine forest, 2) spruce-arbor vitae (*Thuja occidentalis*), 3) mixed spruce-fir (*Abies* spp.)-deciduous, 4) balsam fir (*A. balsamifera*), and 5) birch (*Betula* spp.)-maple (*Acer* spp.)-beech (*Fagus grandifolia*). DeGraaf and Rudis (1983) felt BTGW's to be primarily associated with hemlock in the northeastern states.

In Michigan, BTGW's have been reported to occur primarily in upland forests of pure coniferous (usually red (*P. resinosa*) or white (*P. strobus*) pine) or mixed coniferous-deciduous growth; spruce bogs are also commonly utilized, while pure deciduous forest is used infrequently (Pitelka 1940). BTGW's also inhabit jack pine (*P. banksiana*) plains in northern lower Michigan (Barrows 1912).

Reproduction

BTGW nest locations are almost exclusively in conifers (Pitelka 1940). However, nests in maples, birches, alders (*Alnus* spp.), and grape vine (*Vitis* spp.) have also been reported (Bent 1963, Griscom and Sprunt 1979). Barrows (1912) listed common nest sites in Michigan to be in balsam fir, spruces, hemlock, and tamarack (*Larix laricina*). Pitelka (1940) described nesting in red pine in northern lower Michigan.

Nest locations vary in the tree; nests have been recorded from 5 cm to 22 m (2 up to 7 ft) above the ground (Griscom and Sprunt 1979). Pitelka (1940) described the nest site characteristics of 2 BTGW nests found in red pine in Michigan. The first nest was 7 m (23 ft) above the ground, and 1.68 m (6 ft) horizontally from the trunk; the second nest was 3.66 m (12 ft) high and 1.52 m (5 ft) from the trunk.

Interspersion

Home range sizes for BTGW's averaged 0.64 ha (1.6 ac) in hemlock-beech vegetation types in New York; home ranges varied from 0.2-1.0 ha (0.6-2.5 ac) in size (Kendeigh 1945). Morse (1976a) found BTGW home ranges in spruce forests in Maine to range from 0.2-0.5 ha (0.6-1.2 ac) in size. Home ranges in less preferred northern hardwoods vegetation types averaged 1.2 ha (3 ac) in New Hampshire (Sabo 1980). BTGW territories are vigorously defended during the breeding season (Pitelka 1940, Reading and Hayes 1940, Kendeigh 1945,

Bent 1963). Male BTGW's seldom wander > 100 m (328 ft) outside their established territories (Kendeigh 1945). Wandering is much less common in BTGW's than in other wood warbler species.

No information was available on the minimum contiguous habitat necessary to support viable populations of BTGW's. As BTGW's largely confine their activities and satisfy their life requisites within their relatively small home ranges (Kendeigh 1945), it is likely that any forested tract of acceptable habitat larger than the typical BTGW home range can support this species.

Special Considerations

Quantitative data regarding BTGW habitat requirements are extremely limited in the literature. Collins (1983) hypothesized that BTGW's take advantage of what local habitat structure is available. As a consequence of this, regional analyses of BTGW habitat (even within a single plant community type) may be of limited value for habitat management outside of that particular region.

Habitat Suitability Index Model

The following model has been developed to objectively quantify habitat quality for the black-throated green warbler in the Huron-Manistee National Forest (HMNF), Michigan. This model assumes that the quality of BTGW habitat on the HMNF can be evaluated in terms of 3 habitat variables. These 3 variables are:

- V1: Percent conifer canopy cover.
- V2: Mean height of the overstory.
- V3: Percent of the overstory with a deciduous or coniferous midstory ≥ 5 m in height.

The unit of evaluation for this model is a forest stand.

Model Justification

Optimal cover and reproductive habitat for the BTGW is provided by dense stands of mature conifers with a multi-layered deciduous or coniferous midstory from 30-70% the height of the overstory trees. Species of conifer does not appear to be an important variable, although highest BTGW densities are associated with spruce or spruce-fir communities (Morse 1967, 1976a, 1976b; Hamel et al. 1982). BTGW's will occupy stands of deciduous species as long as a few conifers are present for nesting (Pitelka 1940, Sabo 1980, Holmes and Robinson 1981, Robinson and Holmes 1982, Hamel et al. 1982). BTGW's appear to be totally dependent upon conifers for nesting (Pitelka 1940).

Highest breeding densities and smallest home range sizes for BTGW's are associated with dense pure conifer stands (Tables 20-21). Breeding densities of 40-80 pairs/100 acres have been documented in these vegetation types (Morse 1976a, Hamel et al. 1982). Mixed conifer-hardwood stands or open conifer stands are also suitable habitat, with documented breeding densities of 10-40 pairs/100 acres (Kendeigh 1945, Holmes and Robinson 1981, Robinson and Holmes 1982). Almost pure stands with only occasionally scattered conifers typically support < 10 pairs/100 acres (Sabo 1980, Hamel et al. 1982). Exact quantification of vegetative variables associated with most of these studies is lacking, however. Collins et al. (1982) found optimal BTGW habitat by principal components analysis to consist of stands with a mean canopy closure of 84.3% and an average overstory height of 20.2 m (66 ft). Collins (1983) reviewed descriptions of BTGW habitat from a variety of geographical regions. Total canopy closure ranged from 84.0% to 94.5% for areas occupied by BTGW's; 23-76% of this was conifer (Collins 1983). This model assumes that BTGW habitat will be optimal with a conifer canopy coverage of 75-100% (Fig. 26). Habitat quality will deteriorate moderately as conifer canopy cover drops from 75% to 25%. Below 25% conifer canopy coverage, habitat

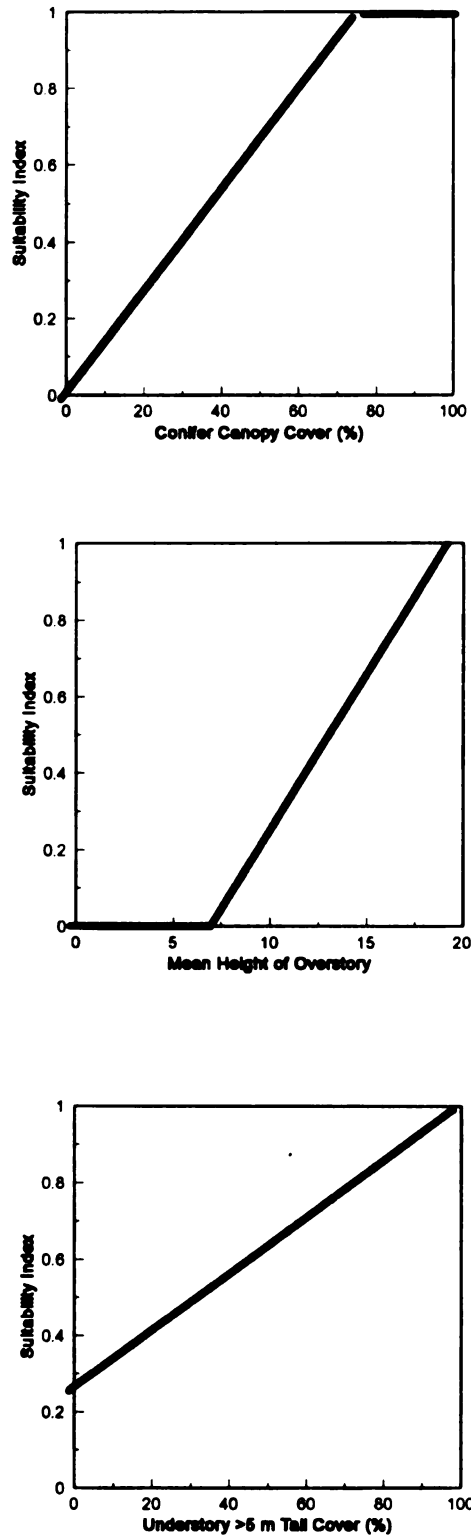


Figure 26. Habitat suitability variables for the black-throated green warbler.

Table 20. Density-habitat relationships for black-throated green warblers in various vegetation types.

Density (Pairs/100 ac)	Vegetation Type	Source
83	Mature white spruce	Morse 1976
71	Mature red spruce	Morse 1976
61	Mature white spruce	Morse 1976
53	Mature red spruce	Morse 1976
51	Mature white spruce	Morse 1976
44	Sawtimber spruce-fir	Hamel et al. 1982
43	Mature white spruce	Morse 1976
36	Mature oak-maple	Stewart and Robbins 1958
33	Mature n. hardwood scattered spruce	Robinson and Holmes 1982
32	Mature n. hardwood with spruce and hemlock	Holmes and Robinson 1981
26	Mature virgin spruce	Rabenold 1978
12	Mature spruce-fir	Rabenold 1978
9-17	Mature hemlock	Kendeigh 1945
9	Mature n. hardwood	Stewart and Robbins 1958
8	N. hardwood	Sabo 1980
7	Sawtimber oak-hickory	Hamel et al. 1982

Table 21. Territory size-vegetation relationships for the black-throated green warbler.

Territory Size (ac)	Vegetation Type	Source
0.2-0.5	Mature spruce	Morse 1976
0.64	Mature hemlock	Kendeigh 1945
1.2	Mature n. hardwoods	Sabo 1980

quality quickly deteriorates. If no conifers are present, habitat suitability is assumed to be 0.00, as BTGW's are dependent upon the presence of conifers for nesting (Pitelka 1940). BTGW'S prefer mature stands of large trees (Pitelka 1940, Bent 1963, Morse 1967, 1976a, 1976b). Collins (1983) reported canopy heights of areas occupied by BTGW's to range from 14 m (46 ft) to 20 m (66 ft) in height. Collins et al. (1982) reported optimal habitat in Minnesota to have a mean canopy height of 20.2 m (66 ft). Robinson and Holmes (1982) found BTGW habitat in the Hubbard Brook Experimental Forest in New Hampshire to be characterized by canopy heights of 25-27 m (82-88 ft). Kendeigh (1945) similarly noted preferences for large mature trees; specifically, 61 m (200 ft) eastern Hemlocks. Other researchers have also noted preferences for large mature trees without quantifying exact canopy heights (Morse 1967, 1976a, 1976b; Rabenold 1978, Sabo 1980, Holmes and Robinson 1981, Hamel et al. 1982).

This model assumes that optimal BTGW habitat consists of forested stands of large mature trees (Fig. 26). Optimal canopy height occurs in stands with a mean canopy height ≥ 20 m (66 ft). Suitability declines as mean canopy height drops from 20 to 7.5 m (66 to 25 ft). Stands with a mean canopy height ≤ 7.5 m (25 ft) are assumed to be unsuitable BTGW habitat.

BTGW's typically forage in an area between 30-70% of the mean overstory height (MacArthur 1958, Morse 1967, Rabenold 1978, Sabo 1980, Mauer and Whitmore 1981). BTGW's glean insects from the inner, middle, and outer branches of all overstory trees in this height strata, as well as from any tall shrubs or midstory that is present in this range. This model assumes that the maximization of foliar biomass in this height strata will optimize habitat by providing maximum foraging opportunities (Fig. 26). As the percentage of the stand with a tall shrub layer or midstory greater than 5 m (16 ft) in height decreases, habitat

suitability for BTGW's will decline. If a stand has no tall shrub or midstory in this strata, the suitability index will be 0.20. This is because BTGW's will still forage in the overstory in the preferred height range, as well as occasionally foraging in the upper canopy (> 70% of the mean canopy height) and low shrubs (< 30% mean overstory height)(MacArthur 1958, Morse 1967, Rabenold 1978, Mauer and Whitmore 1981).

Model Application/HSI Determination

The habitat quality of a forest stand for BTGW's can be evaluated by determining the individual suitability index values for the 3 habitat variables described above (Fig. 26). the overall suitability of a stand for BTGW's is determined by relating the individual suitability values for the 3 model variables utilizing the following equation:

$$HSI = (V^1 * V^2 * V^3)^{0.33}$$

This model will thus generate an overall HSI score between 0.00 and 1.00, with 0.00 representing unsuitable habitat and 1.00 representing optimal habitat. Future verification of this model can be accomplished by comparing densities of BTGW's with HSI scores from various forest stands of differing BTGW habitat quality. A positive correlation should exist between BTGW density and HSI score for this model to be considered valid.

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APPENDIX D

A White-tailed Deer HSI for the Upper Great Lakes Region

****DRAFT****

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A habitat suitability index (HSI) model was developed for white-tailed deer in the Upper Great Lakes Region to assist land managers in assessing and providing for deer habitat in land management practices. This model evaluates deer habitat via 4 distinct sub-models, each dealing with a critical habitat component: Fall and Winter Food, Spring Food, Security Cover, and Thermal Cover. Habitat suitability for each critical component is determined by mathematical equations relating variables that contribute to the habitat component. An overall HSI score quantifying the quality of deer habitat in the evaluation area is then determined as the lowest suitability score of the 4 component sub-models. Quantifying habitat quality for key species such as deer greatly aids land managers in accounting for the habitat needs of wildlife in land management decisions, and assessing the potential impacts of land management decisions on wildlife.

The white-tailed deer are the most popular big game animal in North America (Halls 1978). Highly prized as venison, as a trophy, and for viewing, it is one of the most valuable and important wildlife species in the Upper Great Lakes Region (UGLR). Consideration of the habitat needs of white-tailed deer is therefore extremely important in land management planning.

Although well quantified, the habitat needs of deer have not been extensively modelled. Deer inhabit a broad range of climatic and vegetative conditions across their range

(Halls 1978). Due to this diversity, modelling of deer habitat must be localized. Models that are too geographically broad will be so inclusive as to be trivial. Similarly, models too narrow in focus will lack applicability to any area outside the narrow boundaries. This model attempts to evaluate habitat quality for deer in the UGLR in general, and the Huron-Manistee National Forest, Michigan, in particular. Modelling at this geographic level maximizes model utility while maintaining model complexity within reasonable constraints.

The habitat suitability index model presented here involves the assessment of deer habitat via 4 distinct and independent sub-models, each of which deals with a critical element of deer habitat. These 4 sub-models are:

- (1) FALL AND WINTER FOODS
- (2) SPRING FOODS
- (3) SECURITY COVER
- (4) THERMAL COVER

Deer require all 4 of these habitat components for existence in most of the UGLR.

Evaluation of each critical element of deer habitat via a unique sub-model assures that each life requisite will be given maximum consideration in the evaluation of deer habitat suitability in the UGLR.

GENERAL INFORMATION

White-tailed deer foods follow similar seasonal patterns throughout the Upper Great Lakes Region, despite local differences in vegetation (Rogers et al. 1981). Browse constitutes the primary food source, mainly because it is the only source available year round (Blouch 1984). Other categories of common deer foods include conifer needles, evergreen forbs, deciduous leaves, non-evergreen forbs, grasses, fruit, and fungi (Rogers et al. 1981). Grain residues left in fields after harvest are heavily utilized by deer from September to April

(Gladfelter 1984).

Spring/Summer Foods

Spring foods of deer consist principally of herbaceous vegetation (Healy 1971, Rogers et al. 1981). The most common foods include grasses, sedges, basal rosettes of perennial forbs, and emergent bracken fern (McNeill 1971). Grassy openings are especially important at this time of year (McCafferty and Creed 1969), as are old fields, roadsides, and powerline right-of-ways (Rogers et al. 1981). As spring progresses, new green grasses, emerging forbs, and new leaves of trees and shrubs eventually make up 90% of a deer's diet (Pierce 1975, Rogers et al. 1981). These foods are both nutritious and easily digestible, providing the high quality diet necessary for both survival and reproduction in the face of increased spring metabolic rates (Verme 1963, 1969; Verme and Ullrey 1972).

During late spring-early summer, deer feed heavily on aquatic emergents (Rogers et al. 1981). These plants are nutritious and may provide a source of sodium and other important nutrients for pregnant and lactating does (Jordan et al. 1974).

Summer deer foods consist of leaves of non-evergreen terrestrial plants, mushrooms, and fruits (Kohn and Mooty 1971, McCafferty et al. 1974). Succulent new leaves are the principal component of deer diets during this period (Blouch 1984). Leaves of aspen seedlings < 1 year old are especially preferred.

Fall and Winter Foods

As summer progresses into fall, use of grasses increases while leaf and forb use declines (Blouch 1984). During good mast years, acorns, beechnuts, and other hard mast are highly preferred fall/winter foods. Deer shift their diets to woody browse with the first frosts; the shift to browse is usually complete with the first snows (Blouch 1984). In the George Reserve of Michigan, 7cm of snow changed the diet of deer from 63% to 0%

herbaceous (Coblentz 1970). Where available, agricultural crops, especially corn and soybeans, can also make up a major portion of a deer's fall and winter diet (Nixon et al. 1970).

Woody browse forms the bulk of deer winter diets (Rogers et al. 1981). Northern white-cedar, red maple, eastern hemlock, American mountain-ash, and alternate-leaved dogwood are all highly preferred browse species (Blouch 1984). Eastern white pine, yellow birch, mountain maple, serviceberry, and jack pine are slightly less preferred, followed by aspen, northern red oak, beaked hazelnut, paper birch, balsam fir, and red pine. Last resort foods include speckled alder, black and white spruce, and tamarack. Although cedar is highly preferred, conifer needles are typically a poorer diet than most woody browse (Rogers et al. 1981). Conifer needles comprise only a small proportion of the diets of healthy deer; needles can comprise up to 50-60% of stomach contents of starved deer (Aldous and Smith 1938, Dahlberg and Guettinger 1956).

A prolonged diet of woody browse causes malnutrition and starvation in deer (Mautz 1978). Browse is typically poor nutritionally, and deer will lose weight on a steady browse diet even if browse is available ad libitum (Ullrey et al. 1964, 1967, 1968; Verme and Ullrey 1972, Grigal et al. 1979). As a result, at winters end deer fat reserves are usually depleted and deer are nutritionally stressed (Verme 1969, Mautz 1978).

COVER

Security Cover

Cover can be defined as any structural feature of the environment that is used as protection from the environment (thermal cover) or from predators (security cover) (Boyd and Cooperrider 1986). Security cover is used for protection from predators and/or humans.

White-tailed deer are hidiers; they rely on suitable vegetative cover to conceal themselves from

predators, but they can also run effectively through dense vegetation to escape predators (Boyd and Cooperrider 1986). Thus, security cover for deer consists of dense vegetation in which the animal can conceal itself and flee, if necessary.

Thomas et al. (1979) noted that optimal habitat for deer requires hiding cover, perhaps because it gives the animals a sense of security. Thomas et al. (1979) defined security cover (hiding cover) as vegetation capable of hiding 90% of a standing adult deer at a distance of $\leq 61\text{m}$. Most use of security cover by deer tends to occur within 183m of the edge between cover and forage areas (Reynolds 1966, Harper 1969, Thomas et al. 1979).

Thermal Cover

In the UGLR, snow depth, low temperatures, and lack of protection combine to limit the northern range of deer (Halls 1978, Blouch 1984). Snow limits deer movements and covers food; cold temperatures and wind combine to drive deer energy reserves down. In response to this environmental stress, deer tend to concentrate (yard) in heavy coniferous cover, where snow depth, wind, and radiant heat loss are minimized (Blouch 1984).

White-tailed deer movements begin to be restricted by snow when it reaches depths of 36-43cm (Kelsall 1969). Deer may travel considerable distances to reach suitable thermal cover. Distances traveled in the UGLR range up to 10-48+km (Rongstad and Tester 1969, Dahlberg and Guettinger 1956, Verme 1973).

Many factors determine what constitutes adequate deer thermal cover. Weber et al. (1983) found 4 variables to accurately predict deer use of potential thermal cover areas in New Hampshire: (1) area site index, (2) stand basal area, (3) softwood crown closure, and (4) stand size. These 4 variables were able to predict site utilization with a precision of 95%. Additionally, species of conifer has been shown to be an important factor in deer yard quality in the UGLR (Blouch 1984).

In New Hampshire, site index showed the greatest variation between conifer stands utilized as deer yards and those not used (Weber et al. 1983). Site index is important in deer yard quality, as the largest trees with the most fully developed and well shaped crowns grow where site index is highest. This results in high softwood crown closure. For northern white cedar, deer yards in New Hampshire averaged 17.4m in site index; non-deer yards averaged 15.1m (Weber et al. 1983). Site indices for northern white-cedar in the UGLR are lower than in New Hampshire deer yards, typically 12m on the best sites, and 5m on poorer sites (Johnston 1977).

Total basal area of a conifer stand was found to be inversely related to winter deer utilization (Weber et al. 1983). Deer apparently seek out canopy openings to benefit from incoming radiant solar energy (Aldous 1941). Openings also provide a potential source of food; saplings and young sprouts often grow in areas receiving direct sunlight (Weber et al. 1983). Conifer stands utilized as deer yards in New Hampshire averaged 49.3m²/ha of total basal area (34.5-67.5), while non-deer yards averaged a higher 57.7m²/ha (42.2-72.8) (Weber et al. 1983).

Softwood crown closure has long been recognized as an important component of winter deer habitat (Verme 1965, Nowosad 1967, Kramer 1970, Blouch 1984). Complete conifer canopy closure, however, is not necessary and may be undesirable for deer yards (Aldous 1941, Weber et al. 1983). Euler and Thurston (1980) found that deer use declined in eastern hemlock stands when crown closure increased beyond 71%. Weber et al. (1983) found deer yards in New Hampshire to be characterized by softwood crown closures averaging 66.7%. Non-deer yards averaged 54.3%. Thus, although high softwood crown closure is important in snow depth and wind velocity reduction, total closure is undesirable as it negates the patchiness associated with food production and solar hot spots previously noted.

Weber et al. (1983) also found stand size to be an important factor in deer yard use. Deer yards in New Hampshire were larger than non-deer yards (mean of 63.6 ha versus 40.7 ha). Large areas are more likely to be located by migratory animals, and support greater numbers of deer.

Conifer species is the final important factor in deer yard quality. Northern white cedar provides the best cover and forage, and is heavily utilized (Blouch 1984). Eastern hemlock, jack pine, balsam fir, and other dense stands of upland conifers are also utilized where cedar or mixed conifer swamps are lacking.

MODEL JUSTIFICATION/APPLICATION

This white-tailed deer HSI model is composed of 4 distinct sub-models:

- (1) FALL/WINTER FOODS
- (2) SPRING FOODS
- (3) SECURITY COVER
- (4) THERMAL COVER

The model assesses the quality of the evaluation area for each sub-model habitat component independently. The final HSI value for deer is based on a most critical limiting factor theorem: whichever sub-model HSI value is the lowest (most limiting) will be the overall HSI value for deer in the evaluation area. The reasoning behind this is that each critical habitat component associated with a sub-model is considered equally important to the well being of deer; thus, a high value for one component(s) cannot compensate for a low value of another.

White-tailed deer home ranges can vary significantly, based on geographic area, climate, annual variation in weather, etc (Marchinton and Hirth 1984). Values reported in the literature ranged from ≤ 59 ha to > 520 ha (reviewed by Marchinton and Hirth 1984). Most deer home ranges have radii of less than 1.6 km, however. Therefore, the unit of evaluation

for this white-tailed deer HSI model is an area of contiguous habitat of $\geq 2.56 \text{ km}^2$ (256 ha).

MODEL 1: FALL/WINTER FOODS (FWF)

Fall and winter foods in the UGLR consist primarily of woody browse, mast, and selected agricultural crops (Rogers et al. 1981, Blouch 1984). This model assumes that browse is a function of availability and the relative nutritional quality of the site (Fig. 27). FALL/WINTER FOOD suitability should be optimal as available ($\geq 1\text{m}$ tall, $\leq 6.3\text{cm}$ dbh) woody stems per hectare reach 5000 (M_1V_1 , Fig. 27). Additionally, the nutritional quality of available browse can be related to site quality. This model assumes that higher quality sites (as assessed by site index for red pine or ecological land type phase (ELTP) will produce higher nutritional quality forage (M_1V_2 , and alt M_1V_2 , Fig. 27). The total suitability index (SI) for woody browse is a mean of the individual values for availability and nutritional quality:

$$SI_{\text{BROWSE}} = (M_1V_1 + M_1V_2)/2.$$

The main mast producing species in the UGLR are the oaks. Acorn production is a product of the number of oak trees present, their size, and the diversity of species present, as acorn production within a species can vary greatly annually (Rogers et al. 1981, Armbruster et al. 1987). This model assumes that acorn production will be optimal when oak basal area is $\geq 7.5\text{m}^2/\text{ha}$ (M_1V_3 , Fig. 28) (Armbruster et al. 1987). Similarly, larger oak trees tend to produce the largest and most consistent mast crops. This model assumes that acorn production will be maximized in trees $\geq 50\text{cm}$ dbh (M_1V_4 , Fig. 28) (Armbruster et al. 1987). Production progressively declines as mean diameters drop below 50cm. Below 25cm, oaks are typically too small to yield good acorn crops; hence, suitability drops to 0.0 (Fig. 28).

Three oak groups occur in the UGLR--the white oak group, the red oak group,

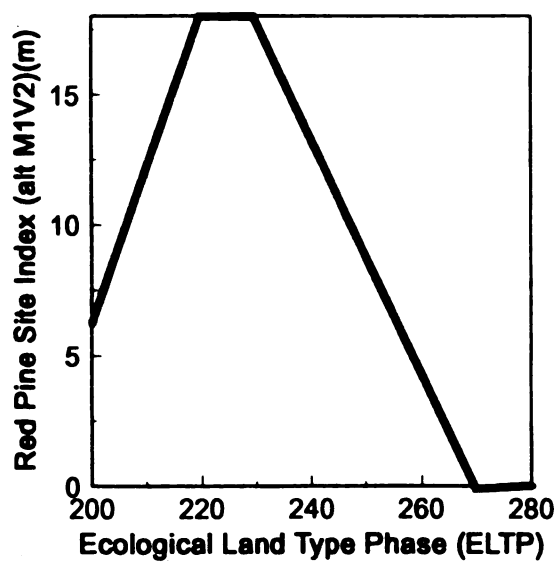
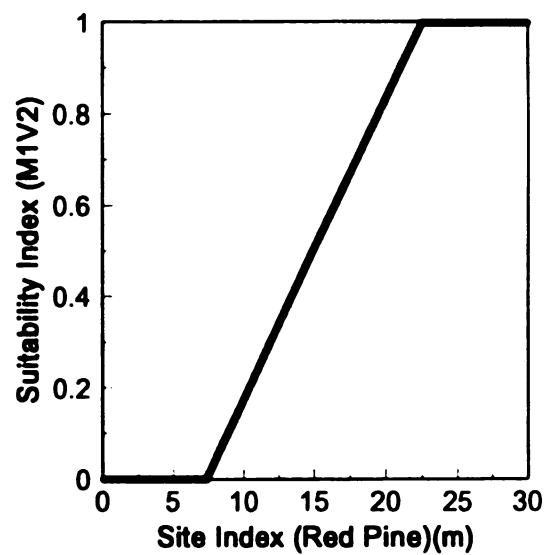
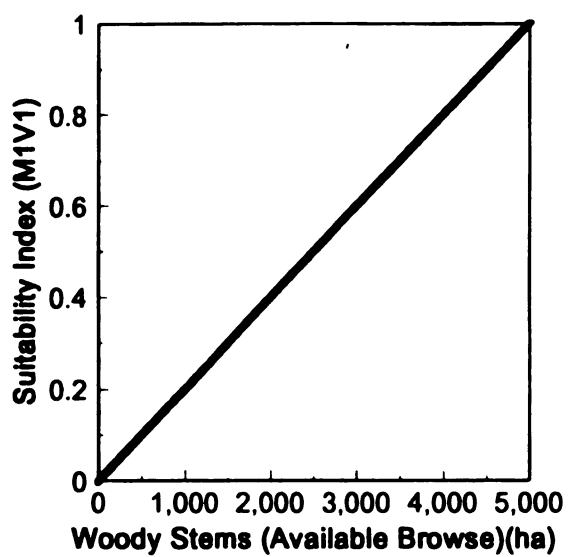


Figure 27. Woody browse variables for FALL/WINTER FOODS.

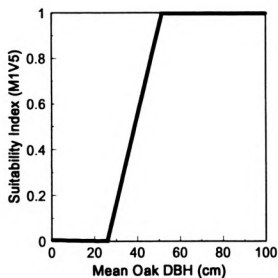
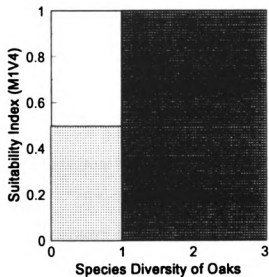
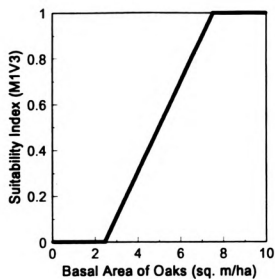


Figure 28. Mast variables for FALL/WINTER FOODS.

and northern pin oak. This model assumes that acorn production will be maximized in areas having > 1 type of oak present (M_1V_4 , Fig 28). Areas with 1 type present are assigned a SI of 0.5. If no oaks are present in an area, the area receives a SI of 0.0 for mast production (Fig. 28). The SI for mast as a deer FALL/WINTER FOOD is then calculated as a reduction function of the above 3 variables:

$$SI_{MAST} = M_1V_3 * M_1V_4 * M_1V_5.$$

This model assumes that browse is at least 4X as important in supplying fall and winter foods as is mast, due to the unpredictability of mast production and the continuous availability of browse (Blouch 1984). Therefore, the FALL/WINTER FOOD SI for a forest stand can be determined by a weighted mean of the individual SI's for browse and mast:

$$SI_{FOR ST} = (4*SI_{BROWSE} + SI_{MAST})/5.$$

Agricultural fields can also be an important source of FALL/WINTER FOODS for deer in certain areas of the UGLR. Suitability of agricultural fields as a deer FALL/WINTER FOOD source is a function of the size of the field, the crop species cultivated, and tillage practices (Thomas et al. 1979, Armbruster et al. 1987). This model assumes that agricultural fields with a maximum width of $\leq 180m$ are entirely available to foraging deer and hence receive a SI of 1.0 (M_1V_6 , Fig. 29). As the maximum width of a field exceeds 180m deer will not fully utilize the area (Thomas et al. 1979); suitability therefore declines. Fields $\geq 360m$ in width will be utilized along their margins only; hence these fields receive a SI of only 0.25.

The crop cultivated also contributes to an agricultural field's suitability as a FALL/WINTER FOOD area. Corn is a highly persistent, highly palatable crop and is assigned a SI of 1.0 (M_1V_7 , Fig. 29). Other crops are less persistent and/or less palatable and are assigned lower SI's (Fig. 29). Additionally, management practices affect the availability

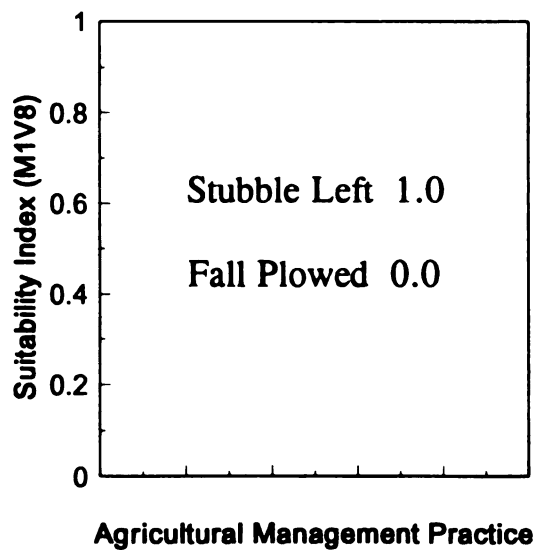
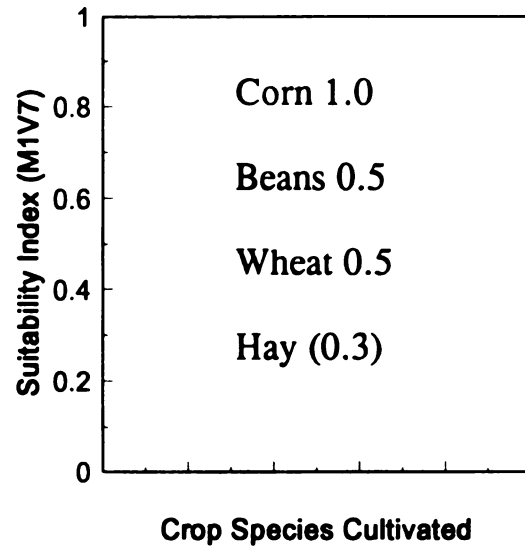
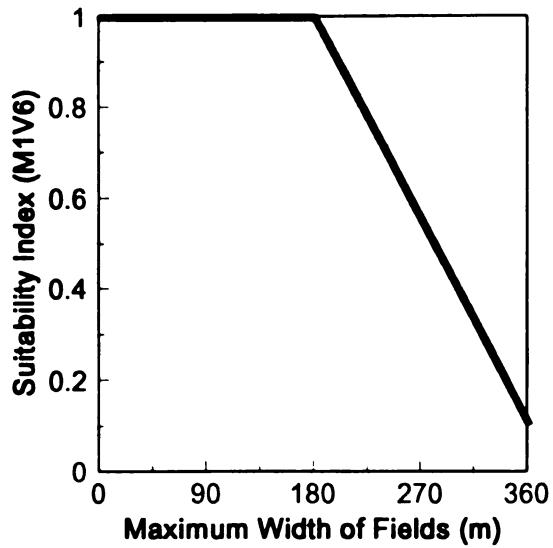


Figure 29. Agricultural field variables for FALL/WINTER FOODS.

of agricultural crops as winter foods. Agricultural fields that are left untilled or under other conservation tillage practices retain maximum crop residues and are assigned a SI of 1.0 (M_1V_8 , Fig. 29). Fall plowed fields have little available crop stubble for deer food, and are assigned a SI of 0.0.

The SI for agricultural crops is calculated as a reduction function in the following manner:

$$SI_{AG} = M_1V_6 * M_1V_7 * M_1V_8.$$

Overall suitability of an evaluation area for deer FALL/WINTER FOOD is determined by summing the individual SI's of all forest stands and agricultural fields (i.e. all potential FALL/WINTER FOOD areas) multiplied by the proportional area the total FALL/WINTER FOOD areas that they represent; i.e. $SI(\text{FWF area } 1) * (\text{area of FWF area } 1 / \text{total area of all FWF types}) + \dots + SI(\text{FWF area } n) * (\text{proportional area of Stand } n)$, or:

$$SI_{FWF} = \text{SUM}_{i=1 \text{ to } n} [(SI_i) * (\text{area of } i / \text{total area in FWF})].$$

This value is then modified by assessing the total percentage of the evaluation area in FALL/WINTER FOOD types (M_1V_9 , Fig. 30). Sixty percent or more of the total evaluation area in FALL/WINTER FOOD types is considered optimal (Thomas et al. 1979); hence, it is assigned a SI of 1.0. Suitability declines as the percentage of the evaluation area in FALL/WINTER FOOD types drops below 60%. The final HSI for FALL/WINTER FOOD for white-tailed deer is then calculated using the following equation:

$$HSI_{FWF} = SI_{FWF} * M_1V_9.$$

MODEL 2: SPRING FOOD (SF)

Spring foods of deer in the UGLR are chiefly a function of forest openings, agricultural fields, and forest ground cover and understory shrubs (Rogers et al. 1981, Blouch 1984). Spring food production and deer utilization of openings is a function of the openings

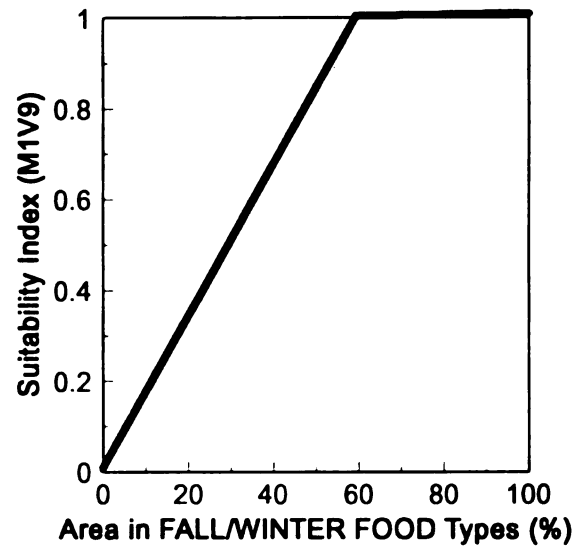


Figure 30. Area function for FALL/WINTER FOODS.

size, the amount of openings in the evaluation area, and the productivity of those openings (Fig. 31). Openings with a maximum width $\geq 180\text{m}$ will not be fully utilized by deer (Thomas et al. 1979), therefore suitability declines as opening widths exceed 180m (M_2V_1 , Fig. 31). Alternatively, size of openings can be substituted for width, with openings being progressively less used as size exceeds 10 ac (alt M_2V_1 , Fig. 31). Total herbaceous productivity, although not quantifying deer foods specifically, provides a relative idea of the amount of deer food potentially available (Crawford and Marchinton 1989). Openings with $\geq 1785 \text{ kg/ha}$ likely will produce ample spring foods (M_2V_2 , Fig. 32). As total productivity drops below 1785 kg/ha, suitability as SPRING FOOD declines (Fig. 32). The SI of herbaceous productivity can also be estimated utilizing a function combining woody cover and herbaceous ground cover (alt M_2V_2 , Fig. 33).

Approximately 10-30% of the evaluation area in openings represents optimal deer habitat (M_2V_3 , Fig. 32) (Armbruster and Porath 1980). Below 10%, insufficient openings are present to allow adequate spring food production. If over 30% of an evaluation area is in openings, cover factors are likely to be limiting (Thomas et al. 1979); however, limitations of cover attributes are assessed using the SECURITY COVER sub-model.

The SI for SPRING FOOD associated with any specific openings is calculated using a reduction function in the following manner:

$$SI_{\text{OPEN}} = M_2V_1 * M_2V_2 * M_2V_3.$$

The contribution of all openings toward SPRING FOOD is determined utilizing an area weighing procedure:

$$SI_{\text{OPEN}} = \text{SUM}_{i=1 \text{ to } n} [(SI_i) * (\text{area of } i / \text{total area in openings})].$$

Agricultural fields can also provide spring deer food in the UGLR. Food production and deer utilization associated with agricultural fields can be assessed by

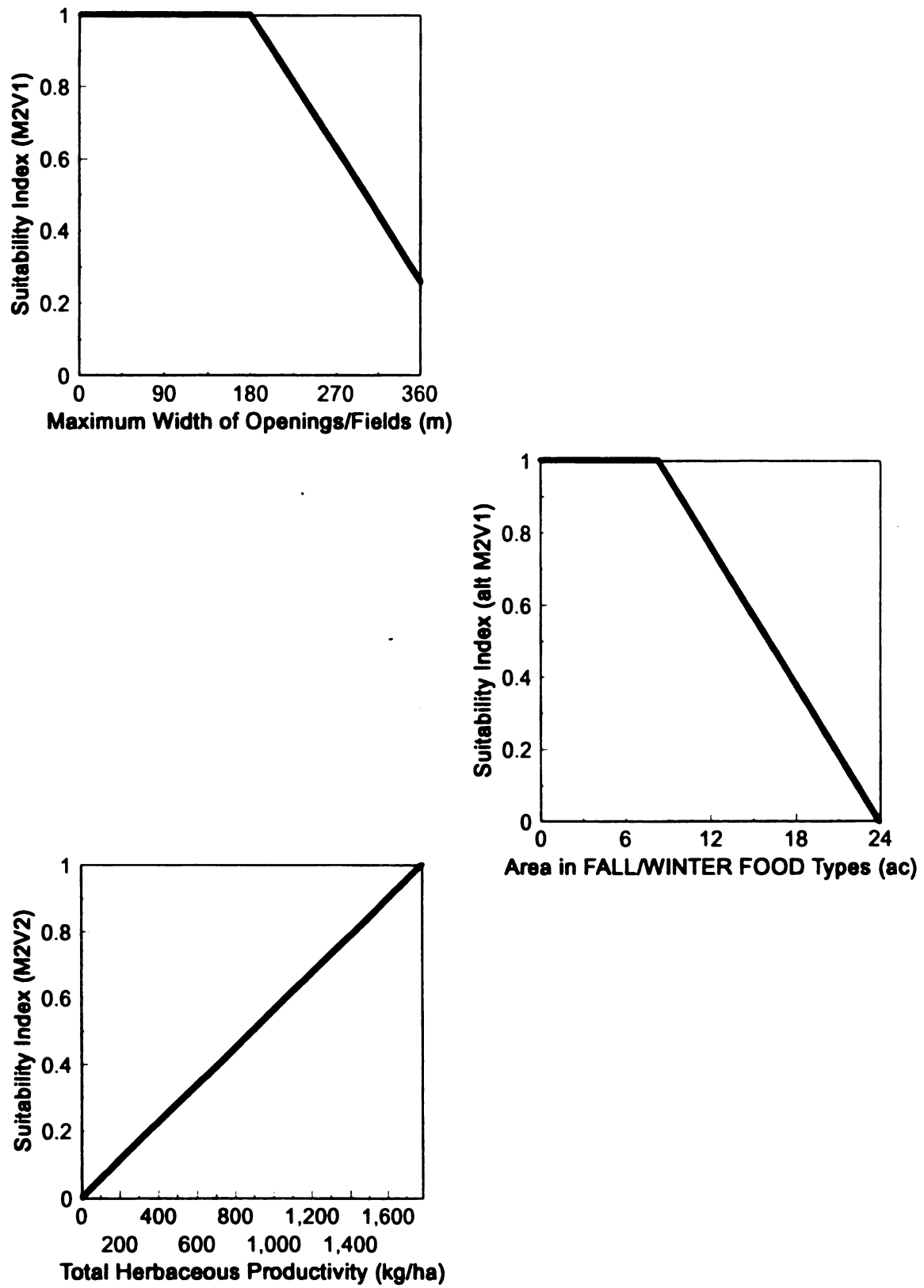


Figure 31. Suitability curves for SPRING FOOD openings components.

alt M2V2

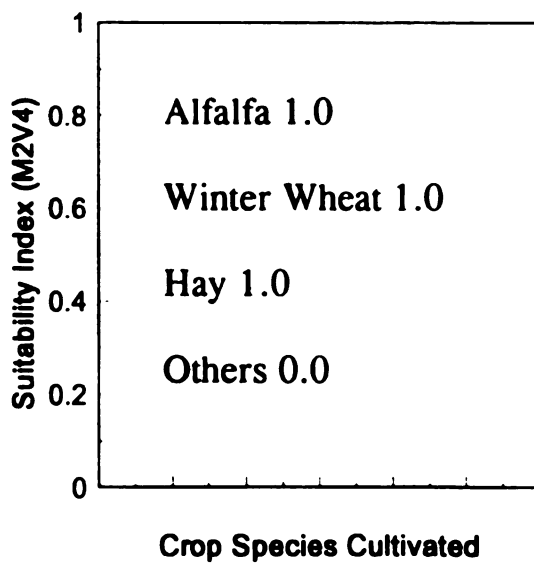
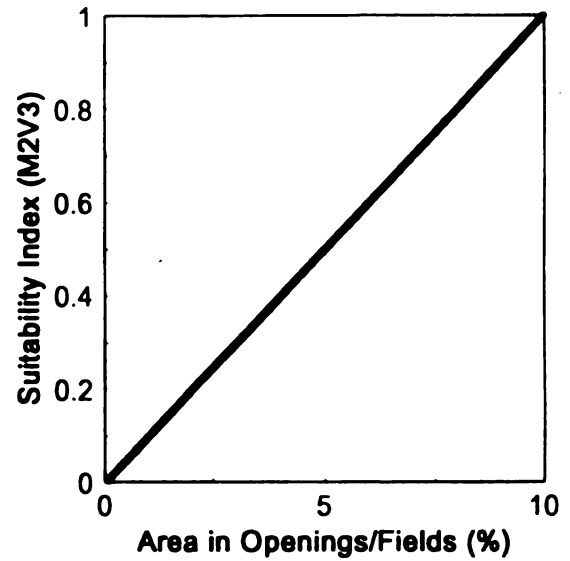
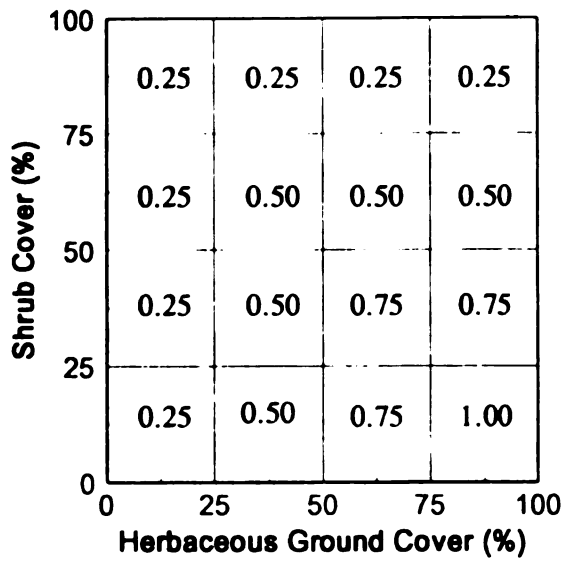


Figure 32. Suitability curves for SPRING FOOD openings components. Suitability of agricultural crops as deer SPRING FOOD.

considering the size of the fields, the percent of the evaluation area in agricultural fields, and the cultivated species. Field size and percent of area in fields is evaluated identically to opening size and percent of area in openings as discussed above (M_2V_1 and M_2V_3 , Figs. 31-32, respectively). Crop species is the third agricultural variable (M_2V_4 , Fig. 32). Three crops likely to be available to deer in the UGLR in spring include alfalfa, winter wheat, and various hays. All are assigned a suitability value of 1.0. (Fig. 32), as all are likely to be green and actively growing during the critical spring period. All other crops have a suitability of 0.0.

The suitability of a field for SPRING FOOD associated with agricultural production is also evaluated using a reduction function:

$$SI_{AG} = M_2V_1 * M_2V_3 * M_2V_4.$$

The overall suitability of an evaluation area for SPRING FOOD associated with agricultural production is determined by area weighing:

$$SI_{AG} = \sum_{i=1}^n [(SI_i) * (\text{area of } i / \text{total area in ag types})].$$

The final component of spring foods, forest ground layer production, can be evaluated by modifying the SI_{BROWSE} determined for FALL/WINTER FOODS by M_2V_5 (Fig. 33). A forest stand with at least 50% ground cover is assumed to contribute to spring food production. Below 50% ground cover too little is contributed, so the SPRING FOOD index value is set at 0.0. The forest floor (FF) contribution to SPRING FOOD is then calculated in the following manner:

$$SI_{FF} = \sum_{i=1}^n [(SI_{BROWSE_i} * M_2V_5) * (\text{Proportional area of } i)].$$

Total SPRING FOOD HSI for the evaluation area is then calculated by summing the 3 SPRING FOOD components in the following manner:

$$HSI_{SF} = SI_{OPEN} + SI_{AG} + (0.2 * SI_{FF}).$$

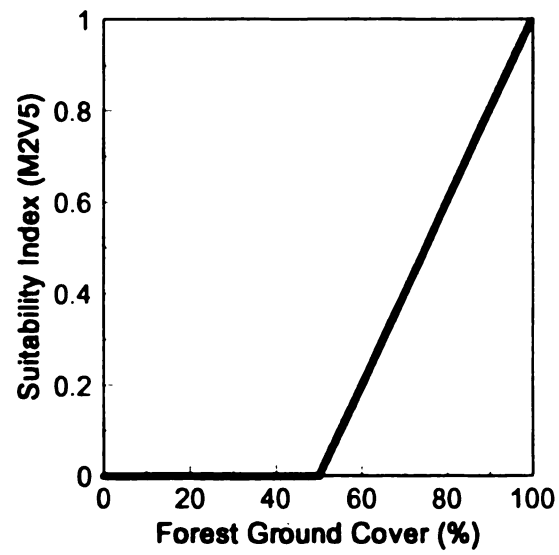


Figure 33. SPRING FOOD suitability of forested stands.

Forest stands are assumed to contribute much less to SPRING FOOD suitability than do openings or agricultural areas; hence, the 0.2 modifier. If the summation of the 3 SPRING FOOD components totals > 1.0 , the value is taken as 1.0.

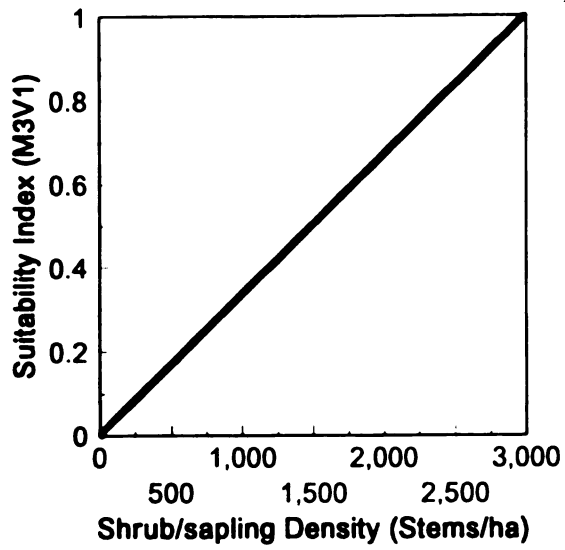
MODEL 3: SECURITY COVER (SC)

Deer need security cover as protection from predators and man (Boyd and Cooperrider 1986). Security cover for deer in the UGLR is principally associated with forested and shrub/sapling stands.

Security cover attributes associated with shrub/sapling stands are primarily a function of the density of woody stems. In the UGLR, shrub/sapling stands with woody densities ≥ 3000 stems/ha will provide optimal security cover and have a suitability value of 1.0 (M_3V_1 , Fig. 34). As mean woody stem densities decrease below 3000/ha, the quality of security cover declines. Patchiness in regeneration/stocking make some areas with a mean of < 3000 stems/ha suitable hiding areas, however, so suitability of a shrub/sapling stand as SECURITY COVER remains greater than 0.0 until woody stem densities decline to 0/ha.

The quality of security cover provided by forested stands (mean overstory dbh ≥ 10.2 cm) is a property of stand density and tree size (Armbruster et al. 1987). Smaller trees require higher densities to produce the same degree of concealment as larger trees. The relationship between tree size (represented by total basal area) and tree density and associated suitability values are shown in M_3V_2 , (Fig. 34).

The third factor important in assessing the SECURITY COVER attributes of an evaluation area deals with the total amount of SECURITY COVER present. Thomas et al. (1979) felt that at least 40% of an evaluation area should be in security cover for optimal deer habitat in the Blue Mountains of Oregon. This model likewise assumes that $\geq 40\%$ of an evaluation area should be in security cover types for deer habitat to be optimal (M_3V_3 , Fig.



Stand Density/Size
Relationship (M3V2)

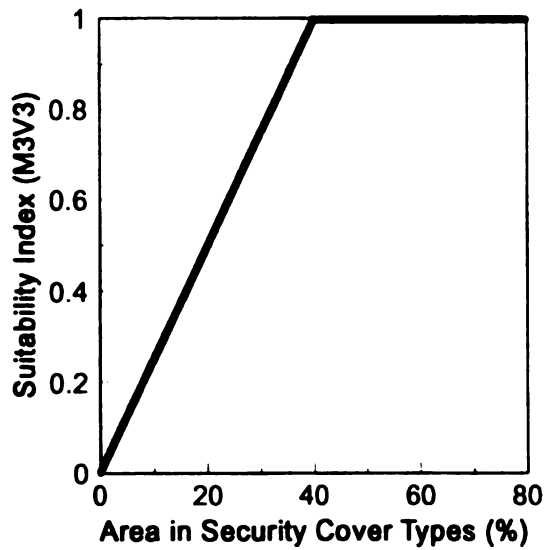
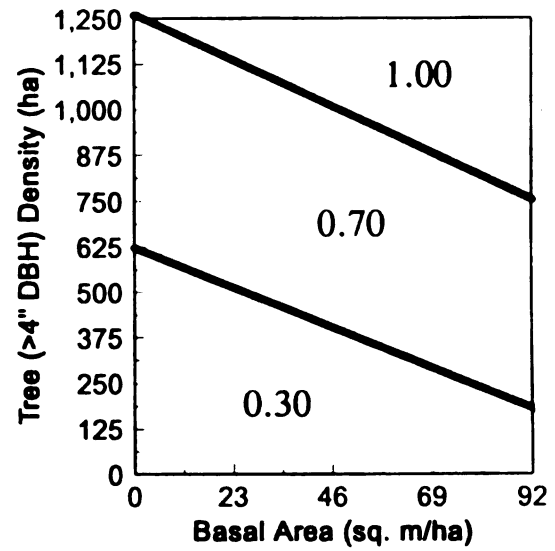


Figure 34. Suitability curves for deer SECURITY COVER variables.

34). As the amount of security cover in an evaluation area drops below 40%, habitat quality declines, as deer lack adequate protection from predation, hunting, and disturbance.

SECURITY COVER in this model is assumed to be an attribute of shrub/sapling stands and forested stands only. Security cover quality is evaluated on a stand by stand basis utilizing M_3V_1 for shrub/sapling stands and M_3V_2 for forested stands. The overall SI for SECURITY COVER for the entire evaluation area is determined by summing the products of each individual stands quality value times the proportion that particular stand contributes area-wise to the area grand total of all SECURITY COVER areas:

$$SI_{SC} = \sum_{i=1}^n [(SI_{STANDi}) * (\text{area of } i / \text{total area in security cover})].$$

The overall HSI for deer SECURITY COVER can then be determined by taking the SI_{SC} value calculated above and modifying it by the percentage of the evaluation area that is in SECURITY COVER types (M_3V_3) (Fig 34):

$$HSI_{SC} = SI_{SC} * M_3V_3.$$

MODEL 4: THERMAL COVER (TC)

This model assumes that thermal cover is necessary and utilized every winter by deer in the UGLR, and that it is the duration of use (and not use itself) which is determined by winter severity. This assumption is undoubtedly valid in the northern parts of the UGLR. It is possible that in southern areas, however, thermal cover may not be needed each year.

Habitat suitability in terms of thermal cover can be evaluated in 2 manners with this model. The first method involves simply defining what constitutes adequate deer thermal cover. The second methodology involves assessing the individual quality of each thermal cover area.

Definition Methodology

This model defines thermal cover as any conifer stand (pole size or larger) with $\geq 75\%$ softwood crown closure that is ≥ 2 ha in size (Thomas et al. 1979). The presence/absence of such stand(s) is determined on the evaluation area utilizing a distance relationship for suitability assessment (M_4V_1 , Fig. 35). If an area(s) satisfying the above definition is located on or within 3.2km of the evaluation unit, the HSI for THERMAL COVER is 1.0. Habitat quality declines as the distance to suitable wintering areas exceeds 3.2 km. Deer in the UGLR will migrate extreme distances to wintering areas--often > 48 km (Dahlberg and Guettinger 1956, Rongstad and Tester 1969, Verme 1973). Such extreme migrations are not indicative of optimal habitat, however. That such migrations do occur, however, accounts for the suitability value of 0.1 applied to all thermal cover areas > 8 km distant.

Quality Assessment Methodology

The second method of thermal cover evaluation involves quality assessment of each individual thermal cover area. This model assumes that 5 variables can adequately evaluate the quality of deer wintering areas:

- (1) Percent conifer canopy closure
- (2) Site index for northern white cedar
- (3) Size of area
- (4) Basal area
- (5) Dominant overstory species

Conifer crown closure is assumed to be optimal at 75% (M_4V_2 , Fig. 35) (Thomas et al. 1979). Below 25% conifer crown closure, a stand is assumed to be unsuitable as deer thermal cover due to loss of snow reduction and wind reduction attributes (Fig. 35).

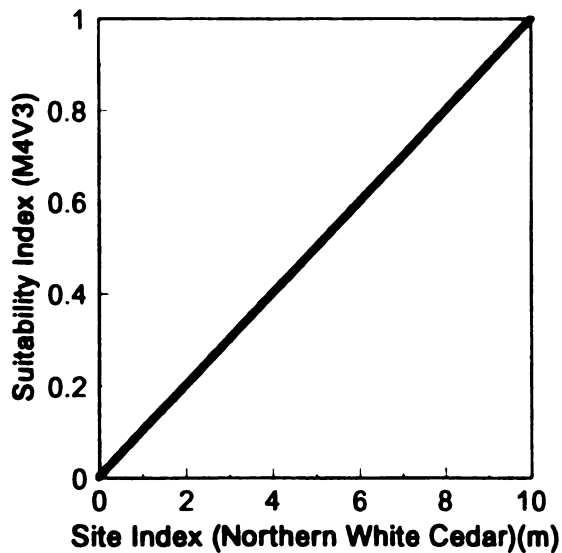
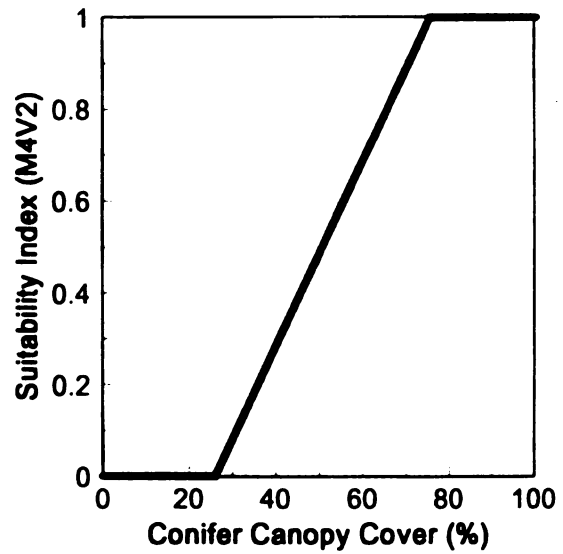
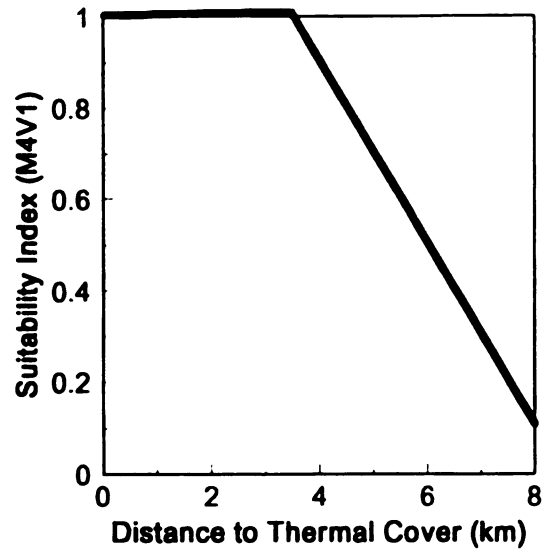


Figure 35. Distance relationship for THERMAL COVER. Suitability curves for THERMAL COVER variables.

Site index gives a relative assessment of the fertility of an area. This model assumes that a site index of $\geq 10m$ for northern white-cedar is optimal, as a site of this fertility should allow large full canopied trees to develop, maximizing snow interception and wind reduction (M_4V_3 , Fig. 35). Below site index 10m, habitat suitability declines as trees are apt to be smaller and less fully crowned.

Deer yards of 50 ha or larger were found to be optimal in New Hampshire (Weber et al. 1983). This model assumes that areas of similar size will be optimal in the UGLR (M_4V_4 , Fig. 36). Areas of this size will winter large numbers of deer and will be easily locatable for migratory deer. Smaller areas are sub-optimal but still provide winter cover (Fig. 36).

Deer benefit from high total basal area in a yard via enhanced wind blocking attributes and forage availability. However, basal areas that are too high result in a lack of small openings (glades), which are important in food production and as "hot spots" where deer can benefit from direct solar radiation (Weber et al. 1983). This model assumes that basal area is optimal between 40 and 60m²/ha ($SI = 1.0$; M_4V_5 , Fig. 36). Above 60m²/ha lack of small openings results in less optimal conditions due to lack of the features noted above. Below 40m²/ha, stands may be insufficiently dense to have optimal wind blockage ability. Stands below 20m²/ha are too open to provide adequate deer THERMAL COVER (Fig. 36).

The final variable in THERMAL COVER quality assessment is the dominant overstory conifer species. Northern white-cedar is considered optimal for deer winter habitat in the UGLR due to its excellent snow and wind limitation characteristics as well as its forage value ($SI = 1.0$; M_4V_6 , Fig. 36) (Blouch 1984). Spruces, hemlock, and balsam fir are less optimal as THERMAL COVER ($SI = 0.7$); they do not provide the excellent forage

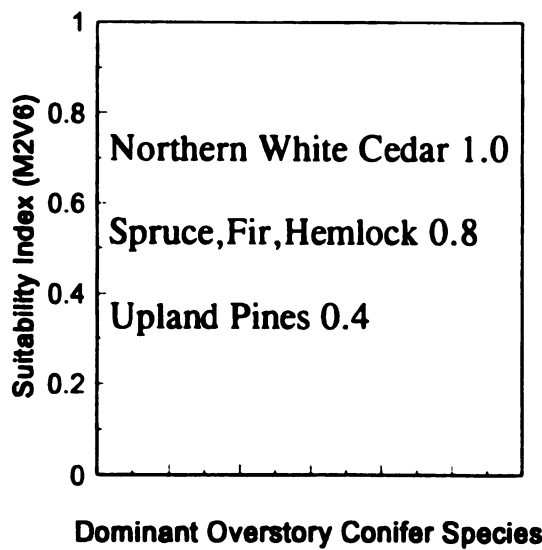
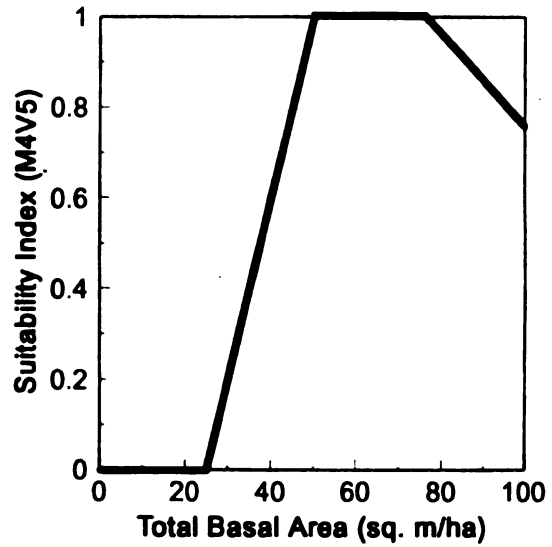
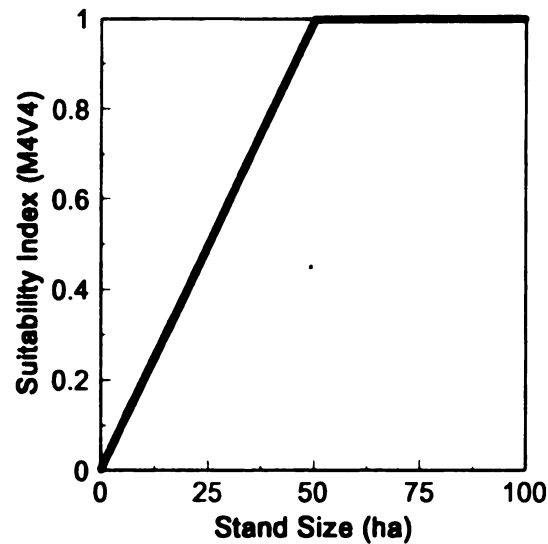


Figure 36. Suitability index curves for THERMAL COVER.

associated with cedar, although their snow and wind blockage characteristics are excellent. Upland pines will be utilized as THERMAL COVER by deer if cedar and other swamp conifers are absent. Upland pines lack the weather modification and/or forage values of the other types, however, and are thus of lower suitability (0.4).

The quality of an area as THERMAL COVER for deer is calculated as a reduction function of the above 5 variables, as each is considered equally important in determining the quality of an area as deer THERMAL COVER. The quality suitability index is thus determined by:

$$SI_{TC} = M_4V_2 * M_4V_3 * M_4V_4 * M_4V_5 * M_4V_6.$$

The HSI for deer THERMAL COVER is then determined by modifying the SI_{TC} calculated above by the distance function (M_4V_1 , Fig. 35) discussed under DEFINITION METHODOLOGY in the following manner:

$$HSI_{TC} = SI_{TC} * M_4V_1.$$

WHITE-TAILED DEER HABITAT SUITABILITY

Habitat suitability for white-tailed deer is determined using a minimum function relationship among the 4 life requisite HSIs described above. The overall white-tailed deer HSI for an evaluation area is the lowest HSI calculated for any of the 4 critical habitat requisites, i.e.,

$$HSI_{WTD} = \min \{HSI_{FWF}, HSI_{SF}, HSI_{SC}, HSI_{TC}\}.$$

Determination of deer habitat suitability in this manner makes the assumption that the 4 critical life requisites modelled are non-compensatory. Additionally, assessing deer habitat suitability in this manner allows identification of which life requisite is most limiting in the evaluation area. This can greatly aid in land management efforts by indicating habitat shortcomings in the evaluation area. Future management efforts could then be tailored to

correct the limiting factors.

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APPENDIX E

Habitat Suitability Index Model for the Eastern Bluebird in Northern Michigan

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HABITAT USE INFORMATION

General

Eastern bluebirds (*Sialia sialis*) range from east of the Rocky Mountains north to southern Canada, and south into El Salvador and Honduras in North America (Bent 1939). Wintering range is usually southward of New York and Ohio in the east, and into southern Kansas and Texas in the west. Eastern bluebirds in Michigan inhabit old fields, and a variety of lumbered and burned vegetation types (Pinkowski 1979).

Serious declines in bluebird populations have occurred in Michigan, as is the case throughout its range. Wallace (1959) noted the rarity of bluebirds statewide, and that they were largely restricted to jack pine (*Pinus banksiana*) areas of the northern counties and to more remote and abandoned or uncultivated farmlands.

Habitat loss and competition with exotic introductions of house sparrows (*Passer domesticus*) and European starlings (*Sturnus vulgaris*) associated with developed areas, have often been cited as the major cause for decline (Bent 1939). Agricultural lands provide vast open areas that are unsuitable habitat and abandoned farmland may revert to unusable, dense second growth timber (Conner 1974). In view of public interest in bluebird conservation over recent years, it is important that key habitat requirements for this species be identified and systematically managed.

Food

The eastern bluebird is primarily insectivorous in feeding habits but will also feed on plant material. Over half of the adult diet is comprised of Orthoptera, Lepidoptera, Arachnida, and Coleoptera (Pinkowski 1979). During the non-breeding season, eastern bluebirds often feed on fruit of staghorn sumac (*Rhus typhina*) and smooth sumac (*Rhus glabra*)(Pinkowski 1979).

Beal (1915) found, from stomach content analysis, that approximately 68% and 32% of the diet consisted of animal and vegetable matter, respectively. Orthoptera accounted for the majority of prey items in every month, comprising more than 50% of the diet in August and September. Coleoptera were the second most important food source, accounting for 22% of the diet. Other important prey items included Lepidoptera, Hymenoptera, and Hemiptera.

Pinkowski (1978) studied feeding of nestling bluebirds in Michigan and found Lepidoptera larvae (32%), Orthoptera (26%), and Arachnida (11%). Early maturing fruit, mainly apples (*Malus* spp.), brambles (*Rubus* spp.), dogwoods (*Cornus* spp.), cherries (*Prunus* spp.), and honeysuckle (*Lonicera* spp.) comprised a small portion of the nestling diet for a seasonally short time.

As summer progressed and vegetation increased in height, ground dwelling prey become increasingly more difficult to catch. In June, bluebirds switch to phytophilous prey species by foraging from perches greater in height (Pinkowski 1979).

Water

No information pertaining to water requirements of eastern bluebirds was found. It is assumed that water requirements are met from food items and free water sources where available.

Cover and Reproduction Requirements

The eastern bluebird is a secondary cavity nester, preferring to nest in cavities of dead trees in park-like situations on dry sites producing sparse ground cover (Pinkowski 1979). Conner (1974) found bluebirds nesting in Virginia clearcuts in standing dead snags averaging 23.6 cm diameter breast height (DBH) and 134 m from woodland edges. Openings to nest cavities ranged from 4-12 cm in diameter. Most nests were constructed by common flickers (*Colaptes auratus*). Conner (1974) observed that suitable nesting habitat can be provided if dead snags are left standing in clearcuts to up to 12 years following harvest.

Pinkowski (1976), on the Huron National Forest of Michigan, found that 89% of nests examined were in dead trees. Tree species used were in proportion to availability with 60% in oaks (*Quercus* spp.), 37% in pines, and one each in white birch (*Betula papyrifera*) and apple. Abandoned woodpecker holes accounted for 66% of the nests, 3% were produced from natural decay and fire. Cavity height averaged 3.6 m, entrance diameter averaged 6.9 cm, cavity depth was 22 cm, and interior diameter averaged 10 cm. Most (80%) excavated cavities were created by common flickers, 15% by hairy woodpeckers (*Picoides villosus*), and 5% by downy woodpeckers (*Picoides pubescens*).

Even-aged timber management eliminates nest sites and foraging perches required by bluebirds (Pinkowski 1976). Pinkowski (1976) recommended that trees > 15-20 cm DBH provide optimum cavity potential, and that management benefitting woodpeckers should also benefit bluebirds.

Interspersion

Eastern bluebirds maintain a breeding territory for mating, nesting, and feeding, and may decrease territory size during the breeding season (Pinkowski 1979). In the Huron National Forest, breeding territories ranged from 0.8-2.9 ha, with a mean of 1.4 ha. Nest

densities averaged 7.9/100 ha. the highest nest densities (9.6/ha) were found in an area that had burned 10-years earlier (Pinkowski 1976).

Special Considerations

Conner (1974) suggested that snags left in clearcuts provide suitable habitat for nesting eastern bluebirds. Of equal importance may be the presence of sparse ground cover and optimal perch distribution so that bluebirds can efficiently utilize drop foraging tactics (Pinkowski 1979).

Studies by Pinkowski (1979) make reference to prey abundance, perch distribution, and ground cover as important habitat variables for bluebirds. Additionally, a study (Pinkowski 1974) demonstrated that management for woodpeckers may be of critical importance. Woodpecker management would apparently rely on basal area and snag availability.

Land management practices involved in perpetuating bluebird habitat include logging and burning. In jack pine stands in northern Michigan, eastern bluebirds have a greater preference for burned areas over timbered tracts (Pinkowski 1977). Land use practices that involve prescribed burning of thinned jack pine will be most beneficial to bluebirds.

HABITAT SUITABILITY INDEX MODEL

Geographic Area

This model was developed for the central and northern lower peninsula of Michigan.

Season

This model was developed to evaluate breeding habitat for the eastern bluebird; late-March through August.

Cover Types

This model is restricted for use in the deciduous forest (DF), evergreen forest (EF), deciduous shrubland (DS), and evergreen shrubland (ES) cover types of undeveloped uplands (terminology follows that of U.S. Fish and Wildlife Service 1981). This model is not applicable to wetland or agricultural cover types.

Minimum Habitat Area

Minimum habitat area is defined as the minimum amount of contiguous habitat required for a species to live and reproduce in an area. Minimum habitat area for bluebirds was not specified in the literature, thus, it is assumed that an area of 0.8 ha must be present or the habitat suitability index (HSI) will equal 0.00.

Verification Level

This model will be evaluated with data collected from the "Ecological Classification System-Wildlife Habitat Model" project on the Huron National Forest, Michigan.

Model Description

Overview

This model evaluates the ability of habitat to meet the cover/reproduction requirements of the eastern bluebird and should be used as an overall indication of breeding habitat suitability. Reproductive requirements are assumed to be the most limiting factor. Therefore, food and cover requirements are assumed to be satisfied by cover types meeting the reproductive requirements. The cover/reproduction component of the model evaluates habitat suitability through measurements of vegetative structure including basal area, number of snags ≥ 20 cm dbh, and percent ground cover.

The following sections provide documentation of the HSI for the eastern bluebird.

Following are sections describing life requisites, definitions of habitat variables, and suitability levels and the relationships between variables.

Cover/Reproduction

Eastern bluebirds are secondary cavity nesters that have been found to nest in snags ≥ 20 cm dbh in low stocked, semi-open vegetation types in north-central Michigan. Optimal habitats are assumed to contain at least 5 snags/ha and 0.5-3 m²/ha basal area with a relatively open, park-like understory. The requirement for basal area can never equal 0.00 due to the tendency of bluebirds to nest in open fields providing a cavity is available. Bluebirds, during the breeding season are almost exclusively insectivorous, preferring to forage in areas with sparse ground vegetation. Optimum habitat is assumed to contain between 60-90% ground cover. Figure 37 illustrates the relationships between the habitat variables and habitat suitability. The suitability index values are related in equation (1).

Equation 1:

$$\frac{[(SI_{Basal} * SI_{Snags} * SI_{Saplings})^{0.33} + SI_{Ground}]}{2}$$

HSI Determination

Habitat quality is expressed as a result of equation 1, with 0.00 and 1.00 representing low and high quality habitat, respectively. If any of the SI values equals 0.00, then the HSI equals 0.00 and the habitat is considered to be unsuitable breeding habitat. The HSI value from equation 1 represent the capability of habitat to support breeding pairs of eastern bluebirds.

Summary of Model Variables

Listed in table 22 are definitions, applicable cover types, and recommended sampling techniques for habitat variables used in the computation of bluebird HSI's.

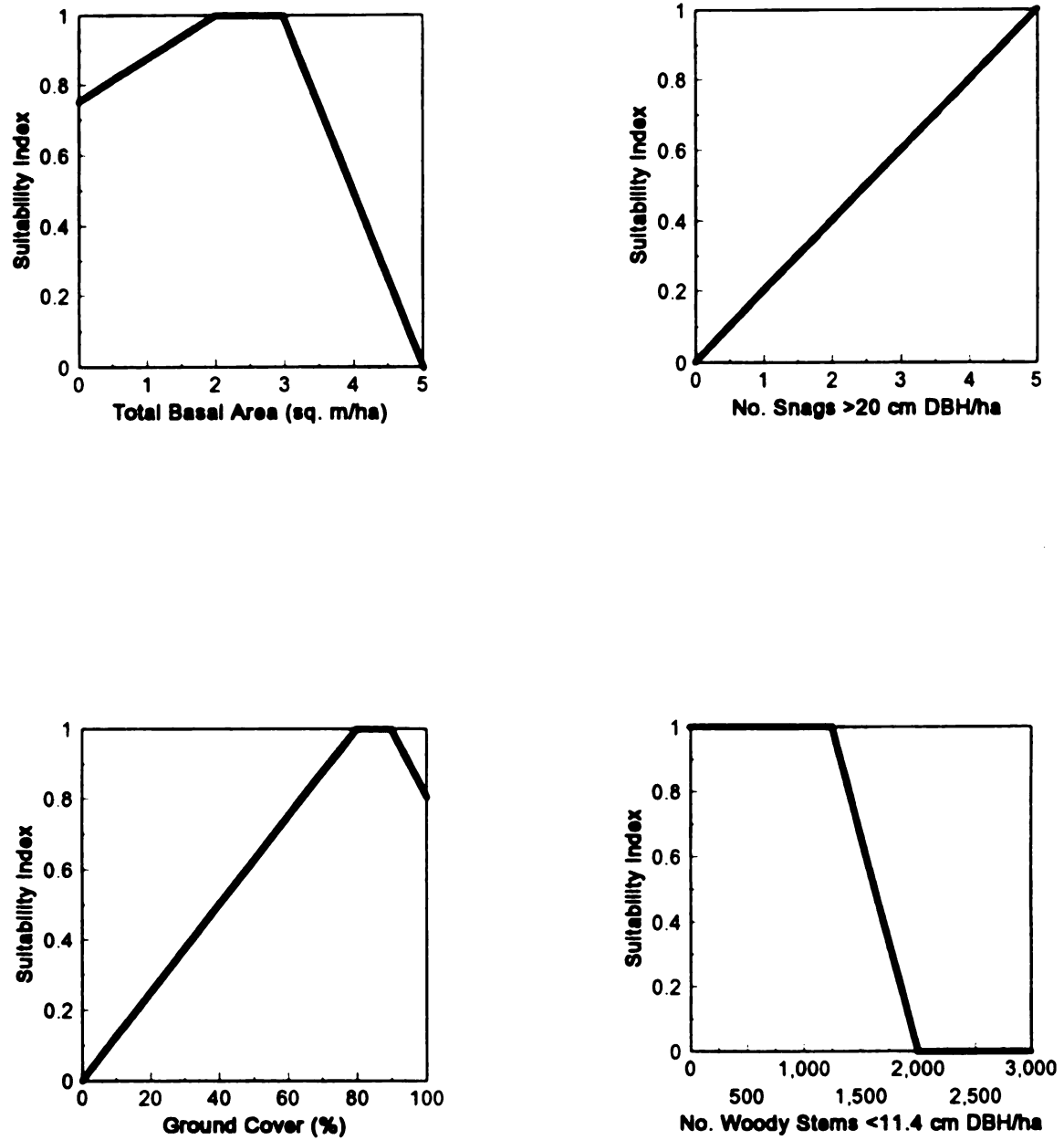


Figure 37. Vegetative variables and habitat suitability index (HSI) relationships for the eastern bluebird.

Table 22. Definitions of habitat variables and associated measuring technique.

HSI Variable & Definition	Cover Types ¹	Sampling Technique
Basal Area Area of exposed stems of woody vegetation if cut horizontally at 1.4 m (4.5 ft) from the ground.	EF,DF,ES,DS	10 x 25 m Plots; Basal Area Prism or Tube
Snags ≥ 20 cm Standing dead trees ≥ 20 cm dbh at 1.4 m (4.5 ft) tall.	EF,DF,ES,DS	10 x 100 m Belt Transects
Saplings The number of woody stems < 11.4 cm (4.5 in) dbh.	EF,DF,ES,DS	2 x 25 m Belt Transects
Ground Cover Percent of the ground that is covered by vegetation ≤ 1 m tall.	EF,DF,ES,DS	Line Intercept

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APPENDIX F

A Habitat Suitability Index Model for the Upland Sandpiper in the Huron National Forest, Michigan

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HABITAT USE INFORMATION

General

The upland sandpiper (*Bartramia longicauda*) is a migratory bird which originally bred throughout the grasslands of the United States and Canada (Bent 1929, Snyder et al. 1987). Currently, upland sandpipers breed from the northeastern coast of the United States to eastern Washington and Oregon and northward to Alaska and south into Oklahoma (Barrows 1912, Bent 1929). The upland sandpiper over-winters in the pampas of South America (Bent 1929).

Formerly abundant across most of the United States and Canada in the mid- to late-1800's, populations declined rapidly after 1900 (Barrows 1912). Factors that caused the population decline were the continuous cultivation of the native prairies (Graber and Graber 1963), market hunting in the United States and Canada (Cooke 1914), and unlimited harvest in South America (Beck 1937). In 1916, the Migratory Bird Conservation Act ended legal hunting of the upland sandpiper in the United States and Canada and populations increased (Mitchell 1967, Kirsch and Higgins 1976). Additionally, upland sandpipers appeared to adapt to changing habitat conditions and nested in cultivated fields when unbroken prairies were no

longer available (Bailey 1930). However, populations have never recovered to the pre-1900 levels. Factors that appear to be restricting post-1916 population growth include habitat changes on the wintering grounds (Osborne and Peterson 1984) and habitat loss in North America, specifically the maturation and replacement of old fields (Tate 1986). Ailes (1980) also suggested that adverse weather conditions during the breeding season may account for annual population fluctuations.

Current breeding populations of the upland sandpiper in the United States are localized, and much of the breeding range has been reduced (Johnsgard 1981, Osborne and Peterson 1984). As of 1977, no upland sandpiper populations were known to be increasing and there were no known improvements in habitat of significant scale (Jurek and Leach 1977). In Michigan, as in other states, prairies have been eliminated (Stearns and Lindsey 1977) but local populations of upland sandpipers appear stable (Wallace 1977). Powell (1981) reported relatively good numbers of sandpipers in 13 Michigan counties.

Foods and Feeding

The upland sandpiper forages extensively on arthropods (phylum Arthropoda) but seeds are also an important food in some locales (Bates 1907, Forbush 1912, McAtee and Beal 1912, Bent 1929). McAtee and Beal (1912) examined 163 stomachs from upland sandpipers and found that 97% of the diet was animal matter. Grasshoppers (Locustidae), crickets (Gryllidae), and weevils (Curculioninae) occurred in greatest quantities (McAtee and Beal 1912).

Vegetable matter in the diet of upland sandpipers consists of the seeds of buttonweed (*Abutilon theophrasti*), foxtail grass (*Setaria* spp.), and sandburs (*Cenchrus* spp.; McAtee and Beal 1912). Forbush (1912) reported that the upland sandpiper fed extensively on crowberries (*Empetrum nigrum*) in Labrador and Bates (1907) reported that waste grain

was an important food in Nebraska.

The use of foraging sites by upland sandpipers is influenced by vegetation height. Pastures that are continually grazed by cattle are preferred over sedge-grass meadows, plowed-seeded fields, ungrazed pastures, and hayfields (Dorio and Grewe 1979, Ailes 1980). Dorio and Grewe (1979) found that the sedge-grass meadow was used as a foraging site only during May (corresponding to low vegetative heights) whereas the grazed pastures were used in May and June. Ailes (1980) reported that feeding areas during late summer were similar to those used throughout the summer. Upland sandpipers used foraging areas that had vegetation heights from 2.5 cm to 15.0 cm, but abandoned foraging areas when vegetation height was over 30.0 cm (Dorio and Grewe 1979). Ailes (1976) had similar results, noting that upland sandpipers foraged in fields with vegetation 5 cm to 10 cm tall and abandoned the fields when vegetation height reached 15 cm.

Upland sandpipers apparently used the grazed pasture-type fields because they were more successful in finding insects in low vegetation (Buss and Hawkins 1939, Bowen 1976, Dorio and Grewe 1979). Additionally, added peripheral visibility in low vegetation may account for the preference (Dorio and Grewe 1979). Fields that are frequently (more than once per year) cropped or mowed provide good foraging areas because the availability of the preferred habitat is extended (Dorio and Grewe 1979).

Water

No information pertaining to the water requirements of upland sandpipers was found in the literature. It is thus assumed that the water requirements of upland sandpipers are satisfied by localized water sources or diet. Therefore, water was not considered as a habitat limiting factor across the upland sandpiper's breeding range.

Cover / Nesting and Incubation Requirements

Nesting activity of the upland sandpiper typically begins about two weeks after the first birds arrive from South America (typically early- to mid-April; Lindmeier 1960, Ailes 1980). Upland sandpipers separate into pairs and select their nesting areas after arrival to the breeding ground (Buss and Hawkins 1939, Bowen 1976), and colonial nesting is not uncommon (Buss and Hawkins 1939). Data suggest that upland sandpipers will re-nest if their first nest is destroyed (Buss and Hawkins 1939, Ailes 1980). In most areas, the principle cause of nest failure is mammalian predation (Kirsch and Higgins 1976, Buhnerkempe and Westemeier 1988), although Ailes (1980) reported high losses from livestock trampling. The typical clutch size is four, however clutches of three and five have been documented (Buss and Hawkins 1939, Lindmeier 1960, Kaiser 1979, Ailes 1980, Buhnerkempe and Westemeier 1988).

In general, upland sandpipers prefer an open, climax prairie-type habitat for nesting that includes native prairie tracts, old fields, pastures, and airports (Buss and Hawkins 1939, Higgins et al. 1969, Kirsch and Higgins 1976, Dorio and Grewe 1979, Kaiser 1979, Ailes 1980, White 1983, Osborne and Peterson 1984, Buhnerkempe and Westemeier 1988). Several researchers have also reported that upland sandpipers nest in wet meadows or marshy areas (Barrows 1912, Johnsgard 1981) and in clearings in spruce forests (Soothill and Soothill 1982).

In order to initiate nesting, upland sandpipers require several specific habitat components. The height of the vegetation is an important factor in nest site selection (Lindmeier 1960, Kirsch and Higgins 1976, Dorio and Grewe 1979, Kaiser 1979, Ailes 1980, Snyder et al. 1987). Kirsch and Higgins (1976), in North Dakota, found 12% of the total upland sandpiper nests in vegetative cover ranging from 0.0 cm to 15.4 cm tall; 62% of the

nests in cover from 15.5 cm to 30.8 cm tall; 25% of the nests in cover from 30.9 cm to 61.5 cm tall; and 1% of the nests in vegetative cover greater than 61.5 cm tall. Similarly, Kaiser (1979) found that vegetation between 12.7 cm and 63.5 cm tall contained 97% of the upland sandpiper nests in South Dakota. Forty-two percent of these nests were in vegetation between 12.7 cm and 33.0 cm tall (Kaiser 1979).

Ailes (1980) found 15% of the upland sandpiper nests in vegetative cover 0 cm to 15 cm tall; 31% in cover 16 cm to 25 cm tall; 54% in cover 26 cm to 40 cm tall; and no nests in vegetative cover that was greater than 40 cm tall in central Wisconsin. Although vegetation height is important at the time of nest initiation, Ailes (1980) found that at the time of hatching, vegetation was 70 cm tall in some areas.

Several researchers have suggested that upland sandpipers require a minimum amount of overhead- and side-concealment around the nests. Vegetation that provides 50% or more overhead concealment appears to be selected by upland sandpipers for nesting (Kirsch and Higgins 1976, Kaiser 1979). Only a small portion of upland sandpiper nests had no side concealment (Buss and Hawkins 1939, Kirsch and Higgins 1976). Kaiser (1979), however, suggested that overhead concealment is more important than side concealment. Although vegetative coverage is important in nest site selection, Lindmeier (1960) demonstrated (via light intensity readings) that coverage surrounding upland sandpiper nests was relatively poor when compared to other ground nesting species.

Kirsch and Higgins (1976), in an attempt to quantify upland sandpiper nest occurrence and vegetative coverage, found that 10% of the nests occurred in poor cover (e.g. overgrazed pastures, feedlots, mowed areas with little regrowth); 76% occurred in fair cover (e.g. standing stubble, moderately grazed pastures, mowed areas with heavy regrowth, undisturbed vegetation on poor soil); 13% occurred in good cover (e.g. lightly grazed

pastures, undisturbed vegetation on good sites); and 1% occurred in excellent cover (e.g. undisturbed tall grass, grass-forb mixtures on good sites).

Dorio and Grewe (1979) suggested that the selection of nesting habitat by upland sandpipers indicated that appearance characteristics of the cover rather than species composition was important. Kirsch and Higgins (1976) noted similar results, finding that upland sandpipers nested in a variety of vegetation including grasses, forbs, and brush. However, upland sandpipers appeared to avoid grass-legume mixtures, presumably because they were too tall for nesting (Kirsch and Higgins 1976, Kaiser 1979, Buhnerkempe and Westemeier 1988).

Buss and Hawkins (1939) suggested that the proximity of suitable loafing and feeding sites to the nest is important. Suitable loafing sites include pastures with low tussocks, prairie meadows with sparse vegetation not taller than the plovers back, or hayfields containing open patches that allow freedom of vision (Buss and Hawkins 1939). Buss and Hawkins (1939) also suggested that the loafing site must be adjacent to the nesting area to make it acceptable, however, Ailes (1976) documented daily movements of approximately 400 m by both male and female during the nesting and incubating period.

A variety of management scenarios provide suitable nesting habitat for the upland sandpiper. Kirsch and Higgins (1976) found that average nest densities were highest on grasslands managed by prescribed burning. Similarly, Buhnerkempe and Westemeier (1988) demonstrated that upland sandpipers selected fields that had been rotary mowed or burned the previous season, however, upland sandpipers did not selectively nest in burned fields past their first year of growth. Bowen (1976) found more nests in unburned pastures than expected based on habitat abundance.

Seeding and management of cool-season grasses appear to be beneficial to upland

sandpipers when given time to diversify (Buhnerkempe and Westemeier 1988). Buhnerkempe and Westemeier (1988) suggested that management of such seeded fields include a three-year rotation of rotary mowing, no disturbance, and prescribed burning.

Agricultural land use patterns and farming practices have important influences on nest site selection and hatching success of upland sandpipers (Higgins et al. 1969). Higgins et al. (1969) found 93% of the upland sandpiper nests in their study in grasses that were either unused by cows or intermittently mowed for hay. Kaiser (1979) suggested that moderate spring grazing in which 20-40% of the current year's growth is removed did not restrict nesting of upland sandpipers. Buss and Hawkins (1939) noted a high percentage of pasture nests were found in tufts of tall grass fertilized by the previous year's cow dung. Typically, few nests are located in annually tilled cropland or in heavily grazed pastures (Buss and Hawkins 1939, Kirsch and Higgins 1976, Ailes 1980).

Cover / Brooding Requirements

Upland sandpipers typically prefer close-cropped, open fields for brood-rearing. Buhnerkempe and Westemeier (1988) observed broods of upland sandpipers in fields of wheat stubble, recently hayed legumes, and on grazed pastures. Dorio and Grewe (1979) and Ailes (1980) observed broods in similar habitats.

Good brood rearing habitat is short in height (0 cm to 10 cm, perhaps up to 20 cm), open, and weedy (Dorio and Grewe 1979, Ailes 1980, Buhnerkempe and Westemeier 1988). Such habitat facilitates movements of the chicks and offers an abundance of insects for food (Buhnerkempe and Westemeier 1988). Broods may also use wind eroded areas within the nesting habitat for foraging, thus upland sandpiper broods will remain in the natal area if suitable brood-rearing habitat is available (Dorio and Grewe 1979, Ailes 1980). However, broods will move considerable distances to reach suitable brood rearing habitat (Dorio and

Grewe 1979, Ailes 1980, Buhnerkempe and Westemeier 1988). Ailes (1980) documented movements by broods of 338 m and 48 m in Wisconsin.

Once the family finds an acceptable brood rearing habitat, daily movements become short (< 100 m) and within 20 ha (Buhnerkempe and Westemeier 1988). Different family units will share brood habitat (Ailes 1980).

Interspersion

No quantitative data were available on the minimum area requirements of upland sandpipers. Bowen (1976) found that nesting pastures ranged in size from 7.3 ha to 40.5 ha. Although not specified, Bowen's findings apparently did not include areas for feeding and loafing, which are important considerations in nest site selection. Ailes (1976) found that home ranges of male and female upland sandpipers was 8.5 ha and 85.6 ha, respectively. Presumably, upland sandpipers were obtaining their life requisites within 86 ha, thus this model assumes that upland sandpiper habitat requirements can be obtained within 86 ha of contiguous cover type. Typically, the vegetation components necessary during the breeding season can be found within one cover type.

White (1983) suggested that nesting fields should be large and unbroken, with a low vegetation "edge" rating. Upland sandpipers will nest near (6.0 to 23.4 m) field edges, however, large amounts of edge are not preferred (Dorio and Grewe 1979, White 1983).

Buss and Hawkins (1939) reported a nesting density of one nest per 0.60 ha. Other researchers have reported densities of one nest per 6.23 ha; one nest per 12.9 ha; and one nest per 21.3 ha (Kirsch and Higgins 1976, Ailes 1980, Snyder et al. 1987, respectively). Thus, nest densities are variable and are probably related to nesting habitat quality.

Bowen (1976) found that upland sandpiper populations were spatially clumped during the nesting season. Buss and Hawkins (1939) also suggested that upland sandpipers

were colonial nesters. The relatively high nesting success of upland sandpipers (often greater than 50%) may be related to social behaviors of mobbing, vocalizations, and the tendency to concentrate nests (Bowen 1976, Kaiser 1979).

Special Considerations

Several researchers have suggested the importance of perch sites for upland sandpiper breeding and security (Bent 1929, Buss and Hawkins 1939, White 1983, Snyder et al. 1987). Upland sandpipers use a variety of perch sites, including fence posts, telephone poles, stumps, and rocks (Bent 1929, White 1983), thus, it is unlikely that preference for a particular type of perch limits breeding habitat suitability.

HABITAT SUITABILITY INDEX (HSI) MODEL

Geographic area. This model was developed for application within the Huron National Forest located in northeastern Michigan.

Season. This model was developed to evaluate the breeding season (typically mid-April through August) habitat of the upland sandpiper.

Cover types. This model was developed to evaluate the habitat quality required by the upland sandpiper in the following cover types: Cropland (C) and Pasture and Hayland (PH) cover types of agricultural and built-up lands; Grasslands (G) and Forbland (F) cover types of undeveloped uplands; and Herbaceous Wetland (HW) and Shore, Bottom Wetland (SW) cover types of wetlands (U.S. Fish and Wildlife Service 1981).

Minimum habitat area. Minimum habitat area is defined as the minimum amount of contiguous habitat that is required before a species will live and reproduce in an area. A quantitative assessment of the minimum amount of contiguous habitat necessary for upland sandpiper breeding was unavailable. However, estimations could be made from home range data. If the amount of contiguous habitat is less than 86 ha, the suitability index is assumed

to be 0.

Verification level. This model will be verified using the data from the Ecological Classification System (ECS) developed in the Huron-Manistee National Forest in north-central Michigan.

Model Description

Overview. This model will evaluate habitat suitability for the upland sandpiper during the breeding season. Components of the model include foraging, reproductive, and brood-rearing requirements, and it is assumed that measurements of these components can predict the habitat suitability of an area. Foraging and brood-rearing components were evaluated for habitat suitability through measurements of vegetation heights. The reproductive components evaluate habitat suitability through measurements of vegetative heights and vegetative coverage.

The following sections provide documentation of the HSI model for the upland sandpiper. Habitat variables of each life requisite and the interrelationships between habitat variables are discussed.

Cover / nesting and incubation. Upland sandpipers prefer to nest in short, open grassland vegetation types. Important habitat variables that influence the selection of nesting sites by upland sandpipers include vegetation height and vegetation coverage.

Upland sandpipers prefer to initiate nesting in vegetation that is 15 cm to 30 cm tall. Nest initiation occurs in vegetation that is 0 cm to 15 cm tall, but these heights are not preferred. Similarly, nest initiation by upland sandpipers occurs in vegetation from 31 cm to 60 cm tall, but these sites are selected less frequently. Upland sandpipers will not initiate nesting in vegetation that is taller than 60 cm.

Upland sandpipers require at least 50% overhead vegetation coverage of the nest,

however, complete coverage is not preferred. Optimal nesting cover contains a diversity of grasses and forbs. Habitat containing brush clumps will be used for nesting by upland sandpipers but brushy areas are typically not preferred. Side concealment is of minor importance in nest site selection.

The minimum area of habitat required for upland sandpiper nesting is 86 ha. Within this area, colonial nesting is not uncommon. The most concentrated nest density reported in the literature was one nest per 0.607 ha. Nesting fields should be relatively open and shaped to minimize edge.

Figure 38 illustrates the relationships between habitat variables and habitat suitability. The relationship of the habitat suitability index values are depicted as the geometric mean between nesting height and cover:

$$HSI = (\text{Vegetation Height Nesting SI} * \text{Overhead Coverage SI})^{0.5}$$

Ecological land type phases (ELTP's) may be useful predictors of habitat suitability for upland sandpiper nesting. For example, the lower ELTP'S (e.g. the 01's to the 13's) are typically less fertile sites and would therefore provide sparser vegetation than a field found on a higher ELTP. Thus, the lower ELTP's may not provide the necessary vertical concealment required by nesting upland sandpipers. Further research is needed to correlate ELTP's and habitat suitability for the upland sandpiper.

HSI Determination

All habitat variables are equally weighted in the HSI model. The model will generate a habitat suitability index between 0 and 1, with 0 representing no suitable habitat and 1 representing optimal breeding habitat for the upland sandpiper. A summary of the vegetation variables and recommended sampling techniques is presented in Table 23.

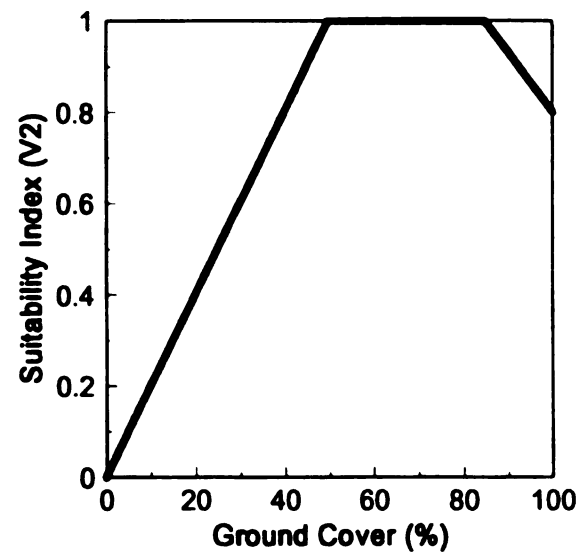
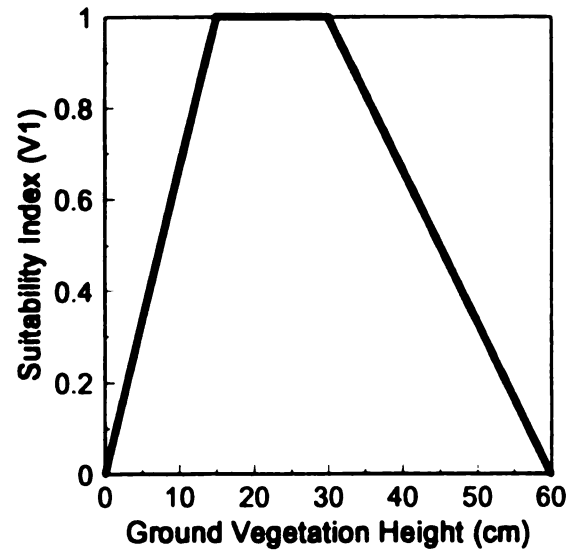


Figure 38. Suitability index graphs for the upland sandpiper.

Table 23. Summary of model variables, associated vegetation cover types, and suggested sampling techniques for habitat suitability index determination.

Variable (Definition)	Cover Types	Suggested Techniques
V₁ Vegetation Height Nesting <i>-the average vertical distance from the ground surface to the dominant height stratum of the herbaceous vegetative canopy during spring.</i>	C, PH, G, F, HW, SW	Graduated Rod
V₂ Percent Overhead Coverage <i>-the percent of the ground that is shaded by a vertical projection of non-woody vegetation.</i>	C, PH, G, F, HW, SW	Line Intercept Domin-Krajina Plot Daubenmire Plot

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APPENDIX G

Habitat Model - American Redstart (*Setophaga ruticilla*)

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HABITAT USE INFORMATION

General

The American redstart inhabits a variety of woodland vegetation types, principally second growth and mature deciduous, mixed and, occasionally, coniferous forests (Hickey 1940, Baker 1944, Bono 1957, Ficken and Ficken 1967, Webb et al. 1977, Sabo 1980). It is a breeding bird of the Eastern United States and Canada but occurs in openings in the boreal forest zone of Canada almost to the Pacific coast (Peterson 1980). Most American redstart over-winter in the West Indies (Bent 1963).

Whelan (1987) suggested that tree species may be an important determinant of habitat selection by insectivorous birds. Accordingly, the American redstart has been reported to favor certain tree species within individual stands for nesting and foraging (Holmes and Robinson 1981, Morris and Lemon 1988a, 1988b). There is, however, no consistency in selection of tree species over its entire range or even among stands in a single forest (Purcell 1987).

Food

American redstarts are primarily insectivores that alter feeding strategies between stands (Ficken 1962, Maurer and Whitmore 1981). Hovering, gleaning, flush-chasing, and hawking are common tactics employed by American redstart for foraging (Sherry 1979,

Maurer and Whitmore 1981, Robinson and Holmes 1982). American redstart commonly forage between 3 and 18 m (10-60 ft) (Sherry 1979, Seidel and Whitmore 1982) with the 12 to 15 m (40-50 ft) range being most frequently utilized (Holmes et al. 1979). Invertebrates within the orders Homoptera, Diptera, Hymenoptera, and Coleoptera are foraged upon more often than those in Lepidoptera and Arachnida (Sherry 1979, Robinson and Holmes 1982). Robinson and Holmes (1982) found that Homoptera comprised 35% of the American redstart spring diet in New Hampshire.

Wiens (1974) found that summer food resources do not limit bird communities. Morse (1973) noticed the mean abundance of food closely correlated with the food requirements of warblers at different stages of the breeding cycle. This study also revealed that, although food abundance fluctuated over years, bird nesting success remained high (Morse 1973). Therefore, in this model, food is not considered a limiting factor for the American redstart.

Water

No information on water requirements was recorded in the literature. It is assumed that the insectivorous feeding strategy minimizes the need for free water.

Cover

The American redstart is primarily a bird of mature forest interiors (Ficken 1962), though it often appears elsewhere (Ficken and Ficken 1967). Baker (1944) found the preferred habitat of American redstart in Michigan to be second growth maple (*Acer* spp.). Collins et al. (1982) noted that the American redstart occurred predominately in aspen (*Populus* spp.) and birch (*Betula* spp.) forests or along edges of other deciduous forests in Minnesota.

Pure coniferous stands are seldom used by American redstart (Capen 1979, Sabo

1980). Morse (1973) found that American redstart males on spruce-clad (*Picea* spp.) islands usually are unsuccessful at nesting, failing in most cases to even attract a mate. Sabo (1980) determined an upper threshold of conifer canopy coverage for American redstart to be 70%. In Minnesota, American redstart were found on sites with very low amounts of coniferous vegetation (Collins et al. 1982).

Reproduction

American redstart enter their breeding range in late April and early May (Bent 1963). They construct a cupped nest in which the nest stands firmly upright, supported mainly from below (Morris and Lemon 1988b). Clutch size averages near 3.5, and normally only one brood per pair is raised each year (Whitcomb et al. 1981, Morris and Lemon 1988b).

Many forest-associated birds, such as the American redstart, typically nest in small openings created by windthrow trees and other natural disturbances in the interior of large wooded areas (Brittingham and Temple 1983). Morris and Lemon (1988b) found that nest placement within each tree species seemed to be influenced by the requirements for the structural support of the nest and reflected differences in branching geometry of the tree species. The most common nest site of the American redstart is an upright, 3- or 4- pronged crotch of a dead or live hardwood sapling, such as a maple, ash (*Fraxinus* spp.), or birch (Bent 1963). Morris and Lemon (1988b) found the average height of American redstart nests to be 2-5 m (6.5-16.5 ft) above the ground, depending on the species of the tree.

Interspersion

Home range size for American redstart average about 0.4 ha (1 acre) or smaller (Table 24). American redstart aggressively defend their territories during the breeding

Table 24. Average home range sizes of the American redstart over its breeding range.

Size (ha)	Location	Study
1.85	New York	Hickey 1940
0.59	Ohio	Sturm 1945
0.40	Ohio	2Sturm 1945
1.06	New Hampshire	Sherry 1979
2.45	New York	Samson 1979
1.98	. ¹	Ficken 1962

¹Location not given.

season (Morse 1973, Sherry 1979), but on 2 occasions their boundaries were observed to break down when young were being fed in the nest (Kendeigh 1945).

Numbers of American redstart seem to be positively correlated with forest area (Stauffer and Best 1980) and require extensive tracts of forest to initially become established (Whitcomb et al. 1981). Temple (1986) found that American redstart only occurred in forest fragments > 100 ha (247 ac). Stauffer and Best (1980) noted that a reduction of wooded or shrub habitat along a riparian zone to less than 200 m (656 ft) in width reduced or even eliminated American redstart populations in Iowa.

HABITAT SUITABILITY INDEX (HSI) MODEL

The following model was developed for the American redstart as a method to objectively assess habitat quality. The model is intended to measure habitat over their entire breeding range during the breeding season (May - August)(Bent 1963). It is based on the assumption that American redstart habitat can be accurately quantified using the following 5 variables:

- V1: Percent Canopy Cover
- V2: Percent of Canopy Cover that is Coniferous
- V3: Number of Trees per Hectare
- V4: Number of Saplings per Hectare

This model can be used to evaluate habitat in deciduous, coniferous and mixed forests.

Overview

Optimal habitat for feeding and reproducing American redstart is characterized by large tracts of second growth deciduous forest with numerous pole-sized trees and 60-85% canopy closure. Sherry and Holmes (1989) cited 3 observations that indicate that American redstart prefer deciduous stands: (1) older male American redstart, which are normally the first returning migrants (Bent 1963), settle in such stands consistently year-after-year, (2) yearling American redstart occur randomly in both hardwoods and mixed conifer hardwoods depending on American redstart population densities, and (3) such stands are routinely the first stands to be settled by American redstart each spring. Morse (1973) noted that American redstart tended to utilize coniferous foliage for nesting when spruce-woods warblers (ie. black-throated green (*Dendroica virens*), blackburnian (*D. fusca*), yellow-rumped (*D. coronata*), and magnolia warblers (*D. magnolia*)) were not present. It was also noted, however, that American redstart foraged heavily within the limited deciduous foliage in the woods, indicating the possibility of the spruce-woods warblers limiting the presence of American redstart in coniferous forests (Morse 1973).

This model assumes the presence of spruce-woods warblers in any forest of high conifer cover. Results from a New Hampshire study showed that American redstart were located in regions predominated by northern hardwoods and were absent when conifer cover reached or exceeded 70% (Sabo 1980). Therefore, any forest with > 70% conifer cover receives a zero value for V2 (Fig. 39). Typical American redstart habitat in Minnesota contained only 4.7% coniferous vegetation (Collins et al. 1982), whereas in New Hampshire American redstart occupied habitats with 38% conifer vegetation (Sabo 1980). Optimal habitat for the American redstart has $\leq 40\%$ conifer cover, and receives a value of 1.0.

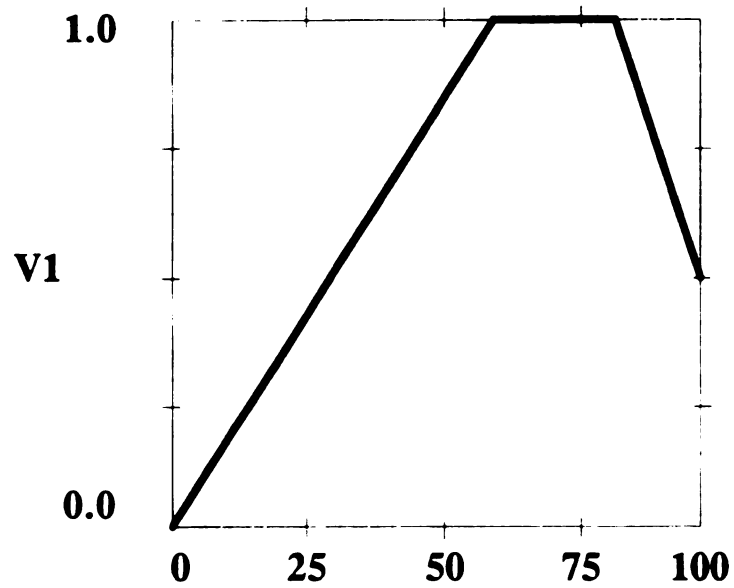
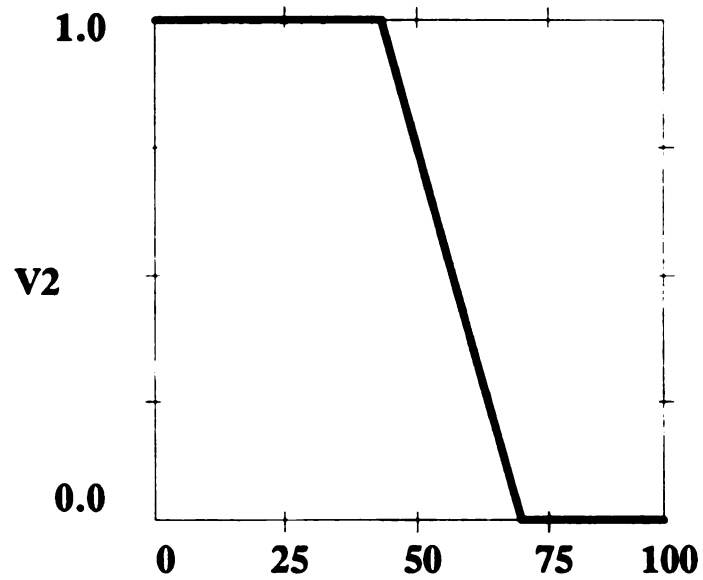
V1: Percent Canopy Cover**V2: Percent of Canopy Cover that is Coniferous**

Figure 39. Percent canopy cover variables for American redstart.

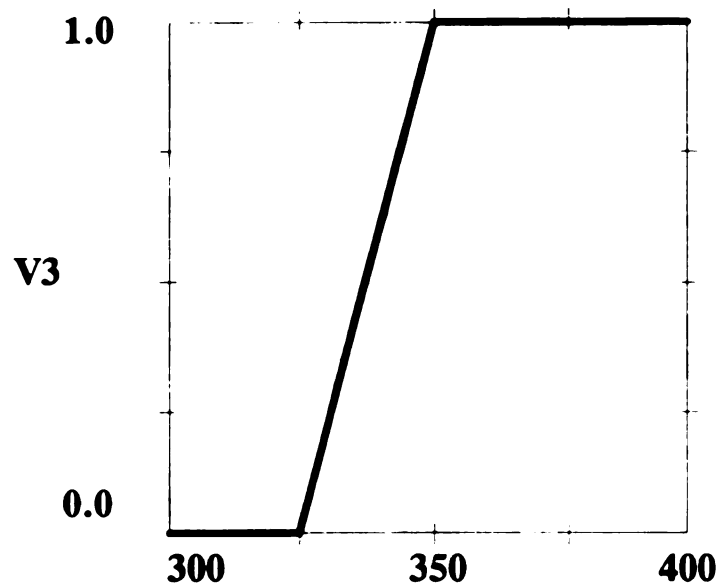
American redstart typically utilize small openings in the forest canopy for nesting (Brittingham and Temple 1983) and feeding (Seidel and Whitmore 1982). Vegetation structure at a particular site provides both opportunities and constraints on the ways that birds can successfully search for and capture prey (Robinson and Holmes 1982). This may be one explanation for the observed tree species preference (Whelan 1987); American redstart concentrate their foraging where food resources are most abundant or available (Holmes and Robinson 1981). Holmes and Robinson (1981) found that yellow birch (*Betula alleghensis*) consistently had more potential prey per sample unit than sugar maple (*Acer saccharum*) and American beech (*Fagus grandifolia*), which were less utilized than the yellow birch.

Similar studies indicate a clear preference of American redstart for areas with a canopy cover between 60 and 85% (Bond 1957, Webb et al. 1977, Sabo 1980, Freedman et al. 1981, Collins et al. 1982). Densities of American redstart dropped rapidly when canopy cover either exceeded 85% or were below 60% (Bond 1957). Thus, any forest with a canopy closure within the range of 60-85% receives a value of 1.0 for V1. The variable value declines outside this range (Fig. 39).

Freedman et al. (1981) found that plots thinned to 460 trees/ha held far fewer American redstarts per unit area than did the 1400-1900 trees/ha uncut plots. Bond (1957) noted that American redstarts in Wisconsin were predominantly found in stands with 350-375 stems/ha. The frequency of American redstarts decreased steadily from stands with 350 stems/ha to stands with 325 stems/ha (Bond 1957). Therefore, stands with ≥ 350 stems/ha receive a 1.0 for V3 and the value declines to 0.0 at 325 stems/ha (Fig. 40).

Purcell (1987) found the number of maple saplings and the basal area (BA) of beech to be significant factors in determining American redstart habitat preference. Collins et al. (1982) discovered the American redstart to be the most dependant species of its guild on

V3: Number of Trees per Hectare¹⁸⁴



V4: Number of Saplings per Hectare

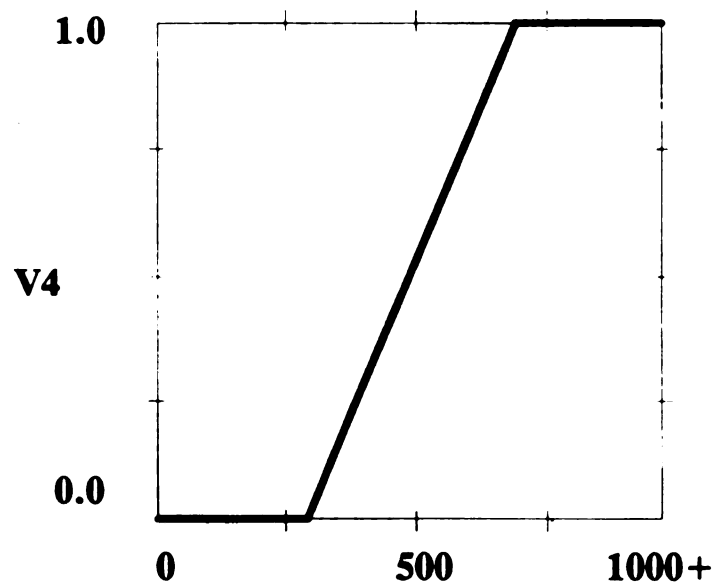


Figure 40. Forest stand description variables for American redstart.

secondary forests with abundant low vegetation on numerous small deciduous trees. Bond (1957) found that the highest densities of American redstart occurred in stands with ≥ 650 saplings/ha (260/ac). Therefore, stands with ≥ 650 saplings/ha (260/ac) receive a 1.0 value for V4 (Fig. 40). Bond (1957) also found that stands with < 300 saplings/ha (120/ac) did not support American redstart, and therefore receive a 0.0 value for V4.

In the eastern and north central forest regions of the United States, many species of birds are dependent on extensive forest systems (Robbins 1979), and consequently are considered to be sensitive to habitat fragmentation (Whitcomb et al. 1981). These species generally exhibit certain characteristics (Whitcomb et al 1981): they are long-distance migrants that winter primarily in the New World tropics; they are obligate inhabitants of forest interior; they build nests in the open rather than in the protection of cavities; they raise only a single brood of young each year; and they have a comparatively small clutch size. The American redstart is listed as such a species (Whitcomb et al. 1979).

MODEL APPLICATION / HSI DETERMINATION

The overall HSI value for any forest stand in the American redstart breeding range can be calculated by determining the value for each of the 4 variables described above (Figs. 39-40), and then using the following equation to compute an index to habitat quality:

$$HSI = (V1 * V2 * (V3 * V4)^{0.5})^{0.25}$$

Variables V3 and V4 are combined because together they sufficiently describe the structure and density of the forest, while the canopy cover variables (V1 and V2) adequately describe the overall canopy structure and composition. The averaging of the pair of combined variables with the canopy (V3, V4) will generate a value between 0.0 and 1.0, where 0.0 represents unsuitable habitat and 1.0 optimal habitat.

This model may be verified in the future by relating the densities of American

redstart to the HSI values for different forest stands. A positive relationship between American redstart density, productivity and HSI values needs to exist for the model to be valid.

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